An Independent Scientific Assessment of Well Stimulation in California

Volume II

Potential Environmental Impacts of Hydraulic Fracturing and Acid Stimulations

July 2015

CCST
CALIFORNIA COUNCIL ON SCIENCE & TECHNOLOGY

BERKELEY LAB
Lawrence Berkeley National Laboratory

An Independent Scientific Assessment of Well Stimulation in California

Volume II

Potential Environmental Impacts of Hydraulic Fracturing and Acid Stimulations

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CCST is a non-profit organization established in 1988 at the request of the California State Government and sponsored by the major public and private postsecondary institutions of California and affiliate federal laboratories in conjunction with leading private-sector firms. CCST's mission is to improve science and technology policy and application in California by proposing programs, conducting analyses, and recommending public policies and initiatives that will maintain California's technological leadership and a vigorous economy.

Note

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Acronyms and Abbreviations

AACOG	Alamo Area County of Governments
ACGIH	American Conference of Governmental Industrial Hygienists
ACS	American Chemical Society
ACToR	Aggregated Computational Toxicology Resource
ADSA	Axial Dimensional Stimulation Area
ANSS	Advanced National Seismic System
APCD	Air Pollution Control District
API	American Petroleum Institute
AQMD	Air Quality Management District
ATSDR	Agency for Toxic Substances and Disease Registry
ATV	All-Terrain Vehicles
AXPC	American Exploration and Production Council
bbl	Barrels
BC	Black Carbon
BLM	Bureau of Land Management
BLS	(U.S.) Bureau of Labor Statistics
BO	Biological Opinion
BOD5	Biological Oxygen Demand
BTEX	Benzene, Toluene, Ethylbenzene, and Xylenes
C_2H_6	Ethane
CALGEM	California Greenhouse Gas Emissions Measurement program
CalOSHA	California Occupational Safety and Health Administration
CalWIMS	California Well Information Management System
CARB	California Air Resources Board
CAS	Chemical Abstracts Service
CASRN	Chemical Abstracts Service Registry Number
CDFW	California Department of Fish and Wildlife
CDPH	California Department of Public Health
CEIDARS	California Emission Inventory Development and Reporting System
CEQA	California Environmental Quality Act
CESA	California Endangered Species Act
CGS	California Geological Survey
CH_4	Methane
CHRIP	Chemical Risk Information Platform
CNDDB	California Natural Diversity Database

CO	Carbon Monoxide
CO2	Carbon Dioxode
CO2eq.	CO2-equivalent
COD	Chemical Oxygen Demand
CTI	California Toxics Inventory
CTMs	Chemical Transport Models
CVRWQCB	Central Valley Regional Water Quality Control Board
CWD	Cawelo Water District
dB	Decibels
dBA	A-Weighted Decibels
DBNPA	2,2-dibromo-3-nitrilopropionamide
DFW	Department of Fish and Wildlife
DOC	(California) Department of Conservation
DOGGR	Division of Oil, Gas, and Geothermal Resources
DOORS	Diesel Off-road On-line Reporting System
DWR	Department of Water Resources
EC	Electrical Conductivity
EC_{50}	Median Effective Concentration
ECHA	European Chemicals Agency
EDC	Endocrine Disrupting Compounds
EDGAR	Emissions Database for Global Atmospheric Research
EDTA	Ethylenediaminetetraacetic Acid
EGS	Enhanced Geothermal System
EHM	Estimated Hazard Metric
EOR	Enhanced Oil Recovery
EPI	Estimation Program Interface
ERCB	Energy Resources Conservation Board
ERG/SAGE	Eastern Research Group, Inc. and Sage Environmental Consulting LP
ESA	Endangered Species Act (Federal)
EU	European Union
EWQL	Environmental Water Quality Laboratory
GAMA	Groundwater Ambient Monitoring and Assessment
GAO	U.S. Government Accountability Office
GHG	Greenhouse Gas
GHGRP	Greenhouse Gases Reporting Program
GHS	Globally Harmonized System
GIS	Geographic Information System
GWP	Global-Warming Potential
H_2S	Hydrogen Sulfide
HAP	Hazardous Air Pollutants

HBACV	Health-Based Air Comparison Values
HC	Hydrocarbon
HCl	Hydrochloric Acid
HCP	Habitat Conservation Plan
HCF	Hydroflourocarbons
HF	Hydrofluoric Acid
HI	Hazard Indices
HSDB	Hazardous Substance Data Bank
IARC	International Agency for Research on Cancer
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
IRIS	Integrated Risk Information System
IUCLID	International Uniform Chemical Information Database
LBNL	Lawrence Berkeley National Laboratory
LC_{50}	Median Lethal Concentration
LCA	Life Cycle Assessment
LCFS	Low Carbon Fuel Standard
LD_{50}	Median Lethal Dose
LPG	Liquefied Petroleum Gas
MADL	Maximum Allowable Dose Levels
MBNPA	2-bromo-3-nitrilopropionamide
MCL	Maximum Contaminant Level
MF	Microfiltration
MIT	Methylisothiazolinone
MITI	Ministry of International Trade and Industry
MRL	Minimal Risk Levels
MRR	Mandatory Reporting Regulation
MSDS	Material Safety Data Sheets
MTBE	Methyl Tertiary Butyl Ether
N_2O	Nitrous Oxide
NEPA	National Environmental Policy Act
NETL	National Energy Technology Laboratory
NF	Nanofiltration
NGWA	National Groundwater Association
NIOSH	National Institute for Occupational Safety and Health
NIOSH REL	NIOSH Reference Exposure Level
NMVOC	Non-Methane Volatile Organic Compounds
NORM	Naturally Occurring Radioactive Materials
NO_x	Nitrogen Oxides
NP	Not Practical

NPDES	National Pollution Discharge Elimination System
NRC	National Research Council
NSRL	No Significant Risk Levels
NYSDEC	New York State Department of Environmental Conservation
OC	Organic Carbon
OCS	(Federal) Outer Continental Shelf
OECD	Organization for Economic Cooperation and Development
OEHHA	Office of Environmental Health Hazard Assessment
OES	Office of Emergency Services
OPGEE	Oil Production Greenhouse Gas Emissions Estimator
OSHA	Occupational Safety and Health Administration
OSPAR	Oslo-Paris Convention, also known as the Convention for the Protection of the Marine Environment of the North-East Atlantic
PA	Planning Area
PA DEP	Pennsylvania Department Of Environmental Protection
PAH	Polycyclic Aromatic Hydrocarbons
PBZ	Personal Breathing Zone
PEL	Permissible Exposure Limits
PFC	Perflourocarbons
PHG	Public Health Goals
PM	Particulate Matter
POTW	Publicly Owned Treatment Works
PPE	Personal Protective Equipment
ppm	Parts Per Million
PSE	PSE Healthy Energy
PSHA	Probabilistic Seismic Hazard Assessment
PSR	Physicians for Social Responsibility
QAC	Quaternary Ammonium Compounds
QCEW	Quarterly Census Of Employment And Wages
REL	Reference Exposure Levels
RfC	Reference Concentrations
RfDs	Reference Doses
RMSE	Root Mean Square Error
RO	Reverse Osmosis
ROG	Reactive Organic Gases
SAF	San Andreas Fault
SB 4	Senate Bill 4
SB 1281	Senate Bill 1281
SC	South Coast
SCAQMD	South Coast Air Quality Management District

SCEDC	Southern California Earthquake Data Center
SF_6	Sulfur hexaflouride
SFs	Slope Factors
SIC	Standard Industrial Classification
SJV	San Joaquin Valley
SNMOC	Speciated Non-Methane Organic Compounds
SoCAB	South Coast Air Basin
SWP	State Water Project
SWRCB	State Water Resources Control Board
TAC	Toxic Air Contaminant
TDS	Total Dissolved Solids
TENORM	Technologically Enhanced Naturally Occurring Radioactive Material
TLV	Threshold Limit Value
TOC	Total Organic Carbon
TOC	Toxic Organic Compound
TOG	Total Organic Gases
TOXNET	National Library of Medicine, Toxicology Data Network
TPH	Total Petroleum Hydrocarbons
TRPH	Total Recoverable Petroleum Hydrocarbons
TWA	Time Weighted Average
UCL	Upper Confidence Level
UCSB	University of California, Santa Barbara
UF	Uncertainty Factor
UF	Ultrafiltration
UGRB	Upper Green River Basin
UIC	Underground Injection Control
UR	Unit Risk Values
URE	Unit Risk Estimates
U.S. DOE	U.S. Department of Energy
U.S. EIA	U.S. Energy Information Administration
U.S. EPA	U.S. Environmental Protection Agency
USDW	Underground Source of Drinking Water
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
USQFF	United States Quaternary Fault and Fold
V/P	Various/Partial
VOC	Volatile Organic Carbon
WS	Well Stimulation
WST	Well Stimulation Treatment
WTR	Well to the Refinery Entrance Gate

Chapter One

Introduction

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1.1. Background

In 2013, the California Legislature passed Senate Bill 4 (SB 4), setting the framework for regulation of well stimulation technologies in California, including hydraulic fracturing. SB 4 also requires the California Natural Resources Agency to conduct an independent scientific study of well stimulation technologies in California. SB 4 stipulates that the independent study assess current and potential future well stimulation practices, including the likelihood that these technologies could enable extensive new petroleum production in the state; evaluate the impacts of well stimulation technologies and the gaps in data that preclude this understanding; identify potential risks associated with current practices; and identify alternative practices that might limit these risks. (See Box 1.1-1 for a short history of oil and gas production in California.) This scientific assessment addresses well stimulation used in oil and gas production both on land and offshore in California.

This study is issued in three volumes. Volume I, issued in January 2015, describes how well stimulation technologies work, how and where operators deploy these technologies for oil and gas production in California, and where they might enable production in the future. Volume II, the present volume, discusses how well stimulation could affect water, atmosphere, seismic activity, wildlife and vegetation, and human health. Volume II reviews available data, and identifies knowledge gaps and alternative practices that could avoid or mitigate these possible impacts. Volume III, also issued in July 2015, presents case studies that assess environmental issues and qualitative risks for specific geographic regions. A final Summary Report summarizes key findings, conclusions and recommendations of all three volumes.

Well stimulation enhances oil and gas production by making the reservoir rocks more permeable, thus allowing more oil or gas to flow to the well. The reports discuss three types of well stimulation as defined in SB 4 (Table 1.1-1 and Volume I, Chapter 2). The first type is “hydraulic fracturing.” To create a hydraulic fracture, an operator increases the pressure of an injected fluid in an isolated section of a well until the surrounding rock breaks, or “fractures.” Sand injected into these fractures props them open after the pressure is released. The second type is “acid fracturing,” in which a high-pressure acidic fluid fractures the rock and etches the walls of the fractures, so they remain permeable after the pressure is released. The third type, "matrix acidizing," does not fracture the rock; instead, acid pumped into the well at relatively low pressure dissolves some of the rock and makes it more permeable.

Table 1.1-1. Well stimulation technologies included in Senate Bill (SB 4).

<table>
<tr><td colspan="3">Hydraulic Fracturing Stimulation
Common feature:
All treatments create sufficient pressure in the well to induce fractures in the reservoir.</td></tr>
<tr><td colspan="2">Proppant Fracturing:
Uses proppant to retain fracture permeability</td><td>Acid Fracturing:
Uses acid instead of proppant</td></tr>
<tr><td>Traditional Fracturing:
Creates long, narrower hydraulic fractures deep into the formation for stimulating flow through lower-permeability reservoirs; proppant injected into fractures to retain fracture permeability</td><td>Frac-Pack:
Creates short, wider hydraulic fractures near wells within higher-permeability reservoirs; objectives are bypassing regions near-the wellbore damaged by drilling and preventing sand from the reservoir entering the well</td><td>Similar to traditional fracturing, but uses acid instead of proppant to retain fracture permeability by etching, or "roughening" the fracture walls; only used in carbonate reservoirs</td></tr>
</table>

<table>
<tr><td colspan="2">Acidizing Stimulation
Common feature: All treatments use acid to dissolve materials impeding flow</td></tr>
<tr><td colspan="2">Matrix Acidizing:
Dissolves material in the near-well region to make the reservoir rocks more permeable; typically only used for reservoirs that are already permeable enough to not require traditional or acid fracturing</td></tr>
<tr><td>Sandstone Acidizing:
Uses hydrofluoric acid in combination with other acids to dissolve minerals (silicates) that plug the pores of the reservoir; only used in reservoirs composed of sandstone or other siliceous rocks</td><td>Carbonate Acidizing:
Uses hydrochloric acid (or acetic or formic acids) to dissolve carbonate minerals, such as those comprising limestone, and bypass rock near the wellbore damaged by drilling; only used in carbonate reservoirs</td></tr>
</table>

Box 1.1-1. The History of Oil and Gas Production in California

California has some highest concentrations of oil in the world and oil and gas production remains a major California industry. For example, Long Beach oil field, in the Los Angeles Basin, once contained about ~ 5 billion m3 (3 billion barrels) of oil within an area of less than 7 km2 (2,000 acres). Four of the ten largest conventional U.S. oil fields are in California: Midway-Sunset, Kern River, and South Belridge in the San Joaquin Basin and Wilmington-Belmont in the Los Angeles Basin. According to the Division of Oil, Gas, and Geothermal Resources (DOGGR) there are 52 giant oil fields in the state, each with more than 16 million m3 (100 million barrels) of known recoverable oil, and many other fields of various sizes. California's oil production ranks third in the nation, behind Texas and North Dakota and provides about 20,000 jobs.

Oil has been exploited since prehistoric times, first by Native Americans and later by Spanish colonists and Mexican residents, who routinely collected "brea" from the numerous natural oil seeps. Commercial production started in the middle of the nineteenth century from hand-dug pits and shallow wells. Exploratory drilling began in the 1860s and 1870s and boomed in the first half of the Twentieth Century. In 1929, at the peak of oil development in the Los Angeles Basin, California accounted for more than 22% of total world oil production (American Petroleum Institute, 1993). California's oil production reached an all-time high of almost 64 million m3 (400 million barrels) in 1985 and has generally declined since then. By 1940 all but four of the giant onshore fields had been discovered. San Ardo, South Cuyama, and Round Mountain were discovered in the 1940s, and the last, Yowlumne field, was discovered in1974. Today California is the third highest producing state, with about 6% of US production but less than 1% of global production. In 1960, almost as much oil was produced in California as was consumed, but by 2012 Californians produced only 32% of the oil they used (31.5 million m3, or 198 million barrels produced in the state out of a total of about 98.7 million m3, or 621 million barrels consumed). Californian's mainly made up the shortfall of about 67.3 million m3 (423 million barrels) mainly with oil delivered by tanker from Alaska, Saudi Arabia, Ecuador, Iraq, Colombia, and other countries.

Over the years, water flooding, gas injection, thermal recovery, hydraulic fracturing, and other techniques have been used to enhance oil and gas production as California fields mature. Water flooding involves injecting water into a reservoir, causing additional oil to flow to production wells. Water flooding was first used in the Los Angeles Basin in 1956 at Wilmington-Belmont field to mitigate subsidence, with the incidental benefit of increased oil recovery. By the 1960s the method had been widely deployed in many fields around the state as an effective means of augmenting production.

California has substantial heavy oil that must be liquefied with heat to make it flow to a well. Steam injection (steam flooding and soak), the most commonly used "thermal recovery" method, involves injecting steam into wells interspersed among production wells. Nearly all production at Kern River field and much of the production from Midway Sunset and many other California fields is heavy oil produced by thermal recovery. Since 1989, when DOGGR first reported oil recovered by water flooding and steam injection, over 70% of production can be attributed to these energy-intensive techniques (DOGGR, 1990; DOGGR, 2010).

The diatomite reservoirs in the western San Joaquin Valley contain billions of barrels of oil in rocks that are not very permeable, and can only be produced with hydraulic fracturing—now accounting for about 20% of California oil and gas production (see Volume I, Chapter 3).

The first offshore oil production in the United States began in 1897 on piers in Santa Barbara County. The first Federal Outer Continental Shelf (OCS) lease sale was held in 1966 and production began from a platform in 1969. That same year a well failure on Union Oil Platform A in Dos Cuadras field, not far from the Santa Barbara Coast, spilled

15,899 m3 (100,000 barrels) in ten days and made a deep negative impression on public opinion that has constrained offshore development ever since. In 1984 a moratorium on development in the Federal OCS went into effect. Billions of barrels of recoverable oil probably remain in the federal offshore, but with no new leases, OCS production has been steadily declining since 1996.

California's oil production reached an all-time high of almost 64 million m3 (400 million barrels) in 1985 and has generally declined since then. In 1960, almost as much oil was produced in California as was consumed, but by 2012 Californians used about 67.3 million m3 (423 million barrels) more than they produced (Figure 1.1-1) with the shortfall mainly delivered by tanker from Alaska, Saudi Arabia, Ecuador, Iraq, Colombia and other countries.

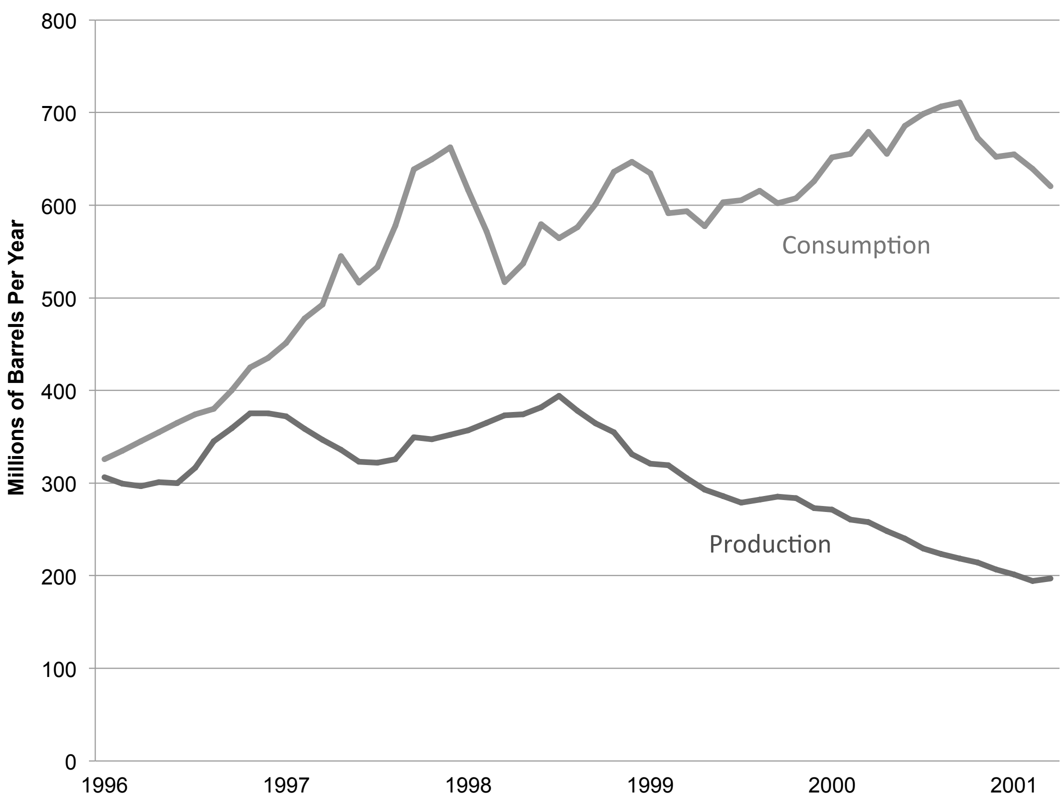


Figure 1.1-1. Total oil production (blue line) and consumption (grey line) from all sources in California from 1960 to 2012 (Data: US EIA, 2014a and b).

Natural gas is much less abundant than oil in California and most of the state's natural gas production is a co-product of oil development, referred to as "associated" gas production. Only the Sacramento Basin has significant non-associated natural gas production, but about three quarters of the gas production in the state is not from dry gas wells, but from wells that primarily produce oil, mostly in the San Joaquin Valley.

1.1.1. California Council on Science and Technology (CCST) Committee Process

The California Council on Science and Technology (CCST) organized and led the study reported on here. Members of the CCST steering committee were appointed based on technical expertise and a balance of technical viewpoints. (Volume II, Appendix B provides information about CCST's Steering Committee.) Under the guidance of the Steering Committee, Lawrence Berkeley National Laboratory (LBNL) and subcontractors (the science team) developed the findings based on the literature review and original technical data analyses. Volume II, Appendix C provides information about the LBNL science team and subcontractors who authored Volumes I, II, and III of this report. The science team reviewed relevant literature and conducted original technical data analyses.

The science team studied each of the issues required by SB 4, and the science team and the steering committee collaborated to develop a series of conclusions and recommendations that are provided in this summary report. Both science team and steering committee members proposed draft conclusions and recommendations. These were modified based on discussion within the steering committee along with continued consultation with the science team. Final responsibility for the conclusions and recommendations in this report lies with the steering committee. All steering committee members have agreed with these conclusions and recommendations. Any steering committee member could have written a dissenting opinion, but no one requested to do so.

SB 4 also required the participation of the California Environmental Protection Agency's Office of Environmental Health Hazard Assessment (OEHHA) in this study. OEHHA provided toxicity and other risk assessment information on many of the chemicals used in hydraulic fracturing, offered informal technical advice during the course of the study, and provided comments on drafts of Volumes II and III. OEHHA also organized a February 3, 2015 public workshop in Bakersfield in which representatives of CCST, LBNL, and subcontractors heard comments from attendees on the topics covered in the report.

This report has undergone extensive peer review. (Peer reviewers are listed in Volume II, Appendix F: "California Council on Science and Technology Study Process"). Seventeen reviewers were chosen for their relevant technical expertise. More than 1,500 anonymous review comments were provided to the authors. The authors revised the report in response to peer review comments. In cases where the authors disagreed with the reviewer, the response to review included their reasons for disagreement. Report monitors then reviewed the response to review and when satisfied, approved the report.

1.1.2. Data and Literature Used in the Report

This assessment reviews and analyzes both existing data and scientific literature, with preference given to findings in the peer-reviewed scientific literature. The study included both voluntary and mandatory reporting of stimulation data, as well as non-peer reviewed reports and documents if they were topically relevant and determined to be scientifically credible by the authors and reviewers of this volume. Finally, the California Council on Science and Technology solicited and reviewed nominations of literature from the public, employing specific criteria for material as described in Volume I, Appendix E, "Review of Information Sources." The science team did not collect any new data, but did do original analysis of available data.

Volumes I, II and III of this report address issues that have very different amounts of available information and cover a wide range of topics and associated disciplines, which have well established but differing protocols for inquiry. In Volume I, available data and methods of statistics, engineering and geology allowed the authors to present the factual basis of well stimulation in California. With a few exceptions, the existing data was sufficient to identify the technologies used, where and how often they are used, and where they are likely to be used in the future (see Volume I Chapter 3). This volume, Volume II, faces the challenge of presenting the impacts of well stimulation. Since many impacts have never been thoroughly investigated, the authors drew on literature describing conditions and outcomes in other places, circumstantial evidence and expert judgment to catalog a complete list of potential impacts. Volume II also identifies a set of concerning situations – "risk factors" (summarized in Appendix D of the Summary Report and Table 6.2-1 of this volume)-- that warrant a closer look and perhaps regulatory attention. We believe this flexible and appropriate use of different (but well established) methods of inquiry under highly variable conditions of data availability and potential impacts serves useful to California.

The SB 4 completion reports provide reliable data to assess certain potential environmental and health impacts such as the use of fresh water for hydraulic fracturing. For most potential impacts, however, only incomplete information and data exist. Few scientific studies of the health and environmental impacts of well stimulation have been conducted to date, and the ones that have been done focus on other parts of the country, where practices differ significantly from present-day practices in California. Generally, environmental baseline data has not been collected in the vicinity of stimulation sites before stimulation. The lack of baseline data makes it difficult to know if the process of stimulation has changed groundwater chemistry or habitat, or how likely any potential impacts might be. No records of contamination of protected water by hydraulic fracturing fluids in California exist, but few targeted studies have been conducted to look for such contamination. Data describing the quality of groundwater near hydraulic fracturing sites is not universally available. The requirement for groundwater monitoring in SB 4 addresses this issue by requiring groundwater monitoring when protected water is present. Applications for hydraulic fracturing operations in locations that have no nearby protected groundwater have been exempted from groundwater monitoring. Consequently information is now being gathered about the quality of water near proposed hydraulic fracturing sites, but the SB 4 requirements have only been in place since 2013.

A complete analysis of the risks posed by well stimulation (primarily hydraulic fracturing) to water contamination, air pollution, earthquakes, wildlife, plants, and human health requires much more data than that available. However, the study authors were able to draw on their technical knowledge, data from other places, and consideration of the specific conditions in California to identify conditions in California that deserve more attention and make recommendations for additional data collection, increased regulation, or other mitigating measures.

1.2. Assessing Impacts of Hydraulic Fracturing in California

This scientific assessment of hydraulic fracturing and acid stimulation impacts covers the application of hydraulic fracturing and acid stimulation technology and resulting oil and gas production activities. The report considers impacts and potential impacts resulting from the development of a well pad and support infrastructure required to drill the well, hydraulic fracturing or acid stimulation and completion, production of oil and/or natural gas, and disposal or reuse of produced water. Figure 1.2-1 shows the parts of the oil and gas system included in this assessment and examples of impacts for each.

This report excludes other stages in the development, production, refining, and use life cycle of oil and gas, including impacts of manufacturing of materials or equipment used in stimulation, impacts of transport of produced oil and gas to refineries or providers, impacts of refining, or impacts of combustion of hydrocarbons as fuel.

Existing California regulations, including the state's new well stimulation regulations effective July 1, cover many of the areas of potential concern or risk raised in this study, 2015. This study does not address the effectiveness of the current regulatory framework in mitigating any potential risks associated with well stimulation technologies, but recommends that the state conduct such assessments in the future.

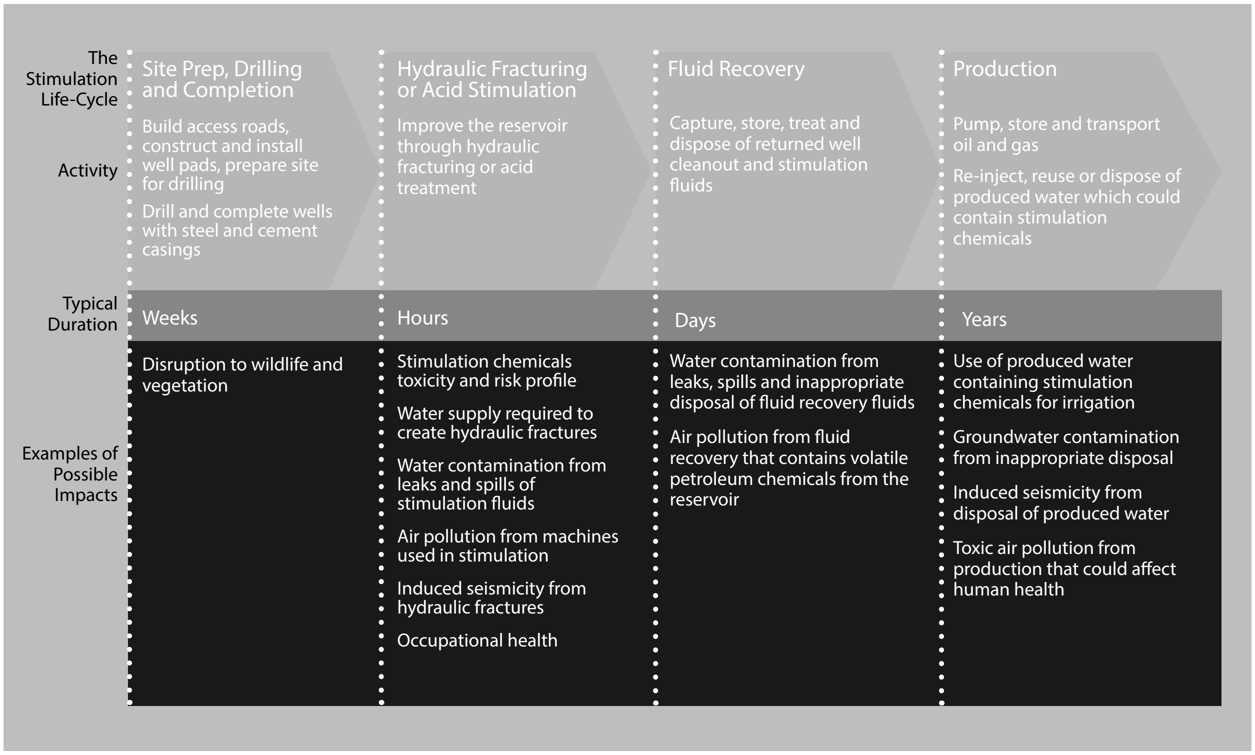


Figure 1.2-1. The sequential parts of the well stimulation system considered in this report.

1.2.1. Direct and Indirect Impacts of Hydraulic Fracturing and Acid Stimulation.

Hydraulic fracturing or acid stimulation can cause direct impacts. Potential direct impacts might include a hydraulic fracture extending into protected groundwater, accidental spills of fluids containing hydraulic fracturing chemicals or acid, or inappropriate disposal or reuse of produced water containing hydraulic fracturing chemicals. These direct impacts do not occur in oil and gas production unless hydraulic fracturing or acid stimulation has occurred. This study covers potential direct impacts of hydraulic fracturing or acid stimulation.

Hydraulic fracturing or acid stimulation can also incur indirect impacts, i.e., those not directly attributable to the activity itself. Some reservoirs require hydraulic fracturing for economic production. All activities associated with oil and gas production enabled by hydraulic fracturing or acid stimulation can bring about indirect impacts. Indirect impacts of hydraulic-fracturing-enabled oil and gas development usually occur in all oil and gas development, whether or not the wells are stimulated.

In some cases, we cannot separate direct and indirect impacts. For example, the inventory of emissions of hazardous air pollutants is for all oil and gas production and does not differentiate between hydraulically fractured and unfractured wells, so the data do not support differentiating direct and indirect impacts. However, as illustrated in the following examples, differentiating direct and indirect impacts can be important for framing investigations and policy.

An indirect impact common to all production, not just production enabled by hydraulic fracturing, means the impacts incurred by just the hydraulically fractured wells represent a small subset of the problem. For example, disposal of produced water through underground injection may carry the risk of inducing an earthquake. If this produced water comes from a hydraulically fractured reservoir, this potential impact would be an indirect impact. In California, about 20% of all produced waters come from stimulated reservoirs. Understanding induced seismicity requires looking at all the wastewater injections, not just those generated by hydraulically fractured wells. In this case, the indirect impact attributed to hydraulically fractured wells represents a small part of a larger problem.

As another example, studies show elevated health risks near hydraulically fractured reservoirs attributable to benzene (Volume II, Chapter 6). But benzene use has been phased out in hydraulic fracturing fluids. These health risks probably occur due to processes associated with oil production, because oil contains benzene naturally. In this case, the health impacts do not occur because of hydraulic fracturing itself; they are indirect impacts that occur because of production. So the same health impacts could occur near any production, whether the wells have been fractured or not. Research that focuses only on benzene impacts near hydraulically fractured wells will likely result in a very poor understanding of both the extent of this problem and the possible mitigation measures. Concern about hydraulic fracturing might lead to studying health effects near fractured wells, but concern about the health effects from benzene should lead to study of all types of oil and gas production, not just hydraulically fractured wells.

As a final example, the activities associated with hydraulic fracturing or acid stimulation can add some new direct occupational hazards to a business that already has substantial occupational hazards. The drilling, completion, and production phases common to all oil and gas production incur significant risk of exposure to many toxic substances and accidents. In general, oil and gas production has significant occupational health issues, but these impacts are not directly attributable to well stimulation activity. In hydraulic fracturing, silica sand used for the proppant in hydraulic fracturing presents an additional occupational health hazard for serious lung disease (silicosis). Potential exposure to silica is a direct impact of hydraulic fracturing and a relatively small part of the total hazard profile for oil and gas development.

While this project was not tasked with a full assessment of the impacts of all oil and gas development in California, we have described indirect impacts in the context of all oil and gas production where the issue and associated data either allows or requires this. This report does include some recommendations for assessment of certain impacts for all oil and gas development in the future.

Table 1.2-1 describes the potential direct impacts of hydraulic fracturing and acid stimulation, plus potential indirect impacts of hydraulic-fracturing-enabled oil and gas development covered in this report.[1] The table includes issues of concern named in the SB 4 legislation or issues that have been raised by the public in the various forums around California and the U.S. regarding well stimulation or were identified by expert judgment. A long list of features, events, and processes related to well stimulation and production could possibly lead to harmful impacts, but these are not all likely or equally likely. A long list of plausible hazards have been described in Volume II, but the reader is cautioned to treat these as a "checklist" of possible impacts, not at all a list of impacts that are generally occurring. Existing regulations prevent or mitigate many of these risks; however, an evaluation of the effectiveness of this regulatory framework was beyond the scope of this study.

Out of the possible plausible hazards, some emerge as especially relevant potential risk factors worthy of further attention through additional data collection or increased scrutiny. Chapter 6 presents a table of these risk issues, which are also the basis of the conclusions and recommendations in this chapter.

1. We do not include indirect impacts of acid stimulation because based on existing data, we did not find reservoirs that required acid stimulation for production.

Table 1.2-1. Examples of direct and indirect impacts considered in this study.

Issue	**Possible Direct Impact**	**Possible Indirect Impact of Hydraulic-Fracturing-Enabled Oil and Gas Development**
Stimulation Chemicals	Chemicals used in stimulation create the potential for introduction of hazardous materials into the environment.	N/A
Water Use	Stimulation uses California fresh water supply.	Freshwater is sometimes used to produce oil in a previously stimulated reservoir, e.g., enhanced oil recovery via injection of water or steam.
Water Supply	Stimulation chemicals could enter produced water that is otherwise of sufficient quality for beneficial uses, such as irrigation, making treatment more complicated.	Additional production enabled by hydraulic fracturing can lead to additional produced water, which, with appropriate treatment, may be of sufficient quality for beneficial uses.
Water Contamination	Intentional or accidental releases of stimulation chemicals and their reaction products could lead to contamination of fresh water supply. Risk of hydraulic fractures acting as conduit for accidental releases of fluids; and risk of high-pressure injection affecting integrity of existing wells.	N/A
Air pollution	Equipment used in stimulation emits pollutants and greenhouse gases (GHGs). Retention ponds and tanks used to store stimulation fluids could contain off-gassing volatile organic compounds (VOC).	Oil and gas development activities cause emissions including VOC emissions from produced water.
Induced Seismicity	Hydraulic fracturing could cause earthquakes.	Disposal of wastewater from hydraulic fracture-enabled production in disposal wells classified by the EPA's Underground Injection Control (UIC) program as "Class II"[1] could cause earthquakes.
Human Health	Releases of stimulation chemicals that pollute water and air, as well as noise and light pollution from the stimulation operation could affect public health.	Proximity to any oil production, including stimulation-enabled production, could result in hazardous emissions to air and water, and noise and light pollution that could affect public health.
Wildlife and Vegetation	Introduction of invasive species; contamination of habitat or food web by stimulation chemicals; and water use for stimulation fluids could impact wildlife and vegetation.	Habitat loss and fragmentation, introduction of invasive species, and water use for enabled enhanced oil recovery could impact wildlife and vegetation.

1. Class II wells are underground injection wells that inject fluids associated with oil and natural gas production. There are three types of Class II wells: enhanced recovery, wastewater disposal, and hydrocarbon storage. For more information, see http://water.epa.gov/type/groundwater/uic/class2/index.cfm.

1.2.2. Impacts Covered in this Volume

The chapters of this volume assess, to the extent possible, the potential impacts of well stimulation on water, air, seismicity, habitat and human health.

Chapter 2 analyzes the hazards and potential impacts of well stimulation on California's water resources including water use in well stimulation, the volumes, chemical compositions, and potential hazards of stimulation fluids, and the characteristics of wastewater including production, management, and the potential release mechanisms and transport pathways by which well stimulation chemicals enter the water environment. The chapter addresses the following questions and for each evaluates the available data, identifies data gaps and ways to mitigate or avoid potential impacts:

- What are the volumes of fresh water used for well stimulation in California, and what are the sources of these supplies (e.g., domestic water supplies, private groundwater wells, irrigation sources)? How does water use for well stimulation compare with other uses in California and in the regions where well stimulation is occurring?

- What are the volumes and chemical compositions—including types of chemicals and quantities—of stimulation fluids? What are the physical, chemical, and toxicological properties of the stimulation chemicals used? To what extent does this chemical use create hazards for and potential impacts on water resources in California?

- What volumes of recovered fluids and produced water are generated from stimulated wells and what are the chemical compositions of those waters? Are volumes of produced water generated from stimulated wells and non-stimulated wells different? Does the chemical composition of produced water from stimulated wells differ from that of non-stimulated wells? What techniques are used to recover fluids and manage produced water (e.g., deep well injection, unlined sumps)? Could existing treatment technologies remove well stimulation chemicals that are being used in California?

- What are the release mechanisms and transport pathways by which well stimulation chemicals could enter surface water and groundwater aquifers? Could the introduction of stimulation chemicals into the environment affect ecosystems and human health (through contamination of aquifers, spills, inappropriate uses of wastewater, etc.)?

Chapter 3 assesses the potential of well stimulation to emit greenhouse gases (GHGs), volatile organic compounds (VOCs), oxides of nitrogen (NO_x), toxic air contaminants (TACs), and particulate matter (PM). Because oil and gas development in general can also have these impacts, the purpose of this chapter is to evaluate what is known about the contribution of well stimulation to general impacts from oil and gas development.

Well stimulation could impact air quality via emission of a large variety of chemical species. These species can have local, regional, or global impacts, mediated by the regional atmospheric transport mechanisms and the natural removal mechanisms relevant for that species. For clarity, this report groups species into four categories of interest, each with unique potential impacts.

1. Greenhouse gases (GHGs);

2. Reactive organic gases (ROGs), and oxides of nitrogen (NO_x) that cause photochemical smog generation;

3. Toxic air contaminants (TACs, a California-specific designation similar to federal designation of hazardous air pollutants (HAPs); and

4. Particulate matter (PM), including dust.

The chapter describes methods of classifying well-stimulation-related air impacts, and the major sources and types of emissions from oil and gas activities. The chapter also describes the treatment of well-stimulation-related emissions in current California emissions inventories. Then, the chapter evaluates the California regions likely to be affected by the use of well-stimulation technology, current best practices for managing air quality impacts of well stimulation, and gaps in data and scientific understanding surrounding well-stimulation-related air impacts.

Chapter 4 assesses the potential for induced seismicity in California caused by injection of fluids into the subsurface. The vast majority of earthquakes induced by fluid injection are too small to be felt at the ground surface. However, induced seismicity can produce felt or, in rare cases, damaging ground motions. Large volumes of water injected over long time periods (i.e. months to years) into zones in or near potentially active earthquake sources can induce earthquakes. This chapter reviews the current state of knowledge about induced seismicity, and the data and research required to determine the potential for induced seismicity in California, including along the San Andreas Fault. The chapter also discusses how existing protocols could be improved to lower the risk from induced seismicity in California.

Chapter 5 evaluates the potential impact of well stimulation on wildlife and vegetation, and how these impacts depend on the density of oil and gas wells and other human land uses in the area. The chapter describes how the impacts of oil and gas production to native wildlife and vegetation depend on the prevailing land use. In some regions, well stimulation takes place in areas where wild habitat has already been displaced by near-continuous well pads or agricultural and urban development. However, in oil fields with little other development and a relatively low density of oil wells, oil and gas development could more directly impact valuable native habitat. Because habitat loss and fragmentation is likely to have the greatest impact on wildlife and vegetation, the chapter explores this topic in greater depth by quantifying habitat loss and fragmentation attributable to well-stimulation-enabled hydrocarbon production. Other potential impacts, such as the introduction of invasive species, releases of harmful fluids to the environment, diversion of water from waterways, noise and light pollution, vehicle collisions, ingestion of litter by wildlife, and the possible release of well stimulation chemicals into the environment are described. Then the chapter reviews regulation of the oil and gas industry with respect to impacts on wildlife and vegetation. The chapter describes measures to mitigate oil field impacts on terrestrial species and their habitats, and major data gaps and ways to remedy the gaps.

Chapter 6 addresses health hazards associated with community and occupational environmental exposures *directly* attributable to well stimulation and *indirect* exposures due to oil and gas development that were facilitated by stimulation in California. The chapter evaluates hazards directly attributable to well stimulation stemming from the chemicals used in stimulation that might contact humans through contaminated water (described in Chapter 2) and air pollution hazards associated with oil and gas development described in Chapter 3 for human health.

1.3. Conclusions and Recommendations

The following conclusions and recommendations are numbered to correspond to the full set of conclusions and recommendations as given in the Summary Report, but only those conclusions and recommendations that derive from this volume are given below. This is the reason that the conclusions and recommendations are not numbered sequentially starting with number 1. For the sake of consistency, some conclusions include information from other volumes as noted.

1.3.1. Direct and Indirect Impacts of Hydraulic Fracturing and Acid Stimulation

Conclusion 3.1. Direct impacts of hydraulic fracturing appear small but have not been investigated.

Available evidence indicates that impacts caused directly by hydraulic fracturing or acid stimulation or by activities directly supporting these operations appear smaller than the indirect impacts associated with hydraulic-fracturing-enabled oil and gas development, or limited data precludes adequate assessment of these impacts. Good management and mitigation measures can address the vast majority of potential direct impacts of well stimulation.

Hydraulic fracturing in California lasts a relatively short amount of time near the beginning of production—less than a day—and requires relatively small fluid volumes. In contrast, the subsequent oil and gas production phase lasts for years and involves very large volumes of fluid, with potential for long-term perturbations of the environment. Consequently, the production phase following well stimulation can have a much larger impact than the stimulation phase.

This study identifies a number of possible pathways for direct impacts from hydraulic fracturing and acid stimulation, such as accidental spills or leaks of hydraulic fracturing or acid fluids or emissions of volatile organic compounds (VOCs) from hydraulic fracturing fluids. Many, if not all, of these potential direct impacts can be addressed with good management practices or mitigation measures. These are described in Volumes II and III.

The recommendations below provide specific measures that could eliminate, avoid, or ameliorate direct impacts. These measures include limiting the use of toxic chemicals, avoiding inappropriate disposal, managing beneficial use of produced water containing stimulation chemicals, providing extra due diligence for shallow fracturing near protected groundwater, and using “green completions” to control emissions in oil and gas wells.

In California, existing or pending regulation already addresses many of these direct impacts. The state’s new well stimulation regulations, going into effect on July 1, 2015, will likely avoid or reduce many, but not all, of the impacts described in this report. The scope of this study did not include judging the adequacy of existing regulation, but this would make sense at some later time when significant experience can be assessed.

Recommendation 3.1. Assess adequacy of regulations to control direct impacts of hydraulic fracturing and acid stimulations.

Over the next several years, relevant agencies should assess the adequacy and effectiveness of existing and pending regulations to mitigate direct impacts of hydraulic fracturing and acid stimulations, such as to: (1) reduce the use of highly toxic or harmful chemicals, or those with unknown environmental profiles in hydraulic fracturing and acid fluids; (2) devise adequate treatment and testing for any produced waters intended for beneficial reuse that may include hydraulic fracturing and acid fluids or disallow this practice; (3) prevent shallow hydraulic fractures from intersecting protected groundwater (Volume II); (4) dispose of produced waters that contain stimulation chemicals appropriately; and (5) control emissions, leaks and spills.

Conclusion 3.2. Operators have unrestricted use of many hazardous and uncharacterized chemicals in hydraulic fracturing.

The California oil and gas industry uses a large number of hazardous chemicals during hydraulic fracturing and acid treatments. The use of these chemicals underlies all significant potential direct impacts of well stimulation in California. This assessment did not find recorded negative impacts from hydraulic fracturing chemical use in California, but no agency has systematically investigated possible impacts. A few classes of chemicals used in hydraulic fracturing (e.g., biocides, quaternary ammonium compounds, etc.) present larger hazards because of their relatively high toxicity, frequent use, or use in large amounts. The environmental characteristics of many chemicals remain unknown. We lack information to determine if these chemicals would present a threat to human health or the environment if released to groundwater or other environmental media. Application of green chemistry principles, including reduction of hazardous chemical use and substitution of less hazardous chemicals, would reduce potential risk to the environment or human health.

Operators have few, if any, restrictions on the chemicals used for hydraulic fracturing and acid treatments. The state's regulations address hazards from chemical use and eliminate or minimize many, but not necessarily all risks. Some of the chemicals used present hazards in the workplace or locally, such as silica dust or hydrofluoric acid. Other chemicals present potential hazards for the environment, such as biocides and surfactants that, if released, can harm fish and other wildlife. Many of the chemicals used can harm human health. If well stimulation did not use hazardous chemicals, hydraulic fracturing would pose a much smaller risk to humans and the environment. Even so, hazardous chemicals only present a risk to humans or the environment if they are released in hazardous concentrations or amounts, persist in the environment, and actually reach and affect a human, animal or plant. Even a very toxic or otherwise harmful chemical presents no risk if no person, animal or plant receives a dose of the chemical. Characterization of the risk posed by chemical use requires information on both the hazards posed by the chemicals and information about exposure to the chemicals (in other words, risk = hazard x exposure).

We have established a list of chemicals used in California based on voluntary disclosures by industry. In California, oil and gas production operators have voluntarily reported the use of over 300 chemical additives. New state regulations under SB 4 will eventually reveal all chemical use. However, knowledge of the hazards and risks associated with all the chemicals remains incomplete for almost two-thirds of the chemicals (Table 1.3-1). The toxicity and biodegradability of more than half the chemicals used in hydraulic fracturing remains uninvestigated, unmeasured, and unknown. Basic information about how these chemicals would move through the environment does not exist. Although the probability of human and environmental exposure is estimated to be low, no direct studies of environmental or health impacts from hydraulic fracturing and acid stimulation chemicals have been completed in California. To the extent that any hydraulic fracturing and acid stimulation fluids can get into the environment, reduction or elimination of the use of the most hazardous chemicals will reduce risk.

Table 1.3-1. Availability of information for characterizing the hazard of stimulation chemicals used in hydraulic fracturing. The Chemical Abstracts Service Registry Number (CASRN) is a unique numerical identifier assigned to chemical substances. Operators do not provide CASRN numbers for proprietary chemicals.

Number of chemicals	**Proportion of all chemicals**	**Identified by unique CASRN**	**Impact or toxicity**	**Quantity of use or emissions**
172	55%	Available	Available	Available
17	5%	Available	Available	Unavailable
6	2%	Available	Unavailable	Available
121	38%	Unavailable	Unavailable	Available

For this study, we sorted the extensive list of chemicals reported in California to identify those of most concern or interest and created tables identifying selected chemicals for each category contributing to hazard (see Summary Report, Appendix H, and Volume II, Chapters 2 and 6). Chemicals used most frequently or in high concentrations rise to a higher level of concern, as do chemicals known to be acutely toxic to aquatic life or mammals. The assessment included chemicals used in hydraulic fracturing that can be found on the Toxic Air Contaminant Identification List, the Proposition 65 list of chemicals known to the State of California to cause cancer and reproductive harm, and the OEHHA list of chemicals with published reference exposure limits. Additional hazards considered include, flammability, corrosivity, and reactivity. These various criteria allow identification of priority chemicals to consider when reducing potential hazards from chemical use during well stimulation.

Strong acids, strong bases, silica, biocides, quaternary ammonium compounds, nonionic surfactants, and a variety of solvents are used frequently and in high concentrations in hydraulic fracturing and acid stimulation. Strong acids, strong bases, silica, and many solvents present potential exposure hazards to humans, particularly during handling, and of are of particular concern to workers and nearby residents. Use of appropriate procedures minimizes the risk of exposure and few incidences of the release of these materials during oil and gas development have been reported in California.

Biocides, quaternary ammonium compounds, nonionic surfactants, and some solvents present a significant hazard to aquatic species and other wildlife, particularly when released into surface water. The study found no releases of hazardous hydraulic fracturing chemicals to surface waters in California and no direct impacts to fish or wildlife. However, there is concern that well stimulation chemicals might have been released and potentially contaminated groundwater through a variety of mechanisms (see Conclusions 4.1, 4.3, 4.4, 5.1, 5.2 below). Many of the chemicals used in well stimulation, such as surfactants, are more harmful to the environment than to human health, but all of these chemicals are undesirable in drinking water. Determining whether chemicals that have been released pose an actual risk to human health or the environment requires further study, including a better understanding of the amounts of chemicals released and persistence of those chemicals in the environment.

Green Chemistry principles attempt to maintain an equivalent function while using less toxic chemicals and smaller amounts of toxic chemicals. It may be possible to forego or reduce the use of the most hazardous chemicals without losing much in the way of functionality. Chemical substitutions can present complications and can also introduce a new set of hazards and require a careful adaptive approach. For example, the use of guar in hydraulic fracturing fluids introduces food to bacteria in the reservoir, and this increases the need for biocides to prevent the buildup of toxic gases generated by bacterial growth. Operators moving to a less toxic but less effective biocide might also need to move away from guar to a less-digestible substitute. Then this choice could introduce new hazards instead of old hazards. For these reasons, the American Chemical Society currently sponsors a Green Chemistry Roundtable on the topic of hydraulic fracturing.

The state could also limit the chemicals used in hydraulic fracturing by disallowing certain chemicals or limiting chemicals to those on an approved list where approval depends on the chemical having an acceptable environmental profile. The latter approach reverses the usual practice, whereby an industry is permitted to use a chemical until a regulatory body proves that the chemical is harmful. Oil and gas production in the environmentally sensitive North Sea uses this pre-approval approach and might provide a model for limiting chemical risk in California. The EPA Designed for the Environment (DFE) list of chemicals may also be useful. Of course, any of these approaches requires that the operators report the unique identifier (CASRN number) of all chemicals.

Recommendation 3.2. Limit the use of hazardous and poorly understood chemicals.

Operators should report the unique CASRN identification for all chemicals used in hydraulic fracturing and acid stimulation, and the use of chemicals with unknown environmental profiles should be disallowed. The overall number of different chemicals should be reduced, and the use of more hazardous chemicals and chemicals with poor environmental profiles should be reduced, avoided, or disallowed. The chemicals used in hydraulic fracturing could be limited to those on an approved list that would consist only of those chemicals with known and acceptable environmental hazard profiles. Operators should apply Green Chemistry principles to the formulation of hydraulic fracturing fluids, particularly for biocides, surfactants, and quaternary ammonium compounds, which have widely differing potential for environmental harm. Relevant state agencies, including DOGGR, should as soon as practical engage in discussion of technical issues involved in restricting chemical use with a group representing environmental and health scientists and industry practitioners, either through existing roundtable discussions or independently (Volume II, Chapters 2 and 6).

Conclusion 3.3. The majority of impacts associated with hydraulic fracturing are caused by the indirect impacts of oil and gas production enabled by the hydraulic fracturing.

Impacts caused by additional oil and gas development enabled by well stimulation (i.e. indirect impacts) account for the majority of environmental impacts associated with hydraulic fracturing. A corollary of this conclusion is that all oil and gas development causes similar impacts whether the oil is produced with well stimulation or not. If indirect impacts caused by additional oil and gas development enabled by hydraulic fracturing cause concern, these concerns in most cases extend to any oil and gas development. As hydraulic fracturing enables only 20% of production in California, only about 20% of any given indirect impact is likely attributable to hydraulically fractured reservoirs.

Without hydraulic fracturing, oil and gas production from certain reservoirs would not be possible. If this oil and gas development did not occur, then the impacts of this development would not occur. Well stimulation is a relatively brief operation done after a well is installed, but oil and gas development goes on for years, involving construction of infrastructure and disruption of the landscape. Operators build roads, ponds, and well pads, and install pumps, field separators, tanks, and treatment systems in reservoirs that are stimulated and in those that are not. Surface spills and subsurface leakage may lead to impacts on groundwater quality as an impact of production. The life of a production well involves production of many millions of gallons of water that must be treated or disposed of properly. Production with or without stimulation can cause emission of pollutants over many years, often in proximity to places where people live, work, and go to school. Whereas the short-term injection of fluids for the purpose of hydraulic fracturing is unlikely to cause a felt or damaging earthquake (a direct impact), the subsurface disposal of millions of gallons of water produced along with oil over the life of a well can present a seismic hazard. The inappropriate disposal of produced water can contaminate protected groundwater, whether this water contains stimulation chemicals or not. All oil and gas development potentially incurs impacts similar to the indirect impacts of hydraulic fracturing.

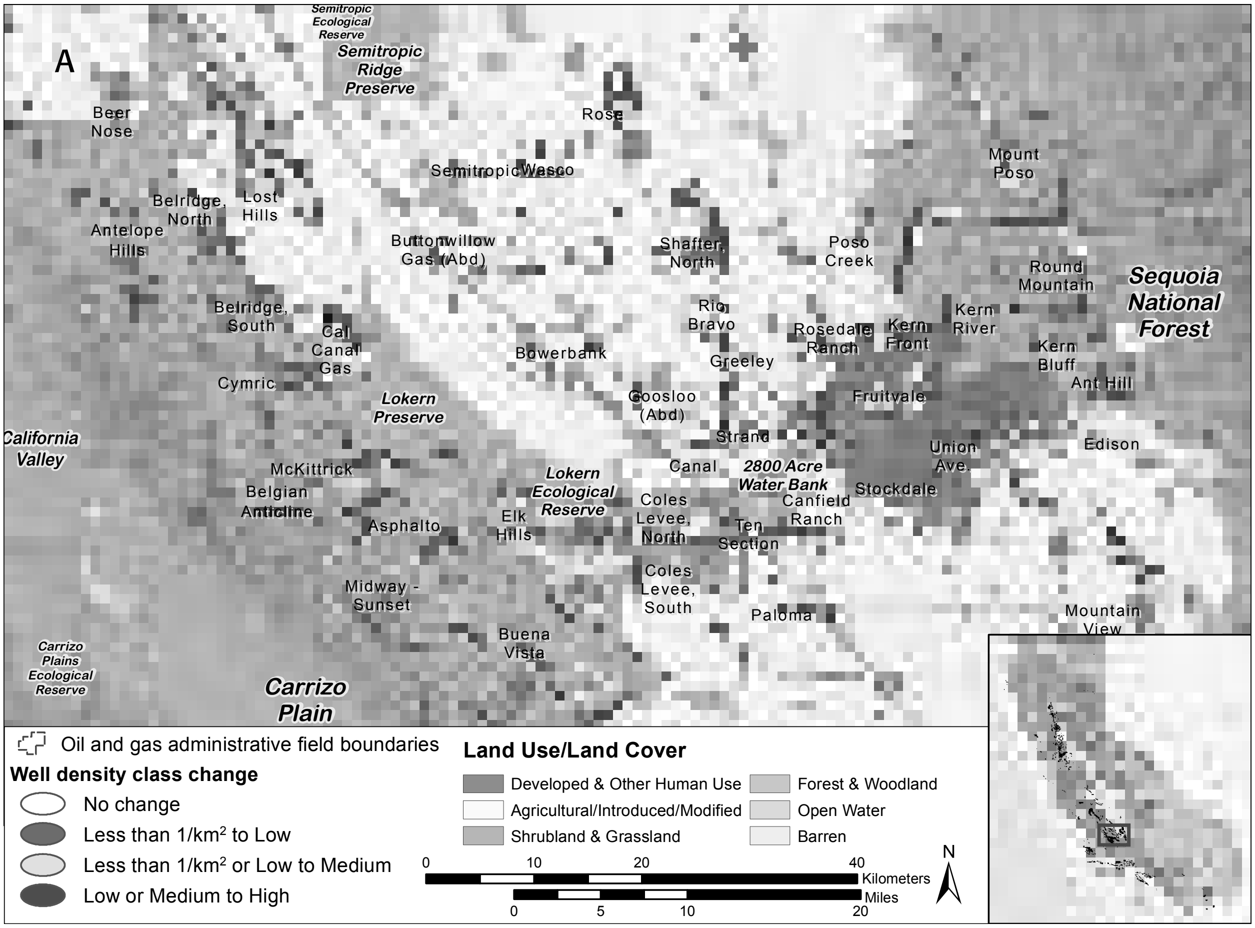
> **Recommendation 3.3. Evaluate impacts of production for all oil and gas development, rather than just the portion of production enabled by well stimulation.**
>
> *Concern about hydraulic fracturing might cause focus on impacts associated with production from fractured wells, but concern about these indirect impacts should lead to study of all types of oil and gas production, not just production enabled by hydraulic fracturing. Agencies with jurisdiction should evaluate impacts of concern for all oil and gas development, rather than just the portion of development enabled by well stimulation. As appropriate, many of the rules and regulations aimed at mitigating indirect impacts of hydraulic fracturing and acid stimulation should also be applied to all oil and gas wells (Volume II, Chapter 6).*

Conclusion 3.4. Oil and gas development causes habitat loss and fragmentation.

Any oil and gas development, including that enabled by hydraulic fracturing, can cause habitat loss and fragmentation. The location of hydraulic fracturing-enabled development coincides with ecologically sensitive areas in Kern and Ventura Counties.

The impact to habitat for native wildlife and vegetation caused by increases in well density depends on the background land use. Some California oil and gas fields are already so densely filled with well pads that other human land uses and native species habitat cannot coexist. Other oil and gas fields have relatively sparse infrastructure interspersed with cities, farms, and natural habitat. The impact caused by increases in well density depends on the background land use. Oil wells installed into agricultural land (such as Rose and Shafter oil fields), or urban areas such as Los Angeles, create only minor impacts to native species. Increases in well density and habitat disturbance from well pads, roads, and facilities cause substantial loss and fragmentation of valuable habitat in those oil and gas fields inhabited by native wildlife and vegetation.

Elk Hills, Mt. Poso, Buena Vista, and Lost Hills fields in Kern County and the Sespe, Ojai, and Ventura fields in Ventura County host substantial amounts of hydraulic fracturing-enabled development as well as rare habitat types and associated endangered species. Portions of oil fields in Kern County are essential to support resident populations of rare species and serve as corridors for maintaining connectivity between remaining areas of natural habitat (including protected areas), and these are vulnerable to expanded production (Figure 1.3-1).


A
Semitropic Ecological Reserve
Semitropic Ridge Preserve
Beer Nose
Rose
Mount Poso
Semitropic
Wasco
Belridge, North
Lost Hills
Antelope Hills
Buttonwillow Gas (Abd)
Shafter, North
Poso Creek
Round Mountain
Sequoia National Forest
Belridge, South
Cal Canal Gas
Rio Bravo
Rosedale Ranch
Kern Front
Kern River
Bowerbank
Greeley
Kern Bluff
Cymric
Lokern Preserve
Goosloo (Abd)
Fruitvale
Ant Hill
California Valley
Strand
Union Ave.
Edison
McKittrick
Canal
2800 Acre Water Bank
Belgian Anticline
Lokern Ecological Reserve
Coles Levee, North
Canfield Ranch
Stockdale
Asphalto
Elk Hills
Ten Section
Midway - Sunset
Coles Levee, South
Paloma
Mountain View
Buena Vista
Carrizo Plains Ecological Reserve
Carrizo Plain
Oil and gas administrative field boundaries
Well density class change
No change
Less than 1/km² to Low
Less than 1/km² or Low to Medium
Low or Medium to High
Land Use/Land Cover
Developed & Other Human Use
Agricultural/Introduced/Modified
Shrubland & Grassland
Forest & Woodland
Open Water
Barren
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0 5 10 20 Miles
N

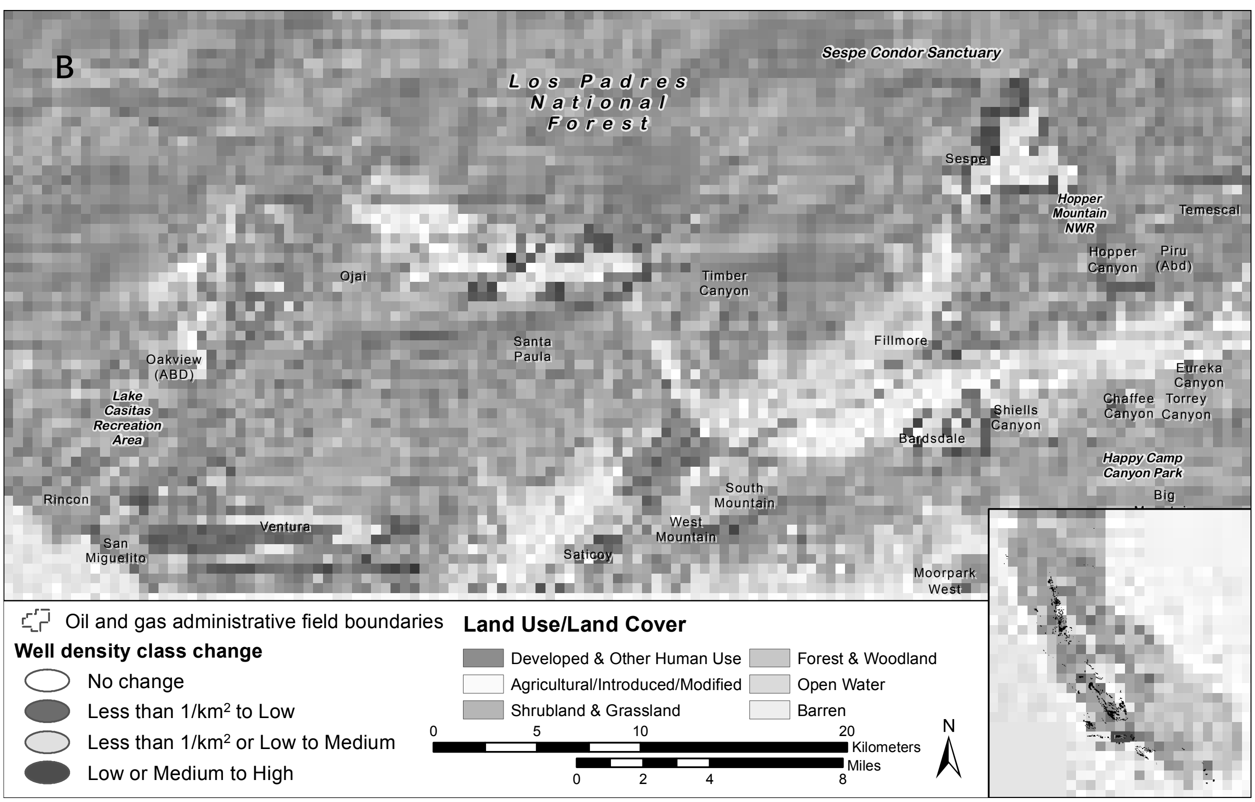
B
Sespe Condor Sanctuary
Los Padres National Forest
Sespe
Hopper Mountain NWR
Temescal
Hopper Canyon
Piru (Abd)
Ojai
Timber Canyon
Santa Paula
Fillmore
Oakview (ABD)
Eureka Canyon
Lake Casitas Recreation Area
Shiells Canyon
Chaffee Canyon
Torrey Canyon
Bardsdale
Happy Camp Canyon Park
South Mountain
Big
Rincon
Ventura
West Mountain
San Miguelito
Saticoy
Moorpark West
Oil and gas administrative field boundaries
Well density class change
No change
Less than 1/km² to Low
Less than 1/km² or Low to Medium
Low or Medium to High
Land Use/Land Cover
Developed & Other Human Use
Agricultural/Introduced/Modified
Shrubland & Grassland
Forest & Woodland
Open Water
Barren
0 5 10 20 Kilometers
0 2 4 8 Miles
N

Figure 1.3-1. Maps of (a) Kern and (b) Ventura Counties showing the increase in well density attributable to hydraulic fracturing-enabled development and land use/land cover between 1977 and 2014. We compared two scenarios for well density in California: actual well density, with all wells present; and a theoretical well density, without hydraulically fractured wells. Foreground colors show areas that have a higher well density with hydraulic fracturing-enabled production. Background shading shows land use/land cover. Kern and Ventura Counties each had oil fields where a substantial proportion of wells were enabled by hydraulic fracturing and where the underlying land use was undeveloped, open land (figure modified from Volume II, Chapter 5).

Ecologically sensitive areas require the conservation of habitat to compensate for new oil and gas development. Currently, no regional planning strategy exists to coordinate habitat conservation efforts in a manner that would ensure continued viable populations of rare species. While possible to compensate only for habitat loss caused by hydraulic fracturing-enabled development, a more logical approach would account for habitat loss from oil and gas production as a whole. Maintaining habitat connectivity in the southwestern San Joaquin will likely require slowing or halting increases in well pad density in dispersal corridors. This type of planning, such as the Kern County Valley Floor Habitat Restoration Plan, has not succeeded in the past, but a renewed effort would safeguard the survival of threatened and endangered species.

Recommendation 3.4. Minimize habitat loss and fragmentation in oil and gas producing regions.

Enact regional plans to conserve essential habitat and dispersal corridors for native species in Kern and Ventura Counties. The plans should identify top-priority habitat and restrict development of those areas. The plan should also define and require those practices, such as clustering multiple wells on a pad and using centralized networks of roads and pipes, which will minimize future surface disturbances. A program to set aside compensatory habitat in reserve areas when oil and gas development causes habitat loss and fragmentation should be developed and implemented (Volume II, Chapter 5; Volume III, Chapter 5 [San Joaquin Basin Case Study]).

1.3.2. Management of Produced Water from Hydraulically Fractured or Acid Stimulated Wells

Large volumes of water of various salinities and qualities get produced along with the oil. Oil reservoirs tend to yield increasing quantities of water over time, and most of California's oil reservoirs have been in production for several decades to over a century. For 2013, more than .48 billion m3 (3 billion barrels) of water came along with some .032 billion m3 (0.2 billion barrels) of oil in California. Operators re-inject some produced water back into the oil and gas reservoirs to help recover more petroleum and mitigate land subsidence. In other cases, farmers use this water for irrigation; often blending treated produced water with higher-quality water to reduce salinity. Disposal or reuse of produced water without proper precautions can cause contamination of groundwater and more so, if this water contains chemicals from hydraulic fracturing and acid stimulation. Underground injection of produced water can cause earthquakes.

Conclusion 4.1. Produced water disposed of in percolation pits could contain hydraulic fracturing chemicals.

Based on publicly available data, operators disposed of some produced water from stimulated wells in Kern County in percolation pits. The effluent has not been tested to determine if there is a measureable concentration of hydraulic fracturing chemical constituents. If these chemicals were present, the potential impacts to groundwater, human health, wildlife, and vegetation would be extremely difficult to predict, because there are so many possible chemicals, and the environmental profiles of many of them are unmeasured.

A commonly reported disposal method for produced water from stimulated wells in California is by evaporation and percolation in percolation surface impoundments, also referred to as percolation pits, as shown in Figure 1.3-2. Information from 2011 to 2014 indicates that operators dispose of some 40-60% of the produced water from hydraulically fractured wells in percolation pits during the first full month of production after stimulation. The range in estimated proportion stems from uncertainties about which wells were stimulated prior to mandatory reporting. Produced water from these wells may contain hazardous chemicals from hydraulic fracturing treatments, as well as reaction byproducts of those chemicals. We do not know how long hydraulic fracturing chemicals persist in produced water or at what concentrations or how these change in time, which means that hazardous levels of contaminants in produced water disposed into pits cannot be ruled out.

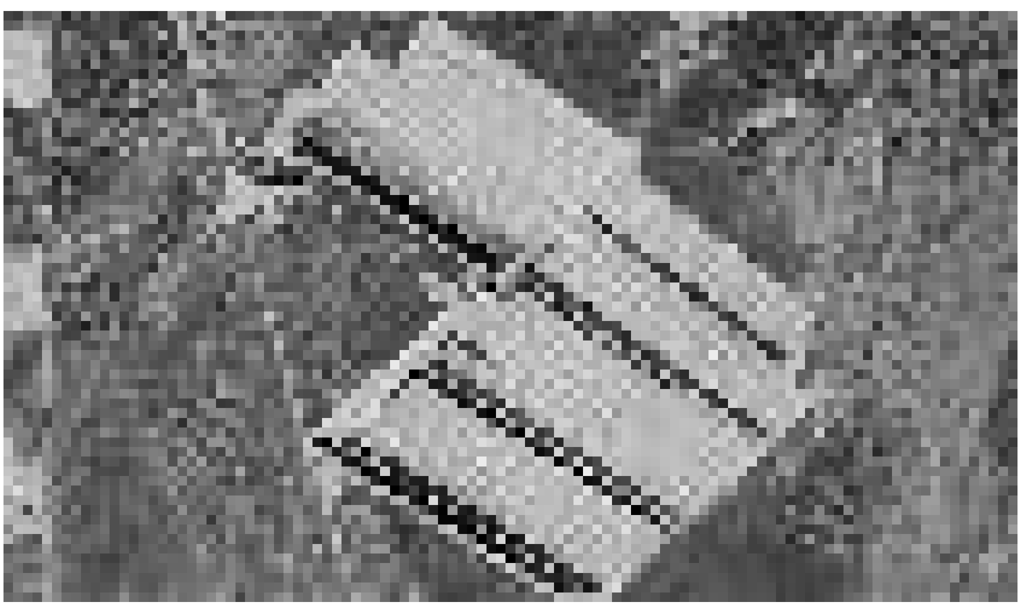

Figure 1.3-2. Percolation pits in Kern County used for produced water disposal (figure modified from Volume II, Chapter 1). Image courtesy of Google Earth.

The primary intent of percolation pits is to percolate water into the ground. This practice provides a potential direct pathway to transport produced water constituents, including returned hydraulic fracturing fluids, into groundwater aquifers. Groundwater contaminated in this way could subsequently intercept rivers, streams, and surface water resources. Contaminated water used by plants (including food crops), humans, fish, and wildlife could introduce contaminants into the food chain. Some states, including Kentucky, Texas and Ohio, have phased out the use of percolation pits for produced water disposal, because their use has demonstrably contaminated groundwater.

Operators have reported disposal of produced water in percolation pits in several California counties (e.g., Fresno, Monterey, and Tulare counties). However, records from 2011 to mid-2014 show that percolation pits received produced water from hydraulically fractured wells only in Kern County. Specifically, wells in the Elk Hills, South Belridge, North Belridge, Lost Hills, and Buena Vista fields were hydraulically fractured, and these fields disposed of produced water to percolation pits in the region under the jurisdiction of the CVRWQCB. An estimated 36% of percolation pits in the Central Valley operate without necessary permits from the CVRWQCB.

The data reported to DOGGR may contain errors on disposition of produced water. For example, DOGGR's production database shows that, during the past few years, one operator discharged produced water to percolation pits at Lost Hills, yet Central Valley Regional Water Quality Control Board (CVRWQCB) ordered the closure of percolation pits at Lost Hills in 2009.[2]

Data collected pursuant to the recent Senate Bill 1281 (SB 1281) will shed light on the disposition of produced water and locations of percolation pits statewide. With the data available as of the writing of this report, we cannot rule out that some produced water from hydraulically fractured wells at other fields went to percolation pits and that this water might have contained chemicals used in hydraulic fracturing. Figure 1.3-3 shows that many of these pits overlie protected groundwater. The pending well stimulation regulations, effective July 1, 2015, disallow fluid produced from a stimulated well from being placed in percolation pits.[3]

2. Order R5-2013-0056, Waste Discharge Requirements for Chevron USA, Inc., Central Valley Regional Water Quality Control Board.

3. Title 14 California Code of Regulations, Section 1786(a)(4)

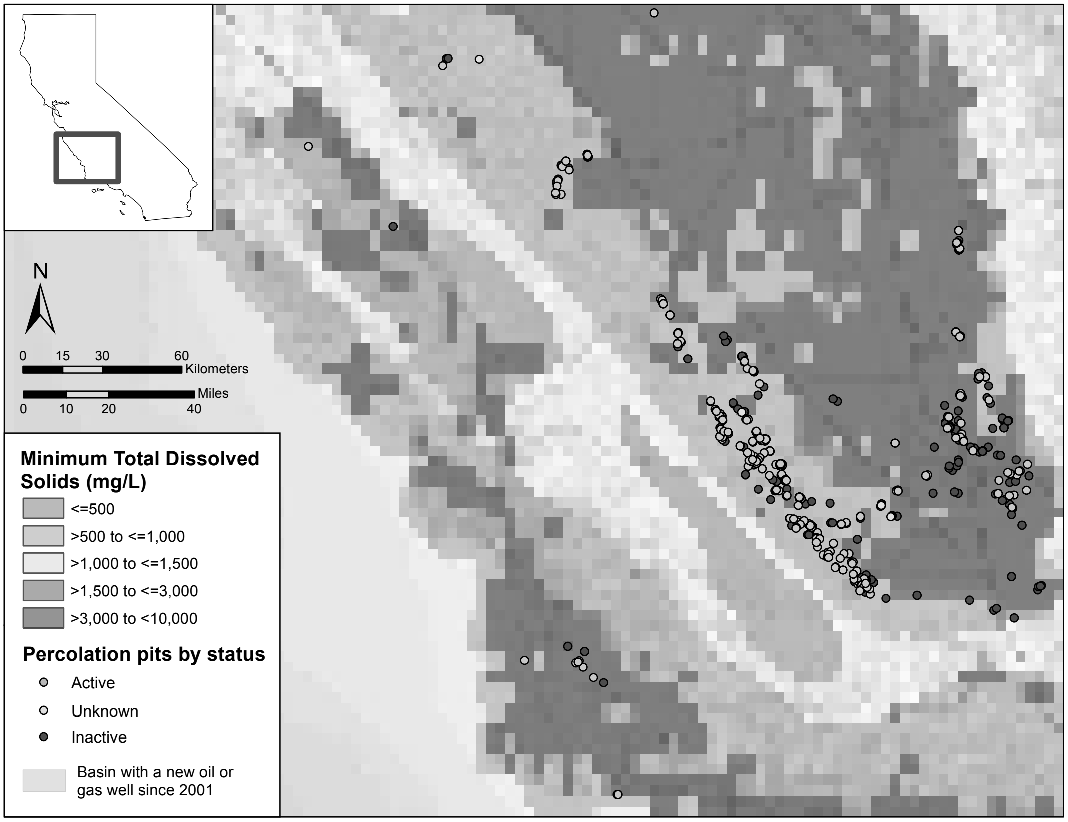

Figure 1.3-3. Location of percolation pits in the Central Valley and Central Coast used for produced water disposal and the location of groundwater of varying quality showing that many percolation pits are located in regions that have potentially protected groundwater shown in color (figure from Volume II, Chapter 2).

Recommendation 4.1. Ensure safe disposal of produced water in percolation pits with appropriate testing and treatment or phase out this practice.

Agencies with jurisdiction should promptly ensure through appropriate testing that the water discharged into percolation pits does not contain hazardous amounts of chemicals related to hydraulic fracturing as well as other phases of oil and gas development. If the presence of hazardous concentrations of chemicals cannot be ruled out, they should phase out the practice of discharging produced water into percolation pits. Agencies should investigate any legacy effects of discharging produced waters into percolation pits including the potential effects of stimulation fluids (Volume II, Chapter 2; Volume III, Chapters 4 and 5 [Los Angeles Basin and San Joaquin Basin Case Studies]).

Conclusion 4.2. The chemistry of produced water from hydraulically fractured or acid stimulated wells has not been measured.

Chemicals used in each hydraulic fracturing operation can react with each other and react with the rocks and fluids of the oil and gas reservoirs. When a well is stimulated with acid, the reaction of the acid with the rock minerals, petroleum, and other injected chemicals can release contaminants of concern in the oil reservoirs, such as metals or fluoride ions that have not been characterized or quantified. These contaminants may be present in recovered and produced water.

An average of about 25 different chemicals are used in each hydraulic fracturing operation. As discussed in Conclusion 3.2, some of these can be quite hazardous alone and chemical reactions can results in new constituents. Acids used in well treatments quickly react with rock minerals and become neutralized. But acids can dissolve and mobilize naturally occurring heavy metals and other pollutants in the oil-bearing formation. Neutralized hydrofluoric acid can release toxic fluoride ions into groundwater. Assessment of the environmental risks posed by hydraulic fracturing and acid use along with commonly associated chemicals, such as corrosion inhibitors, requires more complete disclosure of chemical use and a better understanding of the chemistry of treatment fluids and produced water returning to the surface. We found no characterization of the chemistry of produced water from wells that have been hydraulically fractured or stimulated with acid.

Recommendation 4.2. Evaluate and report produced water chemistry from hydraulically fractured or acid stimulated wells.

Evaluate the chemistry of produced water from hydraulically fractured and acid stimulated wells, and the potential consequences of that chemistry for the environment. Determine how this chemistry changes over time. Require reporting of all significant chemical use, including acids, for oil and gas development (Volume II, Chapters 2 and 6).

Conclusion 4.3. Required testing and treatment of produced water destined for reuse may not detect or remove chemicals associated with hydraulic fracturing and acid stimulation.

Produced water from oil and gas production has potential for beneficial reuse, such as for irrigation or for groundwater recharge. In fields that have applied hydraulic fracturing or acid stimulations, produced water may contain hazardous chemicals and chemical byproducts from well stimulation fluids. Practice in California does not always rule out the beneficial reuse of produced water from wells that have been hydraulically fractured or stimulated with acid. The required testing may not detect these chemicals, and the treatment required prior to reuse necessarily may not remove hydraulic fracturing chemicals.

Growing pressure on water resources in the state means more interest in using produced water for a range of beneficial purposes, such as groundwater recharge, wildlife habitat, surface waterways, irrigation, etc. Produced water could become a significant resource for California.

However, produced water from wells that have been hydraulically fractured may contain hazardous chemicals and chemical by-products. Our study found only one oil field where both hydraulic fracturing occurs and farmers use the produced water for irrigation. In the Kern River field in the San Joaquin Basin, hydraulic fracturing operations occasionally occur, and a fraction of the produced water goes to irrigation (for example, Figure 1.3-4). But we did not find policies or procedures that would necessarily exclude produced water from hydraulically fractured wells from use in irrigation.

Figure 1.3-4. Produced water used for irrigation in Cawelo water district. Photo credit: Lauren Sommer/KQED (figure from Volume II, Chapter 1).

The regional water quality control boards require testing and treatment of produced water prior to use for irrigation, but the testing does not include hydraulic fracturing chemicals, and required treatment would not necessarily remove hazardous stimulation fluid constituents if they were present. Regional water-quality control boards have also established monitoring requirements for each instance where produced water is applied to irrigated lands; however, these requirements do not include monitoring for constituents specific to, or indicative of, hydraulic fracturing.

Safe reuse of produced water that may contain stimulation chemicals requires appropriate testing and treatment protocols. These protocols should match the level of testing and treatment to the water-quality objectives of the beneficial reuse. However, designing the appropriate testing and treatment protocols to ensure safe reuse of waters contaminated with stimulation chemicals presents significant challenges, because so many different chemicals could be present, and the safe concentration limits for many of them have not been established. Hydraulic fracturing chemicals may be present in extremely small concentrations that present negligible risk, but this has not been confirmed.

Limiting hazardous chemical use as described in Recommendation 3.2 would also help to limit issues with reuse. Disallowing the reuse of produced water from hydraulically fractured wells would also solve this problem, especially in the first years of production. This water could be tested over time to determine if hazardous levels of hydraulic fracturing chemicals remain before transitioning this waste stream to beneficial use.

> **Recommendation 4.3. Protect irrigation water from contamination by hydraulic fracturing chemicals and stimulation reaction products.**
>
> *Agencies of jurisdiction should clarify that produced water from hydraulically fractured wells cannot be reused for purposes such as irrigation that could negatively impact the environment, human health, wildlife and vegetation. This ban should continue until or unless testing the produced water specifically for hydraulic fracturing chemicals and breakdown products shows non-hazardous concentrations, or required water treatment reduces concentrations to non-hazardous levels (Volume II, Chapter 2; Volume III, Chapter 5 [San Joaquin Basin Case Study]).*

Conclusion 4.4. Injection wells currently under review for inappropriate disposal into protected aquifers may have received water containing chemicals from hydraulic fracturing.

DOGGR is currently reviewing injection wells in the San Joaquin Valley for inappropriate disposal of oil and gas wastewaters into protected groundwater. The wastewaters injected into some of these wells likely included stimulation chemicals because hydraulic fracturing occurs nearby.

In 2014, DOGGR began to evaluate injection wells in California used to dispose of oil field wastewater. DOGGR found that some wells inappropriately allowed injection of wastewater into protected groundwater and subsequently shut them down. DOGGR's ongoing investigation will review many more wells to determine if they are injecting into aquifers that should be protected.

Figure 1.3-5 is a map of the Elks Hills field in the San Joaquin Basin showing one example where hydraulically fractured wells exist near active water disposal wells. The DOGGR review includes almost every disposal well in this field for possible inappropriate injection into protected water. Some of the produced water likely came from nearby production wells that were hydraulically fractured. Consequently, the injected wastewater possibly contained stimulation chemicals at some unknown concentration.

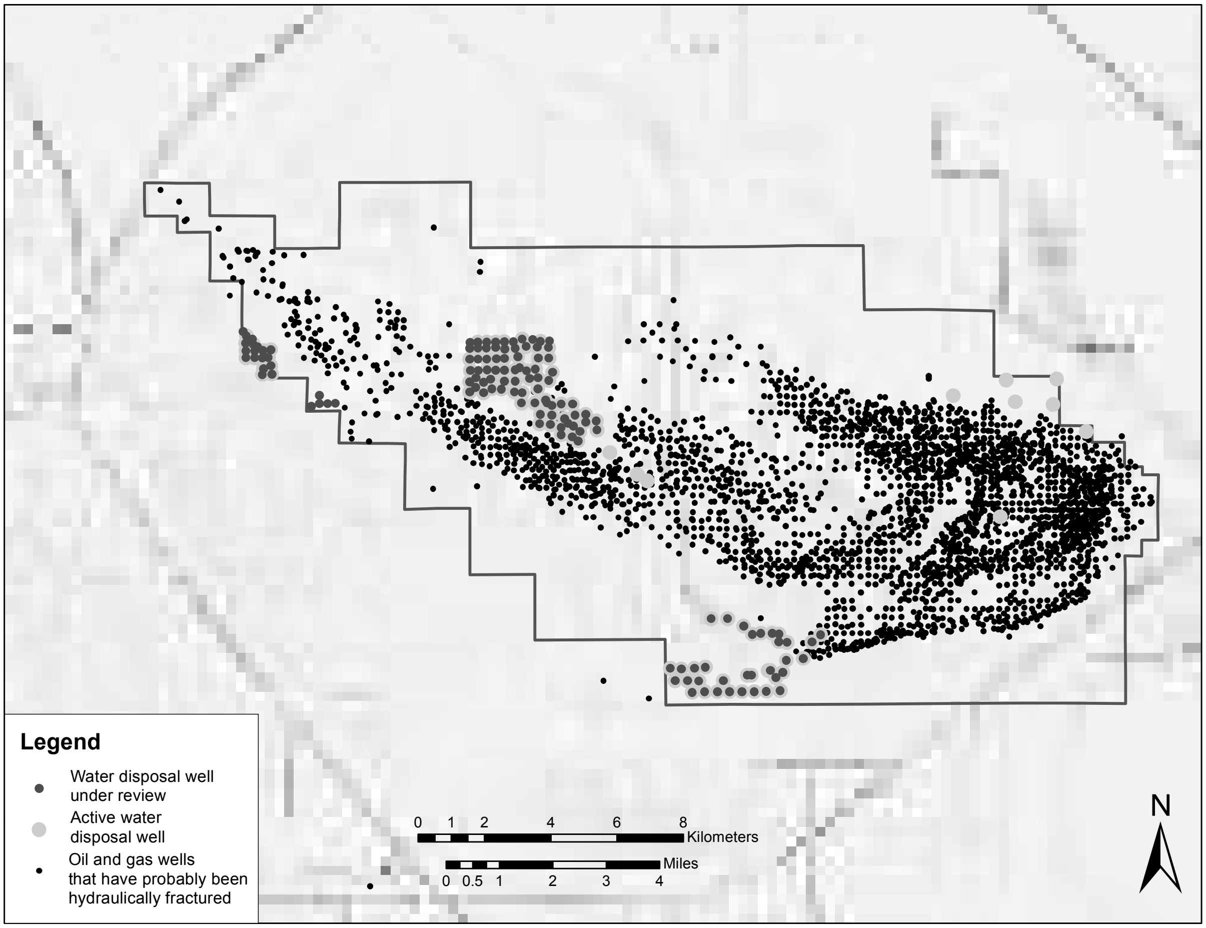

Figure 1.3-5. A map of the Elk Hills field in the San Joaquin Basin showing the location of wells that have probably been hydraulically fractured (black dots). Blue dots are the location of active water disposal wells, and blue dots with a red center are the location of disposal wells under review for possibly injecting into groundwater that should be protected (figure from Volume II, Chapter 1).

Recommendation 4.4. In the ongoing investigation of inappropriate disposal of wastewater into protected aquifers, recognize that hydraulic fracturing chemicals may have been present in the wastewater.

In the ongoing process of reviewing, analyzing, and remediating the potential impacts of wastewater injection into protected groundwater, agencies of jurisdiction should include the possibility that hydraulic fracturing chemicals may have been present in these wastewaters (Volume II, Chapter 2; Volume III, Chapter 5 [San Joaquin Basin Case Study]).

Conclusion 4.5. Disposal of wastewater by underground injection has caused earthquakes elsewhere.

Fluid injected in the process of hydraulic fracturing will not likely cause earthquakes of concern. In contrast, disposal of produced water by underground injection could cause felt or damaging earthquakes. To date, there have been no reported cases of induced seismicity associated with produced water injection in California. However, it can be very difficult to distinguish California's frequent natural earthquakes from those possibly caused by water injection into the subsurface.

Hydraulic fracturing causes a pressure increase for a short amount of time and affects relatively small volumes of rock. For this reason, hydraulic fracturing has a small likelihood of producing felt (*i.e.*, sensed), let alone damaging, earthquakes. In California, only one small earthquake (which occurred in 1991) has been linked to hydraulic fracturing to date (Volume II, Chapter 4).

Disposal into deep injection wells of water produced from oil and gas operations has caused felt seismic events in several states, but there have been no reported cases of induced seismicity associated with wastewater injection in California. The volume of produced water destined for underground injection could increase for a number of reasons, and disposal of increased volumes by injection underground could increase seismic hazards.

California has frequent naturally occurring earthquakes—so many that seismologists have a hard time determining if any of these earthquakes were actually induced by fluid injection. In areas like Kansas that do not have frequent earthquakes, it is much easier to find correlations between an earthquake and human activity. In the future, the amount of fluid requiring underground injection in California could increase locally due to expanded production or a change in disposal practice. Such change in practice might incur an unacceptable seismic risk, but understanding this possible risk requires a better understanding of the current correlation between injection and earthquakes, if any.

California also has many geologic faults. Figure 1.3-6 shows a map of California earthquake epicenters, the location of wastewater disposal wells active since 1981 and faults in the United States Geological Survey (USGS) database in central and southern California. Across all six oil-producing basins, over 1,000 wells are located within 2.5 km (1.5 miles) of a mapped active fault, and more than 150 within 200 m (650 ft).

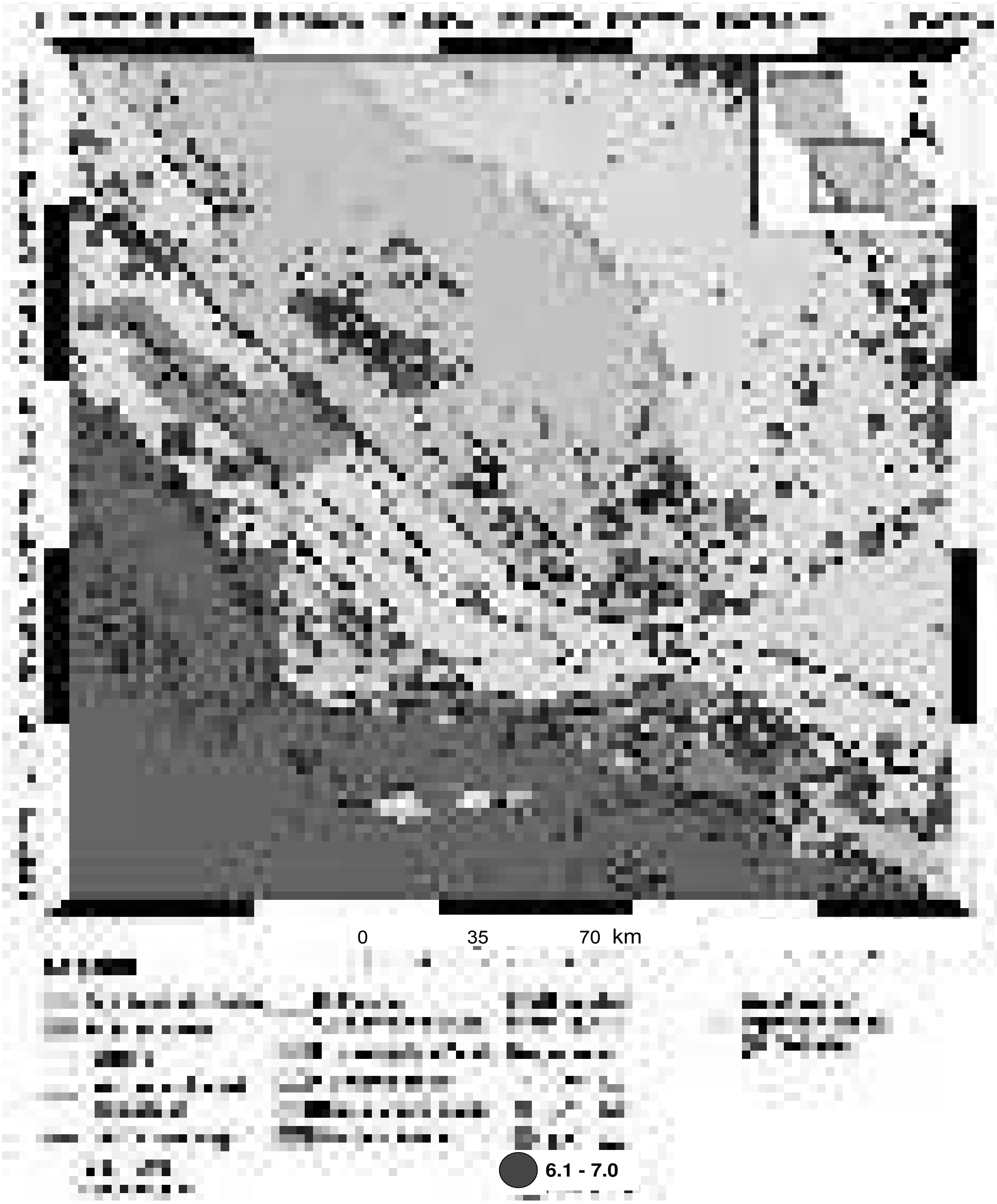

Figure 1.3-6. High-precision locations for earthquakes M≥3 in central and southern California during the period 1981-2011, and active and previously active water disposal wells from DOGGR (figure from Volume II, Chapter 4).

A systematic regional-scale analysis of earthquake occurrence in relation to water injection would help identify if induced seismicity exists in California. This study should include statistical characterizations and geomechanical analysis for induced seismicity and will require more detailed data than that currently reported by industry on injection depth, variations in fluid injection rate, and pressure over time. Currently, operators report the volume of injected water and wellhead pressures only as monthly averages. Analysts will need to know more about exactly when, how, how much, where injection occurred to identify a potential relationship between earthquakes and injection patterns. A systematic study will also require geophysical characterization of oil field test sites, detailed seismic monitoring, and modeling of the subsurface pressure changes produced by injection in the vicinity of the well.

The state could likely manage and mitigate potential induced seismicity, by adopting protocols to modify an injection operation when and if seismic activity is detected. The protocol could require reductions in injection flow rate and pressure, and shutting down the well altogether if the risk of an earthquake rises above some threshold. Currently, ad hoc protocols exist for this purpose. Better protocols would require monitoring the reservoir and local seismic activity, and formal calculation of the probability of inducing earthquakes of concern.

> **Recommendation 4.5. Determine if there is a relationship between wastewater injection and earthquakes in California.**
>
> *Conduct a comprehensive multi-year study to determine if there is a relationship between oil and gas-related fluid injection and any of California's numerous earthquakes. In parallel, develop and apply protocols for monitoring, analyzing, and managing produced water injection operations to mitigate the risk of induced seismicity. Investigate whether future changes in disposal volumes or injection depth could affect potential for induced seismicity (Volume II, Chapter 4).*

Conclusion 4.6. Changing the method of wastewater disposal will incur tradeoffs in potential impacts.

Based on publicly available data, operators dispose of much of the produced water from stimulated wells in percolation pits (evaporation-percolation ponds), about a quarter by underground injection (in Class II wells), and less than one percent to surface bodies of water. Changing the method of produced water disposal could decrease some potential impacts while increasing others.

Figure 1.3-7 shows the results of an analysis of disposal methods of produced water from known stimulated wells in the first full month after stimulation during the period from 2011 to 2014. As much as 60% of the water was sent to percolation pits, also known as evaporation-percolation ponds, as discussed in Conclusion 4.1 Second to this, produced water from stimulated wells was injected into Class II wells for disposal or enhanced oil recovery. With proper regulation, siting, construction, and maintenance, subsurface injection is less likely to result in groundwater contamination than disposal in percolation pits.

However, increasing injection volumes could increase the risk of induced seismicity, discussed in Conclusion 4.5. Also, concerns have recently emerged about whether California's Class II underground injection control (UIC) program provides adequate protection for underground sources of drinking water (USDWs), as discussed in Conclusion 4.4, USDWs are defined as groundwater aquifers that currently or could one day supply water for human consumption. The least common method of dealing with wastewater, disposal to surface bodies of water, can, for example, augment stream flows, but requires careful testing and treatment to ensure the water is safe, especially if stimulation chemicals could be present.

The DOGGR monthly production data either do not specify the disposal method or report as "other" for 17% of the produced water from known stimulated wells. This reporting category could include subsurface injection, disposal to a surface body of water, sewer disposal, or water not disposed of but reused for irrigation or another beneficial purpose, as described in Conclusion 4.3.

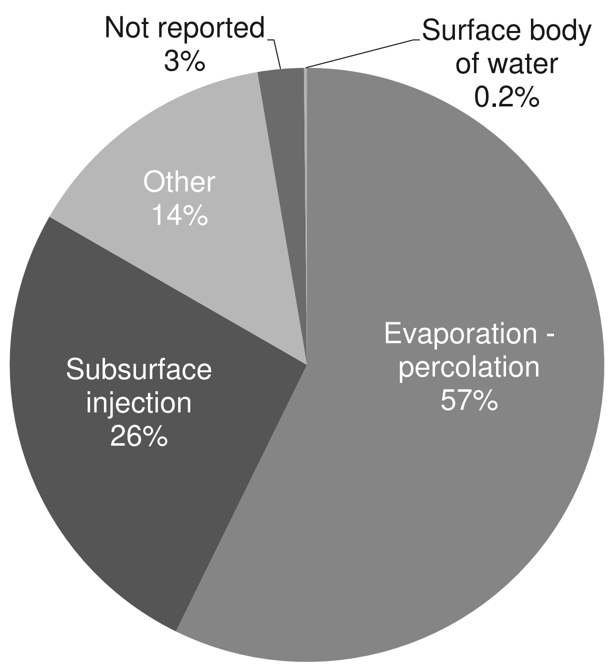

Figure 1.3-7. Disposal method for produced water from hydraulically fractured wells during the first full month after stimulation for the time period 2011-2014 based on data from DOGGR monthly production database. Note: Subsurface injection includes any injection into Class II wells, which include disposal wells as well as enhanced recovery wells used for water flooding and steam flooding (figure from Volume II, Chapter 2).

Changing the method of produced water disposal or reuse will incur tradeoffs. Any attempt to reduce one disposal method must consider the likely outcome that other disposal methods will increase. For example, eliminating disposal in evaporation–percolation pits can lead to an increase in other disposal methods to make up the difference. In particular, closure of percolation pits or injection wells found to be contaminating protected aquifers would increase the use of other disposal methods, and this will require careful planning and management on a regional basis.

Recommendation 4.6. Evaluate tradeoffs in wastewater disposal practices.

As California moves to change disposal practices, for example by phasing out percolation pits or stopping injection into protected aquifers, agencies with jurisdiction should assess the consequences of modifying or increasing disposal via other methods (Volume II, Chapter 2; Volume II, Chapter 4).

1.3.3. Protections to Avoid Groundwater Contamination by Hydraulic Fracturing

Hydraulic fracturing operations could contaminate groundwater through a variety of pathways. We found no documented instances of hydraulic fracturing or acid stimulations directly causing groundwater contamination in California. However, we did find that fracturing in California tends to be in shallow wells and in mature reservoirs that have many existing boreholes. These practices warrant more attention to ensure that they have not and will not cause contamination.

Conclusion 5.1. Shallow fracturing raises concerns about potential groundwater contamination.

In California, about three quarters of all hydraulic fracturing operations take place in shallow wells less than 600 m (2,000 ft) deep. In a few places, protected aquifers exist above such shallow fracturing operations, and this presents an inherent risk that hydraulic fractures could accidentally connect to the drinking water aquifers and contaminate them or provide a pathway for water to enter the oil reservoir. Groundwater monitoring alone may not necessarily detect groundwater contamination from hydraulic fractures. Shallow hydraulic fracturing conducted near protected groundwater resources warrants special requirements and plans for design control, monitoring, reporting, and corrective action.

Hydraulic fractures produced in deep formations far beneath protected groundwater are very unlikely to propagate far enough upwards to intersect an aquifer. Studies performed for high-volume hydraulic fracturing elsewhere in the country have shown that hydraulic fractures have propagated no further than 600 m (2,000 ft) vertically, so hydraulic fracturing conducted many thousands of feet below an aquifer is not expected to reach a protected aquifer far above. In California, however, and particularly in the San Joaquin Basin, most hydraulic fracturing occurs in relatively shallow reservoirs, where protected groundwater might be found within a few hundred meters (Figure 1.3-8). A few instances of shallow fracturing have also been reported in the Los Angeles Basin (Figure 1.3-9), but overall much less than the San Joaquin Basin. No cases of contamination have yet been reported, but there has been little to no systematic monitoring of aquifers in the vicinity of oil production sites.

Shallow hydraulic fracturing presents a higher risk of groundwater contamination, which groundwater monitoring may not detect. This situation warrants additional scrutiny. Operations with shallow fracturing near protected groundwater could be disallowed or be subject to additional requirements regarding design, control, monitoring, reporting, and corrective action, including: (1) pre-project monitoring to establish a base-line of chemical concentrations, (2) detailed prediction of expected fracturing characteristics prior to starting the operation, (3) definition of isolation between expected fractures and protected groundwater, providing a sufficient safety margin with proper weighting of subsurface uncertainties, (4) targeted monitoring of the fracturing operation to watch for and react to evidence (e.g., anomalous pressure transients, microseismic signals) indicative of fractures growing beyond their designed extent, (5) monitoring groundwater to detect leaks, (6) timely reporting of the measured or inferred fracture characteristics confirming whether or not the fractures have actually intersected or come close to intersecting groundwater, (7) preparing corrective action and mitigation plans in case anomalous behavior is observed or contamination is detected, and (8) adaption of groundwater monitoring plans to improve the monitoring system and specifically look for contamination in close proximity to possible fracture extensions into groundwater.

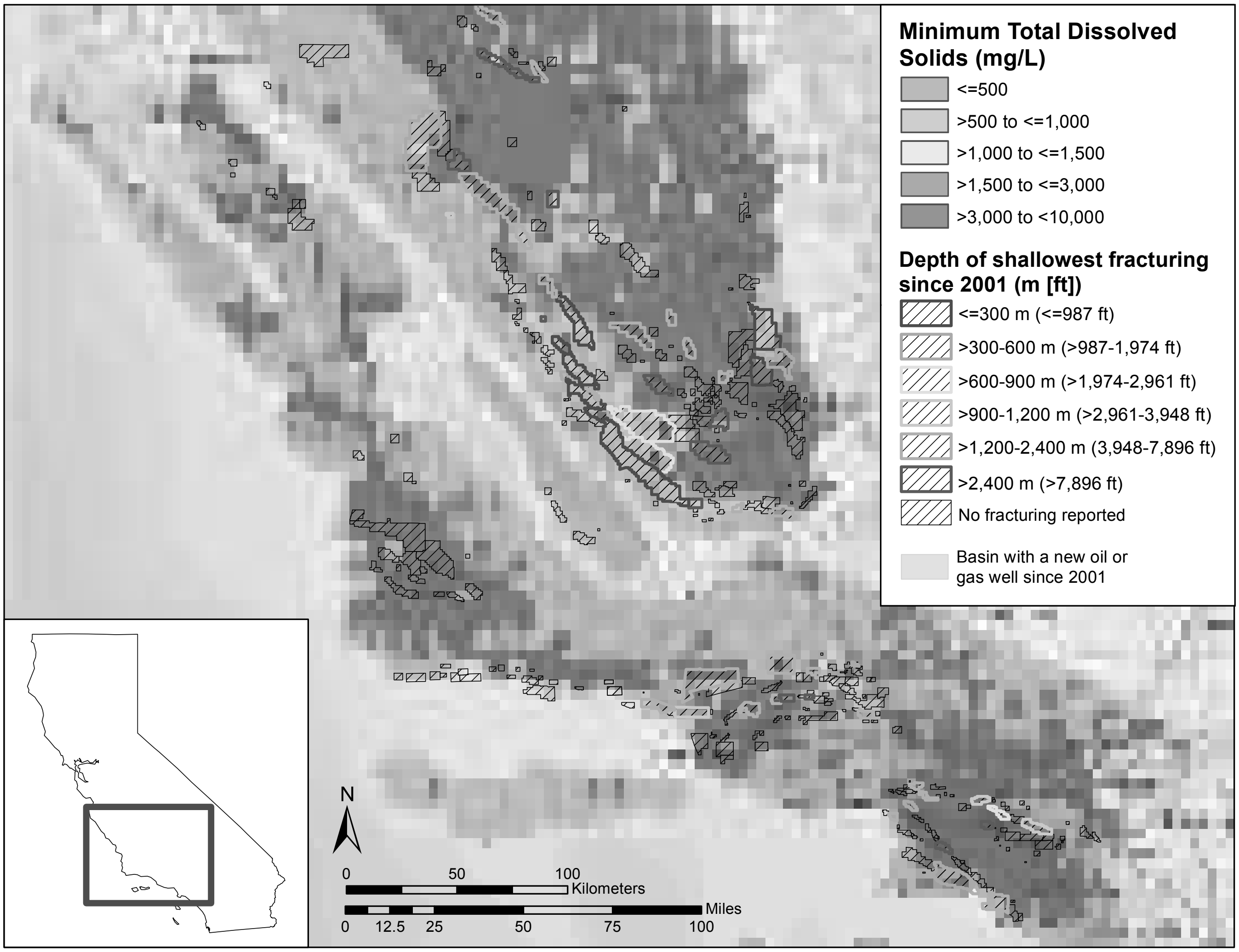

Figure 1.3-8. Shallow fracturing locations and groundwater quality in the San Joaquin and Los Angeles Basins. Some high quality water exists in fields that have shallow fractured wells (figure from Volume II, Chapter 2).

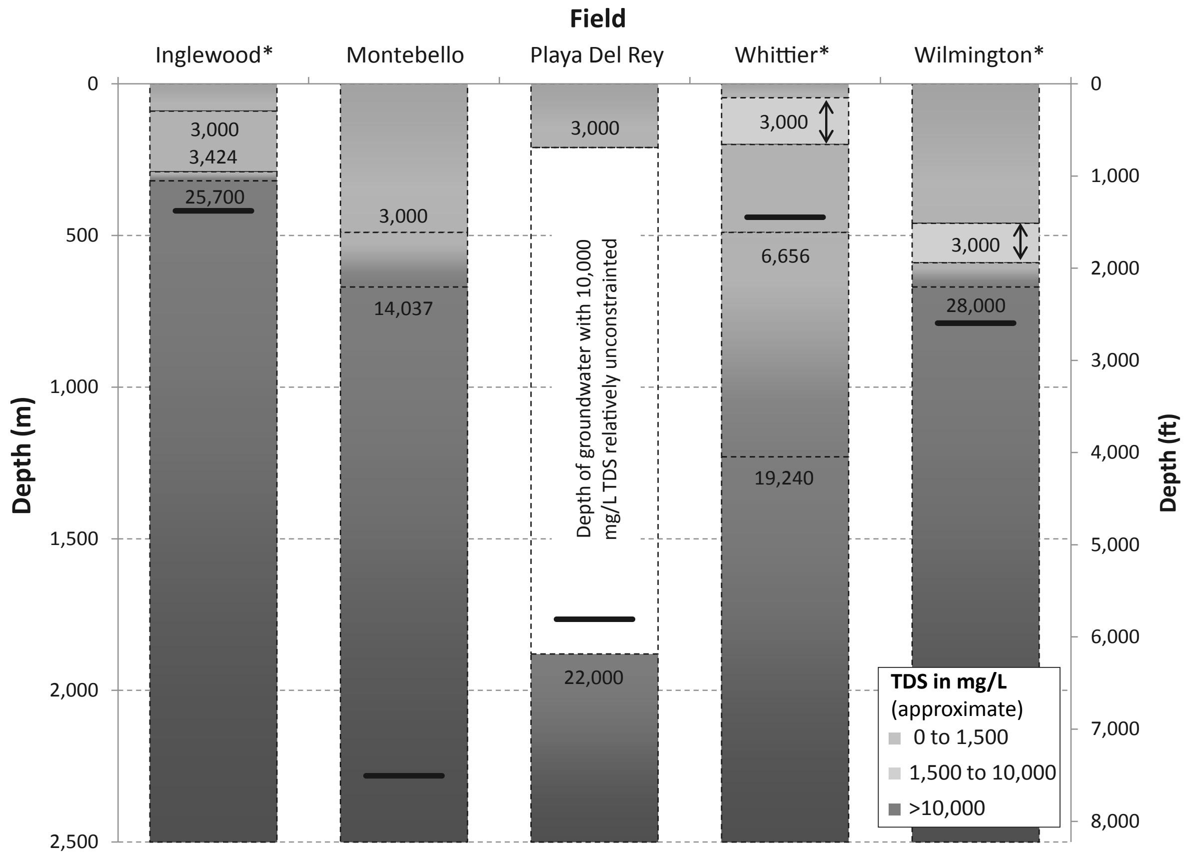

Figure 1.3-9. Depths of groundwater total dissolved solids (TDS) in mg/L in five oil fields in the Los Angeles Basin. The numbers indicate specific TDS data and the colors represent approximate interpolation. The depth of 3,000 mg/L TDS is labeled on all five fields. Blue (<3,000 mg/L) and aqua (between 3,000 mg/L and 10,000 mg/L) colors represent protected groundwater. Depth of 10,000 mg/L TDS is uncertain, but it is estimated to fall in the range where aqua transitions to brown. The heavy black horizontal line indicates the shallowest hydraulically fractured well interval in each field. (Asterisks denote the fields of most concern for the proximity of hydraulic fracturing to groundwater with less than 10,000 mg/L TDS.) (figure from Volume III, Chapter 4 [Los Angeles Basin Case Study]).

The potential for shallow hydraulic fractures to intercept protected groundwater requires both knowing the location and quality of nearby groundwater and accurate information about the extent of the hydraulic fractures. Maps of the vertical depth of protected groundwater with less than 10,000 mg/L TDS for California oil producing regions do not yet exist. Analysis and field verification could identify typical hydraulic fracture geometries; this would help determine the probability of fractures extending into groundwater aquifers. Finally, detection of potential contamination and planning of mitigation measures requires integrated site-specific and regional groundwater monitoring programs.

The pending SB 4 well stimulation regulations, effective July 1, 2015, require operators to design fracturing operations so that the fractures avoid protected water, and to implement appropriate characterization and groundwater monitoring near hydraulic fracturing operations. However, groundwater monitoring alone does not ensure protection of water, nor will it necessarily detect contamination should it occur. The path followed by contamination underground can be hard to predict, and may bypass a monitoring well. Groundwater monitoring can give false negative results in these cases,[4] and does nothing to stop contamination from occurring in any case.

> **Recommendation 5.1. Protect groundwater from shallow hydraulic fracturing operations.**
>
> *Agencies with jurisdiction should act promptly to locate and catalog the quality of groundwater throughout the oil-producing regions. Operators proposing to use hydraulic fracturing operation near protected groundwater resources should be required to provide adequate assurance that the expected fractures will not extend into these aquifers and cause contamination. If the operator cannot demonstrate the safety of the operation with reasonable assurance, agencies with jurisdiction should either deny the permit, or develop protocols for increased monitoring, operational control, reporting, and preparedness (Volume I, Chapter 3; Volume II, Chapter 2; Volume III, Chapter 5 [San Joaquin Basin Case Study]).*

Conclusion 5.2. Leakage of hydraulic fracturing chemicals could occur through existing wells.

California operators use hydraulic fracturing mainly in reservoirs that have been in production for a long time. Consequently, these reservoirs have a high density of existing wells that could form leakage paths away from the fracture zone to protected groundwater or the ground surface. The pending SB 4 regulations going into effect July 1, 2015 do address concerns about existing wells in the vicinity of well stimulation operations; however, it remains to demonstrate the effectiveness of these regulations in protecting groundwater.

In California, most hydraulic fracturing occurs in old reservoirs where oil and gas has been produced for a long time. Usually this means many other wells (called “offset wells”) have previously been drilled in the vicinity of the operation. Wells constructed to less stringent regulations in the past or degraded since installation may not withstand the high pressures

4. Chemical tracers (non-reactive chemicals that can be detected in small concentrations) can be added to hydraulic fracturing fluids and, if groundwater samples contain these tracers, it is evidence that the stimulation fluid has migrated out of the designed zone. However, the use of tracers does not guarantee that leaks to groundwater will be detected. Groundwater flow can be highly channelized and it can be difficult to place a monitoring well in the right place to intersect a possible plume of contaminant. The use of tracers is good practice, but does not “solve” the problem of detecting contamination.

used in hydraulic fracturing. Thus, in California, as well as in other parts of the country, existing oil and gas wells can provide subsurface conduits for oil-field contamination to reach protected groundwater. Old wells present a risk for any oil and gas development, but the high pressures involved in hydraulic fracturing can increase this risk significantly. California has no recorded incidents of groundwater contamination due to stimulation. But neither have there been attempts to detect such contamination with targeted monitoring, nor studies to determine the extent of compromised wellbore integrity.

Historically, California has required placement of well casings and cement seals to protect groundwater with a salinity less than 3,000 mg/L total dissolved solids (TDS). Now, SB 4 requires more stringent monitoring and protection from degradation of non-exempt groundwater with less than 10,000 mg/L TDS. Consequently, existing wells may not have been built to protect groundwater between 3,000 mg/L and 10,000 mg/L TDS. For instance, there may be no cement seal in place to isolate the zones containing water that is between 3,000 and 10,000 mg/L TDS from deeper zones with water that is higher than 10,000 mg/L TDS.

The new well stimulation regulations going into effect in July 1, 2015 require operators to locate and review any existing well within a zone that is twice as large as the expected fractures. Operators need to design the planned hydraulic fracturing operation to confine hydraulic fracturing fluids and hydrocarbons within the hydrocarbon formation. The pressure buildup at offset wells caused by neighboring hydraulic fracturing operations must remain below a threshold value defined by the regulations.

The new regulations for existing wells are appropriate in concept, but the effectiveness of these requirements will depend on implementation practice. For example: How will operators estimate the extent of the fractures, and how will regulators ensure the reliability of these calculations? Is the safety factor provided by limiting concern to an area equal to twice the extent of the designed fractures adequate? How will regulators assess the integrity of existing wells when information about these wells is incomplete? How will regulators determine the maximum allowed pressure experienced at existing wells? Will the regulators validate the theoretical calculations to predict fracture extent and maximum pressure with field observations?

Recommendation 5.2. Evaluate the effectiveness of hydraulic fracturing regulations designed to protect groundwater from leakage along existing wells.

Within a few years of the new regulations going into effect, DOGGR should conduct or commission an assessment of the regulatory requirements for existing wells near stimulation operations and their effectiveness in protecting groundwater with less than 10,000 TDS from well leakage. This assessment should include comparisons of field observations from hydraulic fracturing sites with the theoretical calculations for stimulation area or well pressure required in the regulations (Volume II, Chapter 2; Volume III, Chapters 4 and 5 [Los Angeles Basin and San Joaquin Basin Case Studies]).

1.3.4. Emissions and their Impact on Environmental and Human Health

Gaseous emissions and particulates associated with hydraulic fracturing can arise from the use of fossil fuel in engines, outgassing from fluids, leaks, or proppant, which have potential environmental or health impacts.

Conclusion 6.1. Oil and gas production from hydraulically fractured reservoirs emits less greenhouse gas per barrel of oil than other forms of oil production in California.

Burning fossil fuel to run vehicles, make electricity, and provide heat accounts for the vast majority of California's greenhouse gas emissions. In comparison, publicly available California state emission inventories indicate that oil and gas production operations emit about 4% of California total greenhouse gas emissions. Oil and gas production from hydraulically fractured reservoirs emits less greenhouse gas per barrel of oil than production using steam injection. Oil produced in California using hydraulic fracturing also emits less greenhouse gas per barrel than the average barrel imported to California. If the oil and gas derived from stimulated reservoirs were no longer available, and demand for oil remained constant, the replacement fuel could have larger greenhouse emissions.

Most oil-related greenhouse gas (GHG) emissions in the state come from the consumption of fossil fuels such as gasoline and diesel, not the extraction of oil. According to state emission inventories, GHG emissions from oil and gas production processes equal about four percent of total GHG emissions in California, although some studies conclude these emission inventories may underestimate true emissions. Fields with lighter oil result in low emissions per barrel of crude produced, while fields with heavier oil have higher emissions because of the need for steam injection during production as well as more intensive refining needed to produce useful fuels such as gasoline. Well stimulation generally applies to reservoirs with lighter oil and consequently smaller greenhouse gas burdens per unit of oil. Oil and gas from San Joaquin Basin reservoirs using hydraulic fracturing have a relatively smaller carbon footprint than oil and gas from reservoirs such as those in the Kern River field that use steam flooding (Figure 1.3-10).

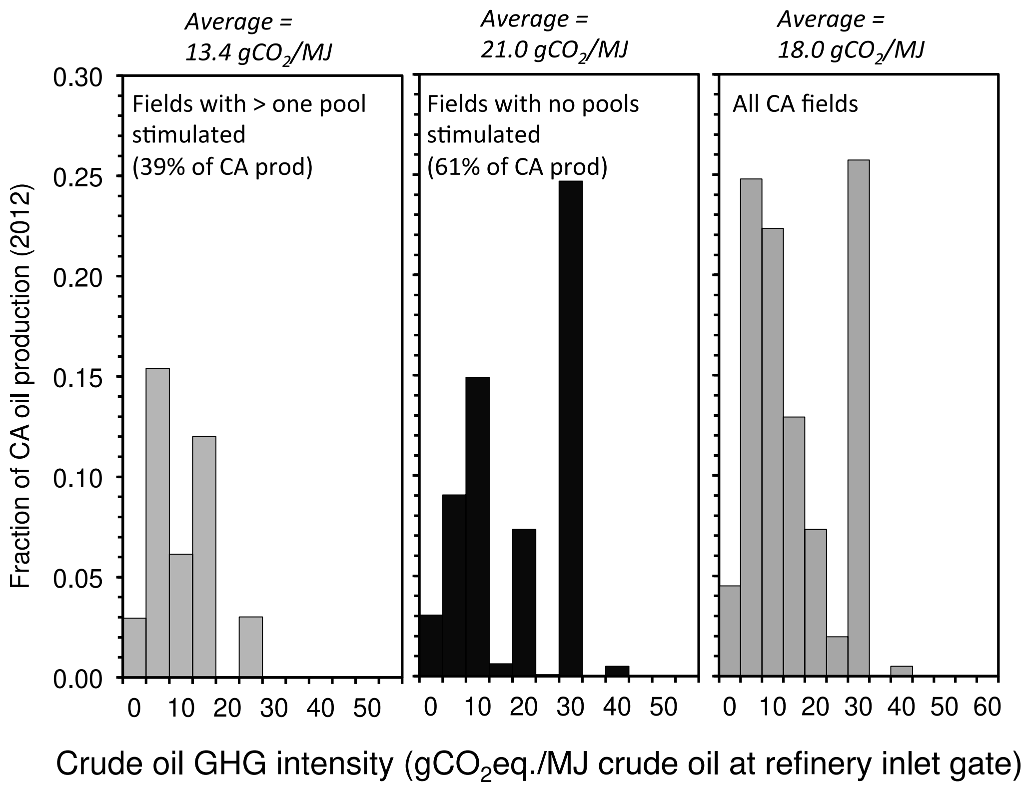

Figure 1.3-10. Distribution of crude oil greenhouse gas intensity for fields containing well-stimulation-enabled pools (left), those that are not stimulated (middle) and all California oilfields (right) (figure from Volume II, Chapter 3).

If well stimulation were disallowed and consumption of oil and gas in California did not decline, more oil and gas would be required from non-stimulated California fields or regions outside of California, possibly with higher emissions per barrel. Consequently, overall greenhouse gas emissions due to production could increase if well stimulation were stopped in California. The net greenhouse gas change associated with the use of hydraulic fracturing requires knowing the carbon footprint of both in-state and out-of-state production, and understanding the scale of impact requires a market-informed life cycle analysis.

Recommendation 6.1. Assess and compare greenhouse gas signatures of different types of oil and gas production in California.

Conduct rigorous market-informed life-cycle analyses of emissions impacts of different oil and gas production to better understand GHG impacts of well stimulation (Volume II, Chapter 3).

Conclusion 6.2. Air pollutants and toxic air emissions[5] from hydraulic fracturing are mostly a small part of total emissions, but pollutants can be concentrated near production wells.

According to publicly available California state emission inventories, oil and gas production in the San Joaquin Valley air district likely accounts for significant emissions of sulfur oxides (SO_x), volatile organic compounds (VOC), and some air toxics, notably hydrogen sulfide (H_2S). In other oil and gas production regions, production as a whole accounts for a small proportion of total emissions. Hydraulic fracturing facilitates about 20% of California production, and so emissions associated with this production also represent about 20% of all emissions from the oil and gas production in California. Even where the proportion of air pollutants and toxic emissions caused directly or indirectly by well simulation is small, atmospheric concentrations of pollutants near production sites can be much larger than basin or regional averages, and could potentially cause health impacts.

In the San Joaquin Valley oil and gas production as a whole accounts for about 30% of sulfur oxides and 8% of anthropogenic volatile organic compound (VOC) emissions. VOCs in turn react with nitrogen oxides (NO_x) to create ozone. Eliminating emissions from oil and gas production would reduce, but not eliminate the difficult air pollution problems in the San Joaquin Valley. Oil and gas facilities also emit significant air toxics in the San Joaquin Valley. They are responsible for a large fraction (>70%) of total hydrogen sulfide emissions and small fractions (2-6%) of total benzene, xylene, hexane, and formaldehyde emissions (Figure 1.3-11). Dust ($PM_{2.5}$ and PM_{10}) is a major air quality concern in the San Joaquin Valley, and agriculture is the dominant source of dust in the region. The amount of dust generated by oil and gas activities (including hydraulic fracturing) is comparatively very small.

5. Toxic air pollutants, also known as hazardous air pollutants, are those pollutants that are known or suspected to cause cancer or other serious health effects, such as reproductive effects or birth defects, or adverse environmental effects. Criteria air contaminants (CAC), or criteria pollutants, are a set of air pollutants that cause smog, acid rain, and other health hazards.

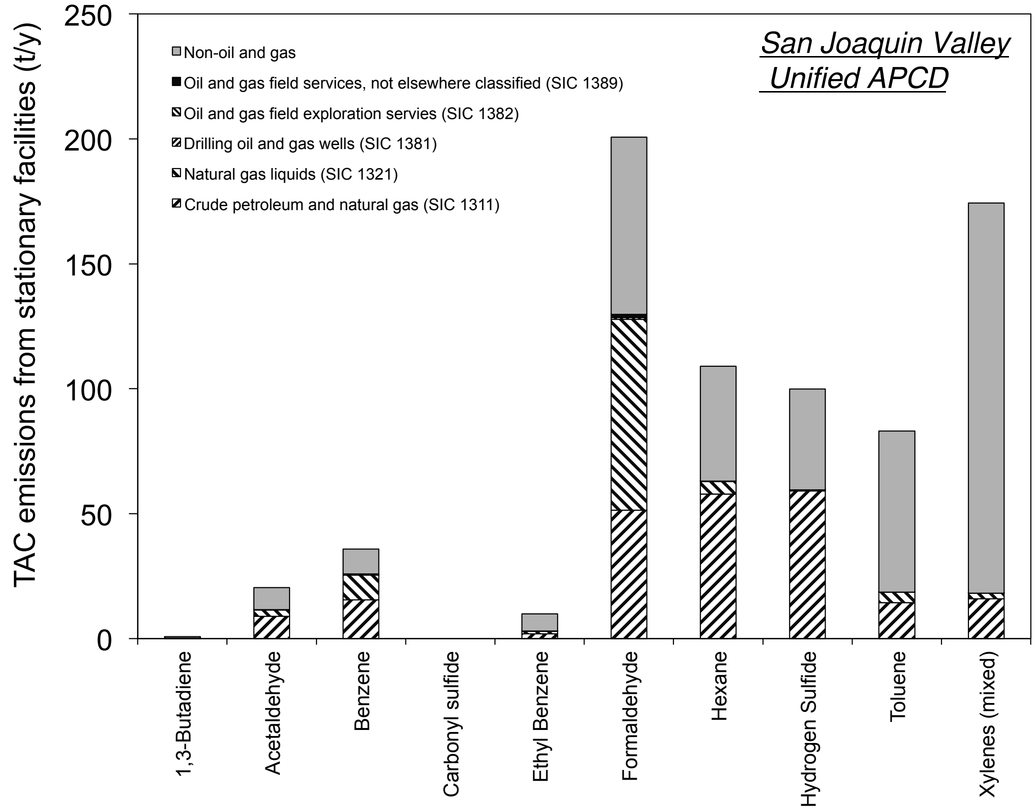

Figure 1.3-11. Summed facility-level toxic air contaminant (TAC) emissions in San Joaquin Valley air district). Facility-level emissions derived from a California Air Resources Board (CARB) facility emissions tool. Total emissions are emissions from all oil and gas facilities in the air district, including gasoline fueling stations (Volume II, Chapter 3) (figure from Volume II, Chapter 3).

In the South Coast Air District (including all of Orange County, the non-desert regions of Los Angeles and Los Angeles County, San Bernardino County, and Riverside County), upstream oil and gas sources represent small proportions (<1%) of criteria air pollutant and toxic air contaminant emissions due to large quantities of emissions from other sources in a highly urbanized area.

> *Produced gas can be emitted during recovery of hydraulic fracturing liquids and therefore be a possible source of direct air emissions from well stimulation. Regulation and control technologies can address these emissions with proper implementation and enforcement. Federal regulations already control emissions during fluid recovery from new gas wells using "green completions," and California is developing similar regulations for oil wells.*

Public data sources provide information about the emissions from all upstream oil and gas production, but do not include information that would allow separating out the contribution of emissions from hydraulically fractured wells. Because well stimulation facilitates or enables about 20% of California's oil recovery, indirect air impacts from well stimulation are likely on the order of one-fifth of total upstream oil and gas air impacts.

Even if upstream oil and gas operations are not a large part of basin-wide air pollution load, at the scale of counties, cities or neighborhoods, oil and gas development can have larger proportional impacts. Even in regions where well stimulation-related emissions represent a small part of overall emissions, local air toxic concentrations near drilling and production sites may be elevated. This could result in health impacts in densely populated areas such as Los Angeles, where production wells are in close proximity to homes, schools, and businesses. Public datasets do not provide specific enough temporal and spatial data on air toxics emissions that would allow any realistic assessment of these impacts.

Recommendation 6.2. Control toxic air emissions from oil and gas production wells and measure their concentrations near productions wells.

Apply reduced-air-emission completion technologies to production wells, including stimulated wells, to limit direct emissions of air pollutants, as planned. Reassess opportunities for emission controls in general oil and gas operations to limit emissions. Improve specificity of inventories to allow better understanding of oil and gas emissions sources. Conduct studies to improve our understanding of toxics concentrations near stimulated and un-stimulated wells (Volume II, Chapter 3; Volume III, Chapter 4 [Los Angeles Basin Case Study]).

Conclusion 6.3. Emissions concentrated near all oil and gas production could present health hazards to nearby communities in California.

Many of the constituents used in and emitted by oil and gas development can damage health, and place disproportionate risks on sensitive populations, including children, pregnant women, the elderly, and those with pre-existing respiratory and cardiovascular conditions. Health risks near oil and gas wells may be independent of whether wells in production have undergone hydraulic fracturing or not. Consequently, a full understanding of health risks caused by proximity to production wells will require studying all types of productions wells, not just those that have undergone hydraulic fracturing. Oil and gas development poses more elevated health risks when conducted in areas of high population density, such as the Los Angeles Basin, because it results in larger population exposures to toxic air contaminants.

California has large developed oil reserves located in densely populated areas. For example, the Los Angeles Basin reservoirs, which have the highest concentrations of oil in the world, exist within the global megacity of Los Angeles. Approximately half a million people live, and large numbers of schools, elderly facilities, and daycare facilities exist, within one mile of a stimulated well, and many more live near oil and gas development of all types (Figure 1.3-12). The closer citizens are to these industrial facilities, the higher their potential exposure to toxic air emissions and higher risk of associated health effects. Production enabled by well stimulation accounts for a fraction of these emissions.

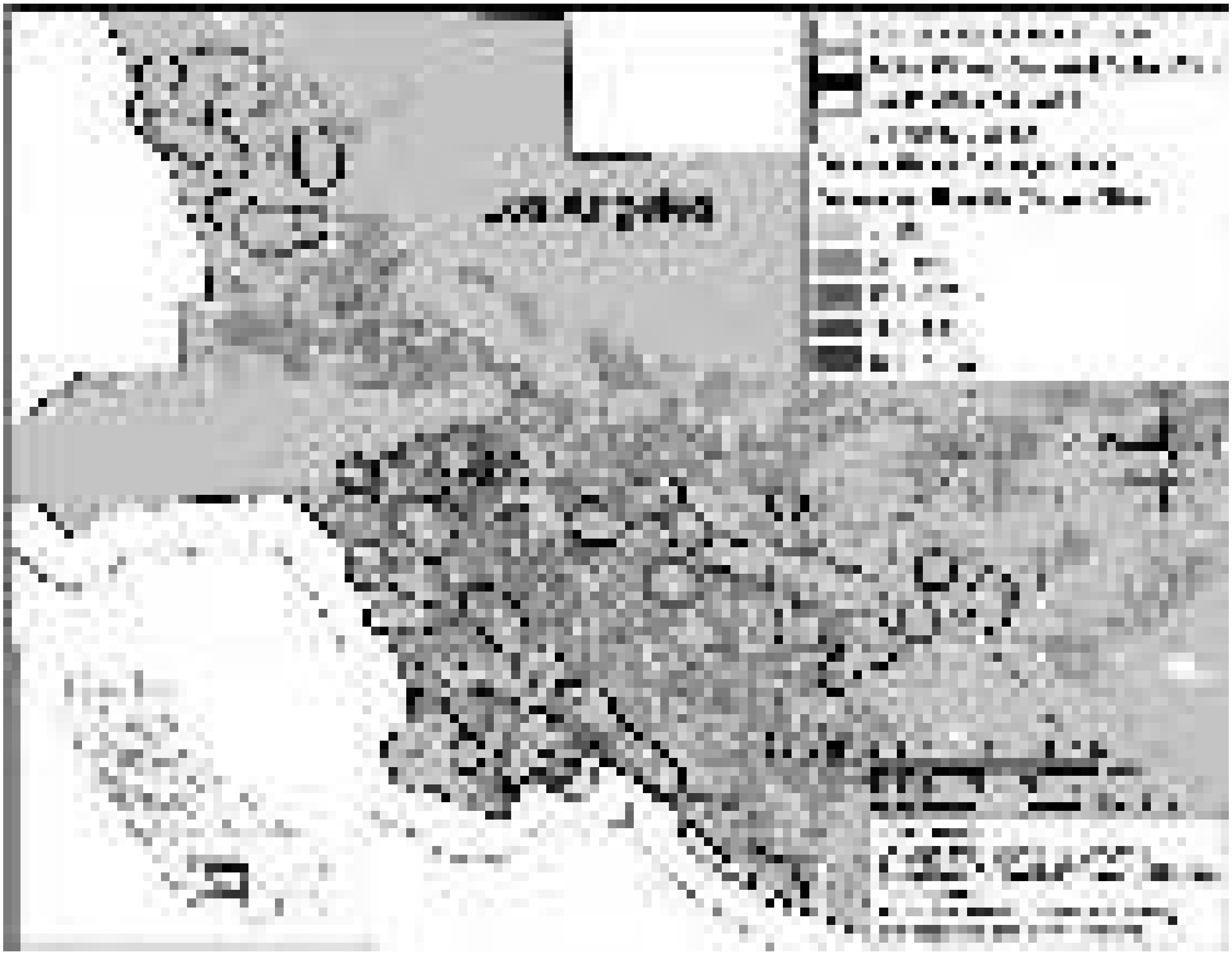

Figure 1.3-12. Population density within 2,000 m (6,562 ft) of currently active oil production wells and currently active wells that have been stimulated (figure from Volume III, Chapter 4 [Los Angeles Basin Case Study]).

Studies from outside of California indicate that, from a public health perspective, the most significant exposures to toxic air contaminants such as benzene, aliphatic hydrocarbons and hydrogen sulfide occur within 800 m (one-half mile) from active oil and gas development. These risks depend on local conditions and the type of petroleum being produced. California impacts may be significantly different, but have not been measured.

Recommendation 6.3. Assess public health near oil and gas production.

Conduct studies in California to assess public health as a function of proximity to all oil and gas development, not just stimulated wells, and develop policies such as science-based surface setbacks, to limit exposures (Volume II, Chapter 6; Volume III, Chapters 4 and 5 [Los Angeles Basin and San Joaquin Basin Case Studies]).

Conclusion 6.4. Hydraulic fracturing and acid stimulation operations add some occupational hazards to an already hazardous industry.

Studies done outside of California found workers in hydraulic fracturing operations were exposed to respirable silica and VOCs, especially benzene, above recommended occupational levels. The oil and gas industry commonly uses acid along with other toxic substances for both routine maintenance and well stimulation. Well-established procedures exist for safe handling of dangerous acids.

Occupational hazards for workers who are involved in oil and gas operations include exposure to chemical and physical hazards, some of which are specific to well stimulation activities and many of which are general to the industry. Our review identified studies confirming occupational hazards directly related to well stimulation in states outside of California. The National Institute for Occupational Safety and Health (NIOSH) has conducted two peer-reviewed studies of occupational exposures attributable to hydraulic fracturing across multiple states (not including California) and times of year. One of the studies found that respirable silica (silica sand is used as a proppant to hold open fractures formed in hydraulic fracturing) was in concentrations well in excess of occupational health and safety standards (in this case permissible exposure limits or PELs) by factors of as much as ten. Exposures exceeded PELs even when workers reported use of personal protective equipment. The second study found exposure to VOCs, especially benzene, above recommended occupational levels. The NIOSH studies are relevant for identifying hazards that could be significant for California workers, but no study to date has addressed occupational hazards associated with hydraulic fracturing and other forms of well stimulation in California.

While both hydrochloric acid and hydrofluoric acid are highly corrosive, hydrofluoric acid can be a greater health risk than hydrochloric acid in some exposure pathways because of its higher rate of absorption. State and federal agencies regulate spills of acids and other hazardous chemicals, and existing industry standards dictate safety protocols for handling acids. The Office of Emergency Services (OES) reported nine spills of acid that can be attributed to oil and gas development between January 2009 and December 2014. Reports also indicate that the spills did not involve any injuries or deaths. These acid spill reports represent less than 1% of all reported spills of any kind attributed to the oil and gas development sector in the same period, and suggest that spills of acid associated with oil and gas development are infrequent, and industry protocols for handling acids protect workers.

Employers in the oil and gas industry must comply with existing California occupational safety and health regulations, and follow best practices to reduce and eliminate illness and injury risk to their employees. Employers can and often do implement comprehensive worker protection programs that substantially reduce worker exposure and likelihood of illness and injury. However, the effectiveness of these programs in California has not been evaluated. Engineering controls that reduce emissions could protect workers involved in well stimulation operations from chemical exposures and potentially reduce the likelihood of chemical exposure to the surrounding community.

Recommendation 6.4. Assess occupational health hazards from proppant use and emission of volatile organic compounds.

Conduct California-based studies focused on silica and volatile organic compounds exposures to workers engaged in hydraulic-fracturing-enabled oil and gas development processes based on the NIOSH occupational health findings and protocols.

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Chapter Two

Impacts of Well Stimulation on Water Resources

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2.1. Abstract

We have analyzed the hazards and potential impacts of well stimulation on California's water resources. Our analysis addresses: (1) the characteristics of water use for well stimulation; (2) the volumes, chemical compositions, and potential hazards of stimulation fluids; (3) the characteristics of wastewater production and management; (4) the potential release mechanisms and transport pathways by which well stimulation chemicals enter the water environment; and (5) practices to mitigate or avoid impacts to water.

Available records indicate that well stimulation in California uses an estimated 850,000 to 1.2 million m^3 (690 to 980 acre-feet) of water per year, the majority of which (91%) is freshwater. Hydraulic fracturing has allowed oil and gas production from some new pools where it was not otherwise feasible or economical. We estimate that freshwater use for enhanced oil recovery in fields where production is *enabled* by stimulation was 2 million to 14 million m^3 (1,600 to 13,000 acre-feet) in 2013. (Well stimulation includes hydraulic fracturing, matrix acidizing, and acid fracturing; enhanced oil recovery includes water flooding, steam flooding, and cyclic steaming, described briefly in Section 2.3 below.) Local impacts of water usage appear thus far to be minimal, with well stimulation accounting for less than 0.2% percent of total annual freshwater use within each of the state's Water Resources Planning Areas, which range in size from 830 to 19,400 km^2 (320 to 7,500 mi^2). However, well stimulation is concentrated in water-scarce areas of the state, and an increase in water use or drawdown of local aquifers could cause competition with agricultural, municipal, or domestic water users.

Over 300 unique chemicals were identified as being used in hydraulic fracturing fluids in California. Of the chemicals voluntarily reported as used for hydraulic fracturing in California, over 200 were identified by their unique Chemical Abstracts Service Registry Number (CASRN). Chemical additives reported without a CASRN cannot be fully evaluated for hazard, risk, and environmental impacts due to lack of specific identification. Many of the chemicals reported for use in hydraulic fracturing are also used for other purposes during oil and gas development, including matrix acidizing. In an analysis of acid treatments, including both routine cleaning and matrix acidizing applications, over 70 chemicals were identified as being used in conjunction with acid, of which over 20 were not reported as used in hydraulic fracturing treatments.

Many of the chemicals used in California do not have the basic suite of physical, chemical, and biological analysis required to establish the chemicals' environmental and health profiles. For example, approximately one-half of chemicals used do not have publicly available results from standard aquatic toxicity tests. More than one-half are missing biodegradability, water-octanol partitioning analysis, or other characteristic measurements that are needed for understanding hazards and risks associated with chemicals.

Wastewater generated from stimulated wells in California includes "recovered fluids" (flowback fluids collected into tanks following stimulation, but before the start of production) and "produced water" (water extracted with oil and gas during production). Some information is known about the volumes of recovered fluids and produced water in California. Data from the Division of Oil, Gas, and Geothermal Resources (DOGGR) indicate that there is no substantive difference between the volume of produced water generated from stimulated wells and non-stimulated wells. Recent data submitted to DOGGR by operators show that the volume of recovered fluids collected after stimulation are a small fraction of the injected fluid volumes (<5%) for hydraulic fracturing treatments, but are higher (~50–60%) for matrix acidizing treatments. The data also show that the recovered fluids are a very small fraction of the produced water generated in the first month of operation. These results indicate that some fraction of returning stimulation fluids is present in the produced water from wells that have been hydraulically fractured.

Little is known about the chemical composition of wastewater from stimulated wells and unconventional oil and gas development. Under new regulations, chemical measurements are being made on recovered fluids, and results show that recovered fluids can contain high levels of some contaminants, including total carbohydrates (indicating the presence of guar) and total dissolved solids (TDS). Some data are available on produced water chemistry from conventional wells in California, but there were no data on the composition of produced waters from stimulated wells available during this study. Lack of understanding of the chemistry of produced water from stimulated wells is identified as a significant data gap.

The recovered fluids are typically stored in tanks at the well site prior to injection into Class II disposal wells. In California, produced water is typically managed via pipelines and disposed or reused in a variety of ways. From January 2011 to June 2014, reports indicate nearly 60% of produced water from stimulated wells was disposed of by infiltration and evaporation using unlined pits. About one-quarter of the produced water from stimulated wells, or about 326,000 m^3 (264 acre-feet), was injected into Class II wells for disposal or enhanced recovery. The disposition method for 17% of the produced water from stimulated wells is either not known or not reported. We note that operators have suggested that the data submitted to DOGGR may not reflect current operating practice due to mistakes in reporting to that agency. Although limited data are available on current treatment and reuse practices in California, it is probable that standard practice for oil-water separation and treatment prior to reuse are unlikely to remove most well stimulation chemicals or their byproducts that may be found in produced water.

Several plausible mechanisms and pathways associated with well stimulation can lead to release of contaminants into surface and groundwater. The release mechanisms of highest priority result from operations that are part of historically accepted practices in the California oil and gas industry, such as disposal of produced water in unlined pits, injection of produced water into potentially protected groundwater, reuse of produced water for irrigation, and disposal of produced water into sewer systems. The concerns related to produced water are relevant to well stimulation because (1) produced water from stimulated wells can contain returned stimulation fluids, and (2) the quality of formation water from stimulated reservoirs could differ from that of conventional reservoirs, and the extent to which they differ is currently unknown. Other concerns of medium priority are accidental releases, some of which need to be better studied. These include the possibility of fractures to serve as leakage pathways (since fracturing depths are much shallower in California than in other parts of the country), leakage through degraded inactive or active wells, and accidents leading to spills or leaks. Finally, there are other releases of low priority, such as operator error and illegal discharges that can be controlled with proper training, oversight, and monitoring.

A few sampling studies have been conducted to assess the impact of hydraulic fracturing on water quality. Only one sampling study has been conducted near a hydraulic fracturing site in California (in Inglewood), but incidents of potential contamination from other regions, such as Pennsylvania (Marcellus formation) and Texas (Barnett, Eagle Ford), can be used to determine potential release mechanisms and hazards, and provide considerations for future monitoring programs in California. While some of the sampling studies indicate that there has been water contamination associated with, and allegedly caused by, well stimulation, other studies did not find detectable impacts due to stimulation. Notably, most groundwater sampling studies do not even measure stimulation chemicals, partly because their full chemical composition and reaction products were unknown prior to this study. In general, groundwater contamination events are more difficult to detect than surface releases, because the effects and release pathways are not visible in the short-term, baseline water quality data are frequently absent, and sufficient monitoring has not been done to confirm the presence or absence of well-stimulation-induced contamination.

2.2. Introduction

Oil and gas development uses water resources and generates wastewater that must be managed by reuse or disposal. There is public concern that well stimulation technologies, especially hydraulic fracturing, may significantly increase water use by the oil and gas industry in California. There is further concern that handling, treatment, or disposal of stimulation fluids may contaminate water resources.

The water cycle of well stimulation consists of five stages (Figure 2.2-1):

1. acquisition of water needed for the stimulation fluids;

2. onsite mixing of chemicals to prepare the stimulation fluids;

3. injection of fluids into a target oil or gas formation during stimulation;

4. recovery of wastewater (flowback and produced water) following stimulation; and

5. treatment and reuse or disposal of wastewaters (after U.S. EPA, 2012a).

Figure 2.2-1. Five stages of the hydraulic fracturing water cycle (U.S. EPA, 2012a).

In this chapter, we describe and evaluate the hazards posed by well stimulation on California's water resources. Our analysis addresses the following questions:

- What are the volumes of freshwater used for well stimulation in California, and what are the sources of these supplies (e.g., domestic water supplies, private groundwater wells, irrigation sources)? How does water use for well stimulation compare with other uses in California and in the regions where well stimulation is occurring?

- What chemicals are being used for well stimulation in California? How often and in what amounts are these chemicals used? What are the physical, chemical, and toxicological properties of the stimulation chemicals used? To what extent does this chemical use create a hazard for and potential impacts on water resources in California?

- What volumes of recovered fluids and produced water are generated from stimulated wells, and what are the chemical compositions of those waters? Are volumes and chemical compositions of produced water generated from stimulated wells and non-stimulated wells different? How are recovered fluids and produced water managed (e.g., disposal by deep well injection or unlined pits)? Would existing treatment technologies for produced water remove well stimulation chemicals that are being used in California?

- What are the release mechanisms and transport pathways related to well stimulation activities that can potentially contaminate surface and groundwater resources in California? Is there evidence of how these releases can impact both surface and groundwater sources? What is the current state of knowledge about groundwater resources in California, particularly in areas where potential releases can occur?

- What are the best practices and measures that would avoid or mitigate impacts to water?

Our sources of information for addressing these questions consist of publicly accessible data, government reports, industry literature, patents, and peer-reviewed scientific literature. To the extent possible, we use data and information specific to California, which originate from several sources. Data sources for chemical and water use information include the FracFocus Chemical Disclosure Registry (www.FracFocus.org) that was available for early 2011 through mid-year 2014, and documentation required from operators under Senate Bill 4 (SB 4), available as of January 1, 2014, which includes *Well Stimulation Notices* (reporting on planned well stimulation activities) and *Well Stimulation Treatment Disclosure Reports* (reporting after stimulation is complete) (DOGGR, 2014a). We obtained information from the South Coast Air Quality Management District (SCAQMD) on water and chemical use during acid treatments that occurred within their jurisdiction between June 2013 and June 2014 (SCAQMD 2013; SCAQMD 2014). Data on the location of oil and gas wells in California, both stimulated and non-stimulated, is compiled and distributed by DOGGR as a "shapefile," or geographic data file (DOGGR, 2014b). Additionally, the Central Valley Regional Water Quality Control Board (CVRWQCB) provided data on the disposal practices associated with unconventional oil and gas development (CVRWQCB, 2014; CVRWQCB, 2015). Data on produced water quantity—from both stimulated and non-stimulated wells—were obtained from the Monthly Production and Injection Database maintained by the California Division of Oil, Gas, and Geothermal Resources (DOGGR, 2014c).

In Section 2.3, we summarize the quantities and sources of water currently being used in California for well stimulation. The information on water use data is presented within the context of regional water use and within the context of other oil and gas production activities. Next, in Section 2.4, we describe the type and amount of chemicals being used in stimulation fluids in California. We discuss what is known about hazards associated with well stimulation chemicals, including the physical, chemical, and toxicological properties of the well stimulation chemicals that are used to evaluate risks associated with chemical use. In Section 2.5, we present analyses on the characteristics of wastewater from unconventional oil and gas development in California, including wastewater volumes and composition, as well as their disposal and beneficial reuse practices. In Section 2.6, we describe the release mechanisms and transport pathways relevant to well stimulation activities in California that can potentially lead to contamination of surface and groundwater resources—occurring through spills, surface and subsurface leaks, and current disposal and reuse practices. In Section 2.7, we discuss the potential impacts that the releases can have on surface and groundwater quality by (1) examining incidents (or the lack thereof) of contamination that have been reported in California and other states, and (2) assessing the current state of knowledge about groundwater in California, particularly in areas that may be impacted by well stimulation activities. We then discuss alternative practices that could potentially mitigate hazards induced by well stimulation in Section 2.8. In Section 2.9, we describe several data gaps that were identified through our analyses. We highlight our major findings in Section 2.10 and present conclusions in Section 2.11.

2.3. Water Use for Well Stimulation in California

2.3.1. Current Water Use for Well Stimulation

In this section, we estimate the volume of water currently used for well stimulation in California. Our estimate is based on (1) the average water-use intensity of well stimulation, i.e., the volume of water used per stimulation operation, and (2) the average number of well stimulations occurring in the state each month. We estimated the water-use intensity for each of the three stimulation methods under consideration (hydraulic fracturing, acid fracturing, and matrix acidization) by analyzing records of stimulation fluid volume reported by operators to state regulators and to the website

FracFocus from January 2011 to June 2014.[1] We estimated the number of well stimulation operations occurring each month from a search of oil and gas well records maintained by the California Department of Conservation's Division of Oil, Gas, and Geothermal Resources (DOGGR 2014). In terms of the number of wells that have been hydraulically fractured, we found that over the last decade, operators fractured about 40%–60% of the approximately 300 wells installed per month in California, leading us to estimate that 125 to 175 wells per month are hydraulically fractured in the state. Additional detail on how these quantities were estimated and the associated data sources is provided in Volume I, Chapter 3, *Historical and Current Application of Well Stimulation Technology in California*. Note that limited data were available for certain types of stimulation operations, such as for offshore operations and acid fracturing.

Figure 2.3-1 shows the range of reported water intensity of well stimulation (or the water volume used per stimulation operation) in California by stimulation method and well type. Table 2.3-1 reports our estimated number of well stimulations occurring each month in California and the *average* or *mean* water use intensity of these operations. Based on these data, we estimate that well stimulation in California uses 850,000 to 1,200,000 m^3 (690–980 acre-feet) of water per year. We report a range of estimated water use to represent the uncertainty in the number of operations that are currently taking place. Operators use some water *directly* for well stimulation; chemicals are added to this "base fluid" and injected during stimulation operations. In addition, the availability of hydraulic fracturing has opened up some new areas to oil production, contributing to ongoing water uses for enhanced oil recovery. An analysis of production *enabled* by stimulation is presented below in Section 2.3.3.

1. No single source contained complete information on well stimulations in California prior to 2014, when reporting became mandatory under new regulations required by SB 4. Data sources included the FracFocus website, DOGGR All Wells shapefile, DOGGR Well Stimulation Notices, DOGGR Completion Reports, Central Valley Regional Water Quality Control Board, and the South Coast Air Quality Management District.

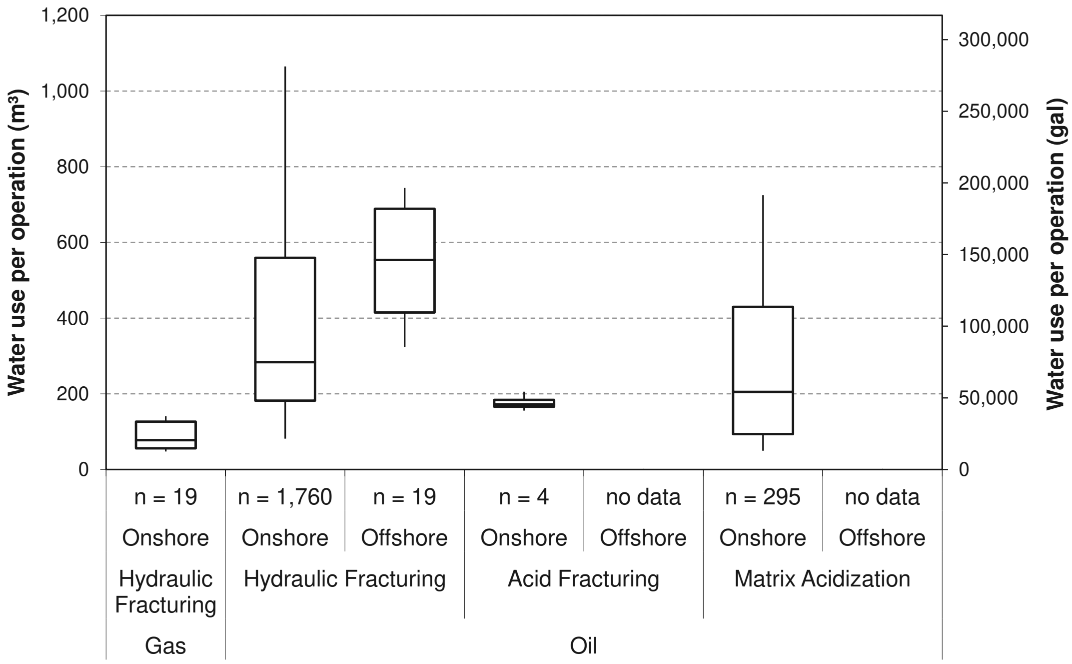


Figure 2.3-1. Boxplots showing range of reported water use per well for well stimulation in California (Jan 2011–Jun 2014) by well type and stimulation type. Box shows the 25th to 75th percentiles of the data. Central line shows the median. Whiskers extend to the 10th and 90th percentiles. Outliers are not shown (Data sources: FracFocus, 2014; DOGGR, 2014a; SCAQMD, 2014; CVRWQCB, 2014).

Table 2.3-1. Estimated volume of water use for oil and gas well stimulation operations in California under current conditions. Number of operations per month estimated for 2004–2014, and average water intensity estimated for Jan 2011 – June 2014.

	Number of operations per month	**Average Water Intensity per well (m^3 $operation^{-1}$)**	**Estimated Annual Water Use (m^3)**	**Annual Water Use (acre-feet $year^{-1}$)**
Hydraulic fracturing	125–175	530	800,000–1,100,000	640–900
Matrix acidizing	15–25	300	54,000–90,000	44–73
Acid fracturing	0–1	170	0–2,000	0–2
Total			**850,000–1,200,000**	**690–980**

Note: We report a range for estimated annual water use to reflect the uncertainty in the number of operations that are currently occurring. As described in Volume I (pages 104-105), we do not know the exact number of stimulation operations that occurred before 2014 because reporting was not mandatory. Our estimate of annual water use was found by multiplying the estimated number of stimulation operations occurring per year in California by the average water-use intensity per operation.

It is worth noting that water use reported to the state by operators for the first 11 months of 2014 (DOGGR, 2014a) was 171,000 m^3 (140 acre-feet), significantly lower than our estimate of the typical annual water use for well stimulation of 850,000 to 1,200,000 m^3 (690 to 980 acre-feet) per year, which was based on data from January 2011 to June 2014 obtained from multiple sources. This discrepancy appears to be due to a slowdown in the number of stimulation operations in 2014 compared to the three previous years. During 2014, there was an average of 44 stimulation operations each month, down from an estimated 140 to 200 operations per month during the years from 2011 through 2013. There could be several causes for this slowdown, including uncertainty among operators related to new regulations, public pressure, or dropping oil prices in the second half of 2014. The average water use per stimulation operation reported by operators in 2014 also appears to be somewhat lower than the historical rates of water use. Operators used an average of 390 m^3 (0.32 acre-feet) for hydraulic fracturing operations in 2014, lower than the average water use of 530 m^3 (0.43 acre-feet) during the previous three years.

2.3.2. Water Sources

We investigated where operators are acquiring water for well stimulation by analyzing data from well stimulation completion reports. Under new SB 4 regulations effective January 1, 2014, operators are required to send DOGGR a *Well Stimulation Treatment Disclosure Report*, referred to here as a "completion report," within 60 days after completing stimulation. On this form, operators identify the source of the water they used as a base fluid for stimulation. They also identify the *type* of water that makes up the base fluid, i.e., "water suitable for irrigation or domestic purposes," "water **not** suitable for irrigation or domestic purposes," or "fluid other than water."

There were 495 completion reports filed by operators and published by DOGGR between January 1 and December 10, 2014 (DOGGR, 2014a). Among these reports, there were 15 where the operator reported the volume of water use as zero, which we believe to be an error. We removed these records, and analyzed the remaining 480 reported stimulations. A summary of reported water use by source is shown in Table 2.3-2.

Operators obtained the water needed for well stimulation from nearby irrigation districts (68%), produced water (13%), operators' own wells (13%), a nearby municipal water supplier (4%), or a private landowner (1%). About a tenth of the total water volume was identified as water **not** suitable for irrigation or domestic use. Why the water was deemed unsuitable was not specified, but it is presumed that the water had high salt content. In California, freshwater is defined as having a TDS content less than 3,000 mg L^{-1} (see Section 2.7).

Table 2.3-2. Water sources for well stimulation according to 480 well stimulation completion reports filed from January 1, 2014 to December 10, 2014.

Water Source	Number of Operations	Total Water Volume m³	Total Water Volume acre-feet	Percent of Total Water Volume
Irrigation district	399	117,000	95	68%
Produced water	43	23,000	18	13%
Own well	28	22,000	18	13%
Municipal water supplier	9	7,000	6	4%
Private landowner	1	2,000	2	1%
Total	**480**	**171,000**	**140**	**100%**

Of the 495 completion reports filed, all but two were for operations in Kern County. Many of the Kern County operations (397, or 83%) used water from the Belridge Water Storage District, which was formed to serve farmers in central Kern County with water provided by the State Water Project. The two submitted completion reports from outside of Kern County were in Ventura County and conducted by Aera Energy. These operations both used water from the Casitas Municipal Water Supply District, which provides water to about 70,000 people and several hundred farms in western Ventura County.

2.3.3. Water Use for Enhanced Oil Recovery

In this section, we analyze water use related to enhanced oil recovery. This analysis serves two purposes: first, to understand how the freshwater demand for well stimulation compares to freshwater demand for enhanced oil recovery; and second, to estimate the additional freshwater demand that occurs when stimulation technology allows production from new zones to be developed. The application of well stimulation technology has enabled production in some new pools where it would not have been likely to occur otherwise. The development of these pools creates additional demands for water, particularly for enhanced oil recovery. This water demand can be considered additional to the water that is used directly as the base fluid for well stimulation operations such as hydraulic fracturing. Below, we examine the water use for what we refer to as production *enabled* by well stimulation.

Water is used for a number of different purposes throughout the oil and gas production process, including drilling, well completion (during which well stimulation occurs), well cleanout, and for some types of enhanced oil recovery (EOR). Initially, oil production consists of simply producing oil and gas from the reservoir (primary production). In California, production in most reservoirs has been occurring for a span of time ranging from several decades to more than a century, so primary production has ended. Continued production requires additional processes including water flooding (secondary recovery) or, in California, steam flooding or cyclic steaming (two of many types of tertiary recovery). Water flooding and steam flooding involve continuous injection to push oil toward production wells, and, in the case of steam, to also reduce the oil's viscosity along with other effects. Cyclic steam injection involves periodic injection of steam followed by a well shut-in period to allow the heat to reduce the oil viscosity, followed by a period of production, after which the cycle repeats.

We obtained information about the location and volume of water used for enhanced oil recovery from DOGGR's Production/Injection Database (DOGGR, 2014c). According to this data, there were 29,061 wells that injected water or steam into oil and gas reservoirs in 2013. DOGGR's database also contained information on the *type* and *source* of water injected.

We performed a series of analysis to determine the volume, type, and source of water used for EOR in California. These results are reported in Table 2.3-3. We found that in 2013, the total volume of water (or water converted to steam) injected by operators totaled 443 million m^3 (360,000 acre-feet).

In terms of water *source*, operators reported that two-thirds of the water injected (288 million m^3 or 233,000 acre-feet) was produced water, or water that is pumped to the surface along with oil and gas, and subsequently re-injected back into the formation, largely forming a closed loop system. Operators using solely produced water for injection are not generally competing with other water users. Approximately one-third of injected water was not produced water, which means operators obtained this water from another source. We refer to this water here as *externally sourced* water. Another 23% of injected water was externally sourced salt water; this includes saline groundwater (94 million m^3, or 76,000 acre-feet) and ocean water (7 million m^3, or 5,000 acre-feet).

In addition to produced water, however, operators are also injecting externally sourced freshwater for enhanced oil recovery. In 2013, operators reported 3% of injected water as "freshwater" (15 million m^3 or 12,000 acre-feet). However, we estimated freshwater use may be as high as 14%, based on ambiguity in the reporting categories in DOGGR's database. DOGGR's database allows operators to report water type in one of five categories; one of these is labeled "freshwater," but some of the other categories may be composed partly or entirely of freshwater. These ambiguous categories include "water combined with chemicals such as polymers," "another kind of water," and "not reported." By combining these categories with the freshwater category, we estimate injected freshwater in 2013 may have been as high as 60 million m^3 (49,000 acre-feet).

In order to understand *where* operators are obtaining freshwater for EOR, we performed another set of queries and analyses using DOGGR's Production/Injection database. In 2013, operators reported that they obtained freshwater for injection from several sources: domestic water systems (72%), water source wells (25%), wastewater from an industrial facility (1.6%), and not reported (1.4%) or reported as "another source or combination of the above sources" (0.1%).

Table 2.3-3. Breakdown of injected water for enhanced oil recovery by source and type of water, in million m^3 per year, in 2013. This does not include water for well stimulation.

All sources:	**million m^3**	
Produced from an oil or gas well	288	65%
Ocean	<0.001	<0.1%
Other Sources	155	35%
Total	**443**	

Breakdown of water type in "Other sources" above:		**million m^3**	
Salt water	94	61%	
Water combined with chemicals such as polymers	22	14%	
Not Reported	17	11%	
Another kind of water	6	3.9%	
Freshwater		15	9.8%
Total other sources		**155**	

Source of freshwater listed above:	**million m^3**	
Domestic water systems	11	72%
Produced from a water source well	4	25%
Wastewater from an industrial facility	0.2	1.6%
Not reported	0.2	1.4%
Another source or combination of the above sources	0.015	0.1%
Total all externally sourced freshwater	**15**	

Note: Table figures may not add due to rounding.

We analyzed how much freshwater is used for EOR in fields where production is *enabled* by well stimulation technology. To do this, we summarized freshwater use for EOR in pools that we had previously categorized as having production enabled by well stimulation. These are typically formations with low transmissivity where oil or gas production is not economically feasible without fracturing. We identified these pools by analyzing well records maintained by DOGGR, and identified 68 pools where the majority of new production wells from 2002 to 2013 were hydraulically fractured (see Volume I for detailed analysis. We estimate that water use for EOR in these pools ranged from 2 million to 14 million m^3 (1,600 to 13,000 acre-feet) in 2013, while freshwater use for EOR in all other oil and gas fields was 13 million to 44 million m^3 (11,000 to 36,000 acre-feet) in 2013, as shown in Figure 2.3-2. Thus, we may conclude that between 15% and 30% of freshwater use for EOR in California in 2013 can be attributed indirectly to the application of well stimulation.

We also compared the total volume of freshwater that oil and gas operators use for well stimulation to the volume used for enhanced oil recovery. Based on our estimates above, operators used from 2 to 15 times more freshwater for EOR than they used for well stimulation in 2013. Figure 2.3-2 compares the estimated volume of water used for well stimulation with the volume of water injected for EOR in 2013.

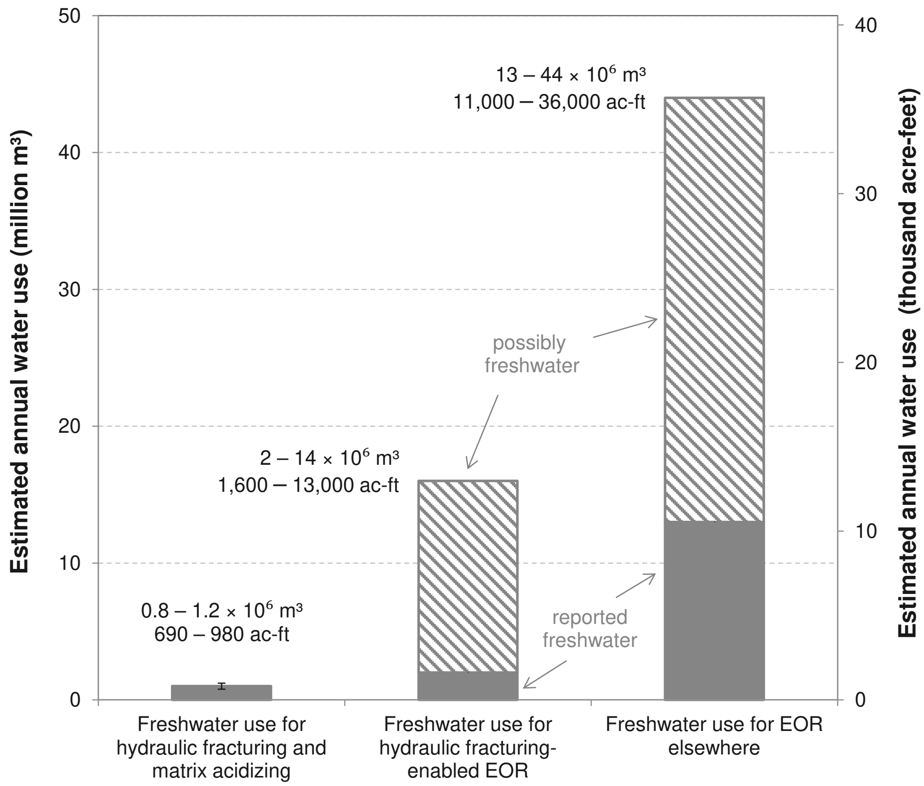


Figure 2.3-2. Estimated annual freshwater use for well stimulation (left), enhanced oil recovery (EOR) in 2013 in reservoir where most wells are hydraulically fractured (middle), and EOR in 2013 in other reservoirs (right). Well stimulation including hydraulic fracturing occurs before the well goes into production. EOR occurs throughout production.

Note: The solid bar in this figure represents water volume explicitly classified as freshwater in the DOGGR Production Database. The hatched area represents water used for EOR that is reported as a type that may be all or part freshwater. When we include "water combined with chemicals such as polymers," "another kind of water," and blank records (unknown water type), freshwater use for enhanced oil recovery may be as high as 60 million m³ (49,000 acre-feet).

2.3.4. Water Use for Well Stimulation in a Local Context

Water use for well stimulation and stimulation-enabled EOR in California is small in the context of the state's total water use; our estimate of this water use for well stimulation is less than 2 million m^3 (<2,000 acre-feet) per year (Table 2.3-1 and Figure 2.3-2), while human water use statewide averages about 56 billion m^3 (45 million acre-feet) per year (DWR, 2014a). Water concerns, however, are local, and the impacts of that water use should be evaluated within a local context. Where oil and gas extraction occurs alongside other uses, it can mean competition over a limited resource, especially where the oil and gas industry is usually willing and able to pay more for water than irrigators or other water users (Freyman, 2014; Healy, 2012).

To get a better sense of water use in regions where well stimulation has been reported, we examined water use within Planning Areas, also referred to as "PAs". PAs are geographic units created by the California Department of Water Resources (DWR) for the planning and management of the state's water resources. DWR divides the state into 56 PAs, ranging in size from 830 to 19,400 km^2 (320 to 7,500 mi^2), with an average size of 6,700 km^2 (2,600 mi^2). PA boundaries typically follow watershed boundaries, but are sometimes coincident with county boundaries or hydrologic features, such as rivers and streams.

From January 2011 to the end of May 2014, well stimulation was documented in 19 of the state's 56 PAs (Table 2.3-4). We estimated the amount of water used for well stimulation and hydraulic-fracturing-enabled EOR by PA and compared that water use to total water use for the area (Table 2.3-4).

Table 2.3-4. Estimated annual water use for well stimulation and hydraulic fracturing-enabled EOR by water resources Planning Area.

Planning Area	For well stimulation operations (m^3)	For enabled EOR (m^3)	Total water use (stimulation + EOR, m^3)	% of water use in Planning Area
Santa Ana	1,300		1,300	0.000082%
Metro Los Angeles	25,000		25,000	0.0013%
Santa Clara	11,000		11,000	0.0018%
Central Coast Southern	270		270	0.000043%
Semitropic	930,000	2,000,000*	2,900,000	0.19%
Kern Delta	2,100		2,100	0.00011%
Kern Valley Floor	18,000		18,000	0.0016%
Uplands	9,300		9300	0.015%
Central Coast Northern	900		900	0.00011%
Western Uplands	2,900		2,900	0.10%
San Luis West Side	260		260	0.000017%
Lower Kings-Tulare	750		750	0.000031%
North Bay	930		930	0.00035%
San Joaquin Delta	440		440	0.000038%
Sacramento River Delta	1,300		1,300	0.00018%
Central Basin, West	480		480	0.000044%
Colusa Basin	2,900		2,900	0.00011%
Butte-Sutter-Yuba	3,100		3,100	0.000098%
Offshore	6,600		6,600	n/a
Total	**1,000,000**	**2,000,000**	**3,000,000**	**0.0057%**

**In this table, we report the low estimate for water use for EOR in fields where production is enabled by well stimulation. In Section 2.3.3, we found that this water use may range from 2 million to 14 million m^3 (1,600 to 13,000 acre-feet).*

Note: Water use estimates for Planning Areas are for the year 2010 (from DWR, 2014b). Numbers may not sum to the total values due to rounding.

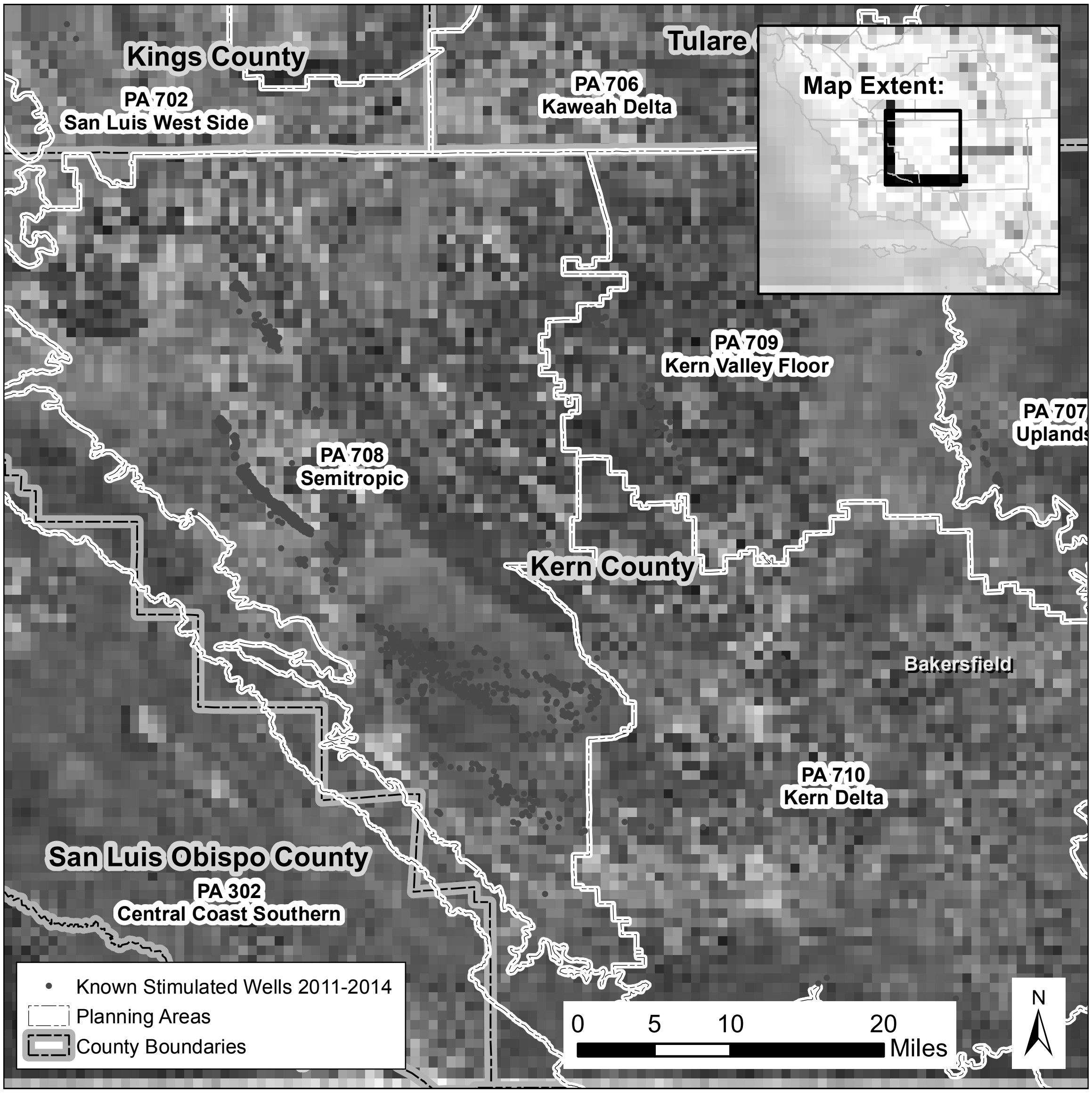


Figure 2.3-3. Map showing oil and gas wells stimulated from January 2011 through June 2014 and Water Resources Planning Areas in the Tulare Lake basin.

The majority of well stimulation operations occurred in western Kern County in the Semitropic PA (Figure 2.3-3). All of the reported matrix-acidizing operations are in this PA as well, as is all the freshwater use for EOR enabled by hydraulic fracturing. Water use for well stimulation and hydraulic-fracturing-enabled EOR comprises less than 0.1% of human water use in almost all PAs where stimulation occurs. Water use by PA attributable to well stimulation ranged from a low of 270 m^3 (0.22 acre-feet) in the Central Coast Southern and San Luis West Side PA, to a high of 2,900,000 m^3 (2,400 acre-feet) in the Semitropic PA (Table 2.3-4). Even within the Semitropic PA, where the vast majority of well-stimulation-related freshwater use occurs, water use for well stimulation accounts for only 0.19% of water use (Table 2.3-5). Within this PA, the largest water use is for irrigated agriculture, which used 1,500 million m^3 (1.2 million acre-feet) in 2010. This is followed by energy production and urban use.

Table 2.3-5. Estimated annual water use for well stimulation and hydraulic fracturing-enabled EOR in the Semitropic Planning Area compared to applied water volumes estimated by DWR for 2010. Note water use for hydraulic fracturing-enabled EOR was subtracted from energy production water volume estimated by the DWR.

	million m^3 $year^{-1}$	**acre-feet $year^{-1}$**
Well stimulation and hydraulic fracturing-enabled EOR	2.9	2,400
Estimated Applied Water in 2010*		
Energy Production	19	15,000
Urban (commercial, industrial, residential)	10	8,000
Agricultural	1,500	1,200,000
Total	**1,530**	**1,220,000**

**Numbers may not sum to total due to rounding*

Despite its relatively low freshwater use, concerns have been raised by some water analysts and environmental organizations that freshwater use for hydraulic fracturing could have a negative impact because it is concentrated in relatively water-scarce regions, and the additional demand could strain available supplies (e.g., Summer, 2014; Center for Biological Diversity, 2015). Competition for water could become more critical in the face of extended drought.

Most of the hydraulic fracturing in California takes place in the San Joaquin Valley, where groundwater has been over-drafted by agriculture for over 80 years, causing a host of problems, including subsidence of the land surface. The 8-meter drop in the land surface near Mendota, California, is among the largest ever that has been attributed to groundwater pumping (Galloway et al., 1999). New water demands on top of already high competition for water could further deplete the region's aquifers, as has been observed in other water-scarce regions of the U.S. where hydraulic fracturing is occurring (Reig et al., 2014). This could cause concern for smaller communities and domestic users that rely on local groundwater. In the San Joaquin Valley, farmers and communities also depend on imported water delivered by canals, deliveries of which have become increasingly unreliable in recent years (DWR, 2014a). On the other hand, in some areas, produced water from oil fields that have low salt concentrations can be a source of water, and is being reused for a variety of beneficial purposes, including for irrigation and groundwater recharge, as discussed in Section 2.6.

2.4. Characterization of Well Stimulation Fluids

2.4.1. Understanding Well Stimulation Fluids

Understanding the composition, or formulations, of well stimulation fluids is an important step in defining the upper limits of potential direct environmental impacts from hydraulic fracturing and other well stimulation technologies. The amounts of chemicals added to well stimulation fluid define the maximum possible mass and concentrations of chemical additives that can be released into the environment. The chemicals added to well stimulation fluid might also influence the release of metals, salts and other materials found naturally in oil and gas bearing geological formations. Due to the economic value of individual well-stimulation-fluid formulations and competition between oil field service companies, operators and service companies have been generally reticent about releasing detailed information concerning the types and amounts of chemicals used in specific formulations. Often when information is released, the information may be incomplete (e.g., Konschnik et al., 2013). This lack of transparency has heightened uncertainty and concerns about the chemicals used in well stimulation fluid.

We investigated the composition of well stimulation fluids that are used in California with the objectives of (1) developing an authoritative list of chemicals used for well stimulation in California, (2) determining the concentrations at which the chemicals are used, and (3) estimating the amount (mass) of each chemical that is used per well stimulation. Characteristics of stimulation chemicals, including aquatic and mammalian toxicity were also evaluated (see below and Chapter 6). Chemical disclosures include information on the volume of water used as a "base fluid" and the concentrations of chemicals present in individual well-stimulation-fluid formulations, from which the mass of chemicals used per stimulation can be estimated.

We compiled the reported uses of chemical additives in hydraulic fracturing and acid treatments, and evaluated the information using numerous approaches. A list containing hundreds of chemicals can be initially bewildering, even to experts, and it is helpful to understand the significance of individual chemicals or chemicals in mixtures in the context of their frequency of use, the amounts used, and their hazardous properties, such as toxicity. Other information to help understand and evaluate chemicals includes the purpose of their use, the class of chemical to which they belong, and other distinguishing characteristics, such as vapor pressure and water solubility. Previous studies have evaluated and characterized chemical additives to well stimulation fluids that are in common use nationally (Stringfellow et al., 2014; U.S. House of Representatives Committee on Energy and Commerce, 2011; U.S. EPA, 2012a). In this study, we examine chemicals specifically used in California and develop a comprehensive list of well-stimulation-fluid additives for California.

In this section, chemicals known to have been constituents of well stimulation fluids in California are ranked and characterized for their hazardous properties in relation to aquatic environments. Chapter 6 addresses hazards in the context of human health.

Understanding hazard is important; however, the risk associated with any individual chemical is a function of the release of the material to the environment, how much material is released, the persistence of the compound in the environment, and many other properties and variables that allow a pathway to human or environmental receptors. A full risk assessment is beyond the scope of this study. However, information on hazard, toxicology, and other physical, chemical, and biological properties developed in this section are fundamental to the understanding of environmental and health risk associated with well stimulation treatments in general, and well stimulation fluid specifically.

2.4.2. Methods and Sources of Information

Prior to the enactment of SB 4 authorized regulation in California in January 2014, all information from industry on the composition of well stimulation fluid was released on a voluntary basis. A primary source of data for the analysis in this section was voluntary disclosures reported to the FracFocus Chemical Disclosure Registry (http://fracfocus.org/). The data used in this analysis include disclosures entered into the Chemical Disclosure Registry for hydraulic fracturing in California prior to June 12, 2014. This analysis includes listing all the chemicals used in 1,623 hydraulic fracturing treatments conducted in California between January 30, 2011 and May 19, 2014 (Appendix A, Table 2.A-1). The mass used per treatment and the frequency of use were only calculated using well stimulation treatments that had complete records (Appendix A, Table 2.A-1). A complete treatment record was a record that included the volume of base fluid used, the concentration of the base fluid and the concentration of each chemical used as percent of total treatment fluid mass, and where the sum of the reported masses was between 95% and 105%. Of the 1,623 reported applications, 1,406 (87%) met the criteria for complete records.

The Chemical Disclosure Registry only includes disclosures for hydraulic fracturing treatments and does not include other well stimulation treatments, such as matrix acidizing treatments. Sources of information for acid treatments include Notices of Intent and Completion Reports submitted to DOGGR since December 2013 under new SB 4 regulations and chemical use reported to SCAQMD under reporting regulations in effect since 2013 (SCAQMD, 2013).

There were an estimated 5,000 to 7,000 hydraulic fracturing treatments in California between 2011 and 2014, suggesting that the voluntary disclosure record represents only one-third to one-fifth of the estimated total hydraulic fracturing treatments. However, the disclosures include the major producers and service companies operating in California, including Baker Hughes, Schlumberger, and Halliburton. The chemical additives listed in the voluntary disclosures were consistent with additives described in information available from industry literature, patents, scientific publications, and other sources, such as government reports (e.g., Gadberry et al., 1999; U.S. EPA, 2004; Baker Hughes Inc., 2011; 2013; Stringfellow et al., 2014). Therefore, it is concluded that this list is representative of chemical use for well stimulation in California.

The hazard that a material may present if released to the environment is assessed using a number of criteria, including the toxicity of the chemical to aquatic species selected to represent major trophic levels of aquatic ecosystems. Common standard test species include the fathead minnow (*Pimephales promelas*); various species of trout; daphnia, such as *Daphnia magna*; and various species of green algae (U.S. EPA, 1994; OECD, 2013). The test species represent a basic aquatic food chain of primary producers (algae), grazers (daphnia), and predators (minnows). The species tested are typically selected on the basis of availability, regulatory requirements, and past successful use. Other test species (e.g., trout) may be selected for testing based on commercial, recreational, and ecological importance. Standardized test data for lethality are typically reported as median lethal dose (LD_{50}) for mammals and median lethal concentration (LC_{50}) for fish. In the case of aquatic crustaceans and algae, the effective concentration at which 50% of the test population is adversely affected is determined and reported as the median effective concentration (EC_{50}). Since aquatic toxicity tests are highly standardized, the results can be used to compare and contrast industrial chemicals (Stringfellow et al., 2014). Experimental tests against aquatic species are an important component of an ecotoxicological assessment.

For this study, we examine the acute toxicity of individual chemicals to fathead minnows, daphnia, and algae. Acute toxicity data were collected only for the chemicals used in well stimulation in California that were identified by CASRN. Toxicity data were gathered from publicly available sources as shown in Table 2.4-1. Computational methods (EPI Suite) were applied in an attempt to fill data gaps when chemicals have not been thoroughly tested using experimental methods (Mayo-Bean et al., 2012; U.S. EPA, 2013c). The U.S. EPA cautions that EPI Suite is a screening-level tool and should not be used if acceptable measured values are available (U.S. EPA, 2013c). In this study, we only included EPI Suite results if experimental results were not available. In the case of green algae, insufficient experimental results were found, and only EPI Suite results were used in the analysis. The EPI Suite values for freshwater fish were also used to fill data gaps for both fathead minnow and trout toxicity (Appendix B, Figure 2.B-1).

Ecotoxicity results were interpreted in the context of the Globally Harmonized System (GHS) criteria for the ranking and classification of the acute ecotoxicity data. A similar approach was taken to evaluate mammalian toxicity and is described in Chapter 6. The United Nations Globally Harmonized System (GHS) of Classification and Labeling of Chemicals was used to categorize chemicals based upon their LD_{50}, LC_{50}, or EC_{50} values (Appendix A, Tables 2.A-2 and 2.A-3) (United Nations, 2013). In the GHS system, lower numbers indicate greater toxicity, with a designation of “1” indicating the most toxic compounds (Appendix A, Tables 2.A-2 and 2.A-3). Chemicals for which the LD_{50}, LC_{50}, or EC_{50} exceeded the highest GHS category were classified as non-toxic.

Physical and chemical data for fracturing fluid additives was obtained from online chemical information databases, government reports, chemical reference books, materials safety data sheets, and other sources as previously described (Stringfellow et al., 2014).

Physical and chemical data are mostly based on laboratory tests using pure compounds. Physical, chemical, and toxicological properties were selected for inclusion in this study based on their use in environmental fate and transport studies, treatability evaluations, remediation efforts, and risk assessments (Stringfellow et al., 2014). Chemicals used in well stimulation were categorized as *non-biodegradable* or *biodegradable* using OECD guidelines (OECD, 2013). Biodegradability is useful for determining the effectiveness of biological treatment for wastewaters and the fate of chemicals released into the environment. In the absence of measured biodegradation data, computational methods developed for the U.S. EPA (e.g., BIOWIN) were used to estimate biodegradability (U.S. EPA, 2012b).

Table 2.4-1. Sources for physical, chemical, and toxicological information for chemicals used in well stimulation treatments in California.

U.S. EPA (Environmental Protection Agency), ACToR (Aggregated Computational Toxicology Resource) Database, 2013, http://actor.epa.gov/actor/faces/ACToRHome.jsp
National Library of Medicine, ChemIDplus Advanced. http://chem.sis.nlm.nih.gov/chemidplus/
U.S. EPA (Environmental Protection Agency) and Office of Pesticide Programs, ECOTOX Database Version 4.0, 2013, http://cfpub.epa.gov/ecotox/
European Chemicals Agency (ECHA), International Uniform Chemical Information Database (IUCLID), CD-ROM Year 2000 Edition, 2000.
National Institute of Technology and Evaluation, Chemical Risk Information Platform (CHRIP). http://www.safe.nite.go.jp/english/db.html
R.J. Lewis, N.I. Sax, Sax's dangerous properties of industrial materials, 9th ed., Van Nostrand Reinhold, New York, NY, 1996
Syracuse Research Corporation PhysProp Database
National Library of Medicine, Toxicology Data Network (TOXNET) Hazardous Substance Data Bank (HSDB), 2013, http://toxnet.nlm.nih.gov/cgi-bin/sis/htmlgen?HSDB
SciFinder, Chemical Abstract Service, Colombus, OH, https://scifinder.cas.org
Materials Safety Data Sheets from Sigma-Aldrich, BASF, Spectrum, ExxonMobil, Alfa Aesar, Clariant, and other chemical suppliers
Organization for Economic Cooperation and Development (OECD) - Screening Information Data Set
California Prop 65, Chemicals Known to the State to Cause Cancer or Reproductive Toxicity. http://www.oehha.ca.gov/prop65/prop65_list/files/P65single050214.pdf
International Agency for Research on Cancer (IARC), IARC Monographs on the Evaluation of Carcinogenic Risks to Humans, World Health Organization. http://monographs.iarc.fr/ENG/Classification/ClassificationsCASOrder.pdf
U.S. EPA (Environmental Protection Agency), EPI Suite, Experimental Values
Toxic Substance Control Act Test Submissions 2.0, 2014, http://yosemite.epa.gov/oppts/epatscat8.nsf/ReportSearch?OpenForm

2.4.3. Composition of Well Stimulation Fluids

2.4.3.1. Chemicals Found in Hydraulic Fracturing Fluids

A list of chemical additives reported to have been used in California for hydraulic fracturing treatments is shown in Appendix A, Table 2.A-1. The list includes frequency of use, concentration, and mass of chemicals used for hydraulic fracturing in California, as reported to the FracFocus Chemical Disclosure Registry prior to June 12, 2014. The list contained in Table 2.A-1 includes only the subset of hydraulic fracturing treatment data for which the sum of the reported additives was 100% ± 5%.

As can be seen in Table 2.A-1, not all additives were identified by CASRN, which is a standardized system for the clear and singular identification of chemicals, otherwise known by various common names, trade names, or product names, which may or may not be specific. Of the disclosed chemical additives, there were approximately 230 chemicals or chemical mixtures identified by CASRN; others were identified by name only. Over 100 chemicals could not be positively identified because a CASRN was not provided. After analysis and standardization of chemical names, over 300 chemicals or chemicals mixtures were identified by unique name or CASRN. Since in many cases generic names were used for chemical additives on the disclosures (e.g., surfactant mixture, salt, etc.), any enumeration of the number of chemicals used in hydraulic fracturing should be considered approximate (Table 2.A-1). Many of the additives used in hydraulic fracturing are also used in other routine oil and gas operations, such as well drilling. Other chemicals are specific to well stimulation, such as guar and borate cross linkers.

Disclosures that do not provide CASRN for each entry do not allow definitive identification of the well-stimulation-fluid additive. However, chemical names are generally informative, and each identified substance was investigated and, where possible, referenced to specific products sold by the major suppliers of well stimulation services and chemicals in California. There was a median of 23 individual components—including base fluids, proppants, and chemical additives—used per treatment (Figure 2.4-1). The number of unique components used as reported here differs from a recent study by the U.S. EPA, which reported a median of 19 chemical additives used per treatment in an analysis of 585 disclosures (U.S. EPA, 2015a). The difference between these two studies results in part because of differences in the number of disclosures examined (585 vs. 1,406 for this study), but also because the number here includes base fluids and proppants, while the U.S. EPA study did not include these in developing the median value of 19 (U.S. EPA, 2015a). The disclosures include descriptions for chemicals added for the purpose of stimulation (e.g., water, gelling agents, biocides, etc.) and entries for so-called impurities found in the chemicals used for formulating well-stimulation fluid. In many cases, impurities are reported without concentration data or mass concentrations of <0.001% of the mass of the injected fluid.

Impurities are common in industrial-grade chemicals, which are rarely 100% pure. Impurities are frequently residual feedstock materials from the manufacturing process or solvents and other materials added to control product consistency or handling properties. Table 2.A-1 gives the reported median chemical concentration in well stimulation fluid. Chemicals can be added at hundreds and sometimes thousands of mg kg^{-1} of fluid. Even the impurities, which are not specifically added for a purpose directly related to well stimulation, can occur at high concentrations in well stimulation fluid. For example, magnesium chloride and magnesium nitrate are inactive ingredients (e.g., impurities) found in biocides containing 2-methyl-3(2H)-isothiazolone and 5-chloro-2-methyl-3(2H)-isothiazolone (Miller and Weiler, 1978). Even though impurities are not added specifically for well stimulation, they must be considered during an evaluation of the hazards associated with hydraulic fracturing.

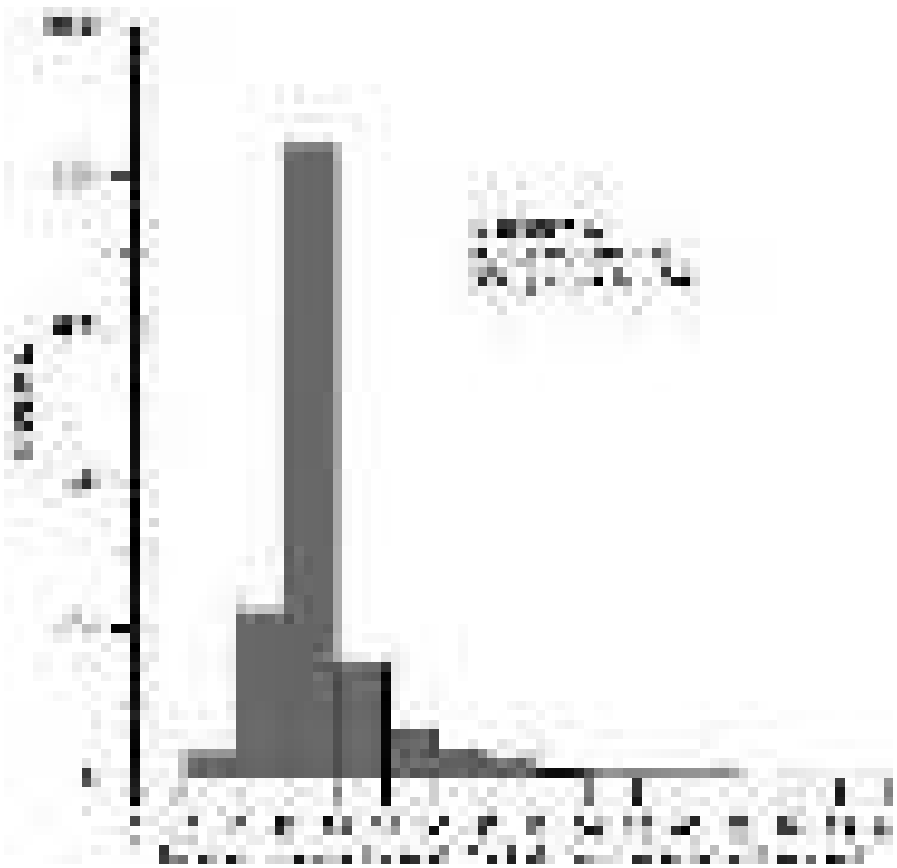


Figure 2.4-1. Frequency distribution of the number of components used per hydraulic fracturing operation in California. Only complete records were included in the analysis where the sum of the treatment components was 100 ± 5% (N=1,406).

2.4.3.2. Chemicals Found in Matrix Acidizing Fluids

There are well stimulation treatments used in California that involve the use of strong acids, including hydrochloric and hydrofluoric acid (see Volume I, Chapter 2 and 3 and California Council on Science and Technology (CCST) et al., 2014). Due to the absence of state-wide mandatory reporting on chemical use in the oil and gas industry, it is not known how much acid is used for oil and gas development throughout California. However, available information suggests that there are approximately twenty matrix acidizing treatments in California per month, but detailed chemical information on specific treatments are not available. Parts of southern California have mandatory reporting on the use of all chemicals used for well drilling, reworks, and well completion activities (http://www.aqmd.gov/). Analysis of these data suggests acid use is widespread and common for many applications in the industry.

As of December 2013, under interim regulations, DOGGR has required operators to submit a "Notice of Intent" for well stimulation treatments, including matrix acidizing. These notices include a list of chemicals that may be used in a planned well stimulation treatment. Analysis of these mandatory Notices of Intent that were publicly available between December 2013 and June 2014 found 70 chemicals identified by CASRN. Seven compounds reported in Notice of Intent documents for matrix acidizing were not found in voluntary notices reported to the Chemical Disclosure Registry for hydraulic fracturing treatments (Table 2.4-2).

Table 2.4-2. Seven compounds submitted to DOGGR in a Notice of Intent to perform matrix acidizing that were publicly available between December 2013 and June 2014 that were not found in voluntary notices reported for hydraulic fracturing to the FracFocus Chemical Disclosure Registry (Table 2.A-1). Notices of Intent are required for all well stimulation treatments as of December 2013 under interim regulations.

Chemical Name	CASRN	Also reported as used in hydraulic fracturing (Table 2.A-1)
Hydroxylamine hydrochloride	5470-11-1	No
Benzaldehyde	100-52-7	No
Cinnamaldehyde	104-55-2	No
Amine oxides, cocoalkyldimethyl	61788-90-7	No
Copper dichloride	7447-39-4	No
Ethylene oxide	75-21-8	No
Sodium iodide	7681-82-5	No

As of January 2014, under SB 4, DOGGR has also required operators to submit a "Well Stimulation Treatment Disclosure Report" within 60 days of completion of well stimulation treatments, including matrix acidizing. These reports include a list of chemicals that were actually used in a well stimulation treatment. Analysis of the disclosure reports available as of May 2015 identified 25 chemical compounds by CASRN used in matrix acidizing that were not found in the voluntary notices reported to the Chemical Disclosure Registry between 2011 and June 2014 (Table 2.4-3). However, of the 25 compounds identified as being used in matrix acidizing, 11 are also reported in the DOGGR disclosure reports as being used for hydraulic fracturing in 2015. Of the seven compounds submitted to DOGGR in the Notices of Intent for matrix acidizing (Table 2.4-2) that were not reported to the Chemical Disclosure Registry, only three were reported in the Well Stimulation Treatment Disclosure Reports. These results indicate that there is overlap in chemical use between matrix acidizing and hydraulic fracturing, and that mandatory reporting will include some chemicals not listed on voluntary disclosures prior to 2014.

Table 2.4-3. Chemicals used for matrix acidizing in California, as reported in DOGGR's Well Stimulation Treatment Disclosure Reports prior to May 5, 2015 that were not reported for hydraulic fracturing in the FracFocus Chemical Disclosure Registry (Appendix A, Table 2.A-1). Well Stimulation Treatment Disclosure Reports are required within 60 days of cessation of well stimulation treatment under SB 4.

Chemical Name	CASRN	Also reported as used in hydraulic fracturing in DOGGR's Disclosure Reports
1-Eicosene	3452-07-1	Yes
Hydroxylamine hydrochloride	5470-11-1	No
Acetaldol	107-89-1	No
1-Tetradecene	1120-36-1	Yes
1-Octadecene	112-88-9	Yes
Ammonium fluoride	12125-01-8	Yes
Benzyldimethylammonium chloride	122-18-9	Yes
Lauryl hydroxysultaine	13197-76-7	Yes
Benzododecinium chloride	139-07-1	Yes
Miristalkonium chloride	139-08-2	Yes
Nitrilotriacetic acid	139-13-9	No
Fatty acids, C18-unsatd., dimers	61788-89-4	No
Amines, hydrogenated tallow alkyl, acetates	61790-59-8	Yes
1-Hexadecene	629-73-2	Yes
Benzoic acid	65-85-0	No
Poly(oxy-1,2-ethanediyl), alpha-(nonylphenyl)-omega-hydroxy-, branched, phosphates	68412-53-3	No
Benzenesulfonic acid, C10-16-alkyl derivs., compds. with 2-propanamine	68584-24-7	Yes
Benzenesulfonic acid, C10-16-alkyl derivs., compds. with triethanolamine	68584-25-8	Yes
Copper dichloride	7447-39-4	No
Ethylene oxide	75-21-8	Yes

Chemical Name	CASRN	Also reported as used in hydraulic fracturing in DOGGR's Disclosure Reports
Potassium iodide	7681-11-0	No
Nitrogen	7727-37-9	No
Calcium phosphate, tribasic	7758-87-4	Yes
Aluminum chloride	7784-13-6	No
1,3-Propanediaminium, 2-hydroxy-N,N,N,N′,N′-pentamethyl-N′-(3-((2-methyl-1-oxo-2-propenyl) amino)propyl)-, dichloride, homopolymer	86706-87-8	No

As of June 2013, SCAQMD, which regulates air quality in the Los Angeles Basin, has required operators to report information on chemical use for well drilling, completion, and rework operations. Reports from June 2013 through May 2014 were examined for treatments and operations that used hydrochloric acid; it was found that over 70 other chemical compounds identified by CASRN were used in conjunction with hydrochloric acid, according to these mandated reports. Over 20 compounds were identified from this list that were not found in the voluntary notices reported to the Chemical Disclosure Registry (Table 2.A-4).

A full analysis of the environmental risks associated with the use of acid and associated chemicals, such as corrosion inhibitors, requires a more complete disclosure of chemical use. Many of the same chemicals that are used for hydraulic fracturing are also used for matrix acidizing and other acid applications. Concerns specific to matrix acidizing, that may or may not apply to other well maintenance activities or hydraulic fracturing, include the dissolution and mobilization of naturally occurring heavy metals and other pollutants from the oil-bearing formation. The significance of this risk, if any, cannot be evaluated without a more complete understanding of the chemicals being injected and of the fate and effect of well stimulation fluids in the subsurface. The composition of the fluids returning to the surface as return flows and produced water needs to be better understood (Section 2.5).

2.4.4. Characterization of Chemical Additives in Well Stimulation Fluids

2.4.4.1. Characterization by Additive Function

Chemicals added to well stimulation fluids have a variety of purposes, including thickening agents to keep sand and other proppants in suspension (e.g., gels and crosslinkers) and chemicals (breakers) added at the end of treatments to remove thickening agents, leaving the proppant to hold open the newly created fractures (King, 2012; Stringfellow et al., 2014). Table 2.4-4 lists chemical use by function, where the function could be positively identified. It is apparent that treatments using gels and cross-linking agents are more common in California than treatments using friction reducers (Table 2.4-4). In other regions of the country where stimulation is used for gas production, friction reducers (slicking agents) are commonly used (King, 2012; Stringfellow et al., 2014; U.S. EPA, 2015a). Over 80% of the treatments use an identified biocide and many formulations also include chemicals such as clay control additives. More information on the purposes of various chemicals used in hydraulic fracturing can be found elsewhere (King, 2012; Stringfellow et al., 2014; U.S. EPA, 2015a).

Disclosures frequently include descriptions of the purpose of the chemical added to well stimulation fluid. In the voluntary disclosures examined as part of this study, it was determined that the information entered for the purpose was very frequently inaccurate or misleading. In many cases, the purpose of the chemical additive is obscured because the disclosure reports list multiple purposes for each chemical disclosed. In other cases, the disclosed purposes are obviously incorrect. Impurities are typically not identified as such, and are instead given the same purpose description as the active ingredient in the chemical product. A more transparent explanation of the purpose of each chemical additive would contribute to a better understanding of the risks associated with well stimulation fluids.

Table 2.4-4. Hydraulic fracturing chemical use in California by function, where function was positively identified. This analysis was based on all records (N=45,058), consisting of 1,623 hydraulic fracturing treatments.

Function	Chemicals used for each function	Treatments using chemicals with this function
Breaker	11	1,599
Proppant	20	1,598
Gelling Agent	2	1,593
Carrier	23	1,515
Crosslinker	13	1,405
Biocide	10	1,392
Clay Control	7	1,184
Scale Inhibitor	10	865
Corrosion Inhibitor	8	182
Iron Control	2	60
Friction Reducer	1	13
Diverting Agent	3	10
Antifoam	1	6

2.4.4.2. Characterization by Frequency of Use

Although there are a large number of chemical additives used in well stimulation fluid (Appendix A, Table 2.A-1), the reported frequency of use of these compounds varies. As part of an environmental and hazard evaluation involving such an extensive list of chemicals, it is necessary to set priorities for which chemicals to evaluate first. Although any individual chemical use is potentially important, it is not practical to evaluate all chemicals simultaneously. In this study, we use frequency of use as one of several parameters (including toxicity and amount used) for recommending specific chemicals for priority evaluation. The more frequently a chemical is used, the more likely any associated hazard, if any, could become an environmental or health risk.

Table 2.4-5 lists the 20 reported additives used most frequently in California. This list excludes proppants (e.g., quartz), bulk fluids (e.g., water), and diatomaceous earth, which is added as a stabilizer or carrier to biocides and other active ingredients (Greene and Lu, 2010). Frequently used chemicals on the list include gels and cross-linkers (e.g., guar gum, boron sodium oxide), biocides (e.g., 5-chloro-2-methyl-3(2H)-isothiazolone), breakers (e.g., ammonium persulfate, enzymes), and other treatment additives. Additives in Table 2.4-5 include solvents and a clay stabilizer. As discussed previously, reporting of chemical use is not mandatory, but the most frequently reported chemicals (Table 2.4-5) are in alignment with what is expected from other lines of inquiry and reported literature (e.g., Stringfellow et al., 2014; U.S. EPA, 2004).

Table 2.4-5. Twenty most commonly reported hydraulic fracturing components in California, excluding base fluids (e.g., water and brines) and inert mineral proppants and carriers. This analysis was based on all records (N=45,058), consisting of 1,623 hydraulic fracturing treatments.

Chemical	CASRN	Treatments using this chemical
Guar gum	9000-30-0	1,572
Ammonium persulfate	7727-54-0	1,373
Sodium hydroxide	1310-73-2	1,338
Ethylene glycol	107-21-1	1,227
2-Methyl-3(2H)-isothiazolone	2682-20-4	1,187
Magnesium chloride	7786-30-3	1,187
Magnesium nitrate	10377-60-3	1,187
5-Chloro-2-methyl-3(2H)-isothiazolone	26172-55-4	1,184
Isotridecanol, ethoxylated	9043-30-5	1,171
Hydrotreated light petroleum distillate	64742-47-8	1,167
Distillates, petroleum, hydrotreated light paraffinic	64742-55-8	1,129
2-Butoxypropan-1-ol	15821-83-7	1,119
Hemicellulase enzyme	9025-56-3	1,098
1,2-Ethanediaminium, N1,N2-bis[2-[bis(2-hydroxyethyl)methylammonio]ethyl]-N1,N2-bis(2-hydroxyethyl)-N1,N2-dimethyl-, chloride (1:4)	138879-94-4	1,076
1-Butoxypropan-2-ol	5131-66-8	973
Phosphonic acid	13598-36-2	790
Amino alkyl phosphonic acid	Proprietary	668
Boron sodium oxide	1330-43-4	666
Sodium tetraborate decahydrate	1303-96-4	520
Enzyme G	Proprietary	480

In Appendix A, Table 2.A-5 contains a list of the approximately 150 chemical additives that were reported less than ten times in 1,623 applications. From a search of product literature, patents, and scientific literature, it can be determined with some certainty that many of the compounds in Table 2.A-5 are impurities (e.g., sodium sulfite), but many are clearly specific products applied for the purpose of well stimulation (e.g., FRW-16A, which is a stimulation fluid additive sold by Baker Hughes). Although the voluntary reporting indicates that these compounds are not widely used in California, the lack of mandatory reporting means that the frequency of use of these chemicals cannot be determined with certainty. Based on our analysis that the voluntary disclosure regime appears to produce representative data, we conclude that the additives that are reported less frequently (Table 2.A-5) deserve a lower priority for a complete risk analysis than compounds that are used more frequently (e.g., Table 2.4-5).

2.4.4.3. Characterization by Amount of Materials Used

Another criterion for selecting priority chemicals for a more thorough evaluation is the amount of material that is used. The concentrations for chemical additives that are used in median quantities greater than 200 kg (440 lbs) per hydraulic fracturing treatment are compiled in Appendix A, Table 2.A-6. This table does not include base fluids (water, saline solutions, or brine), which can account for over 85% of the mass of the well stimulation fluid. As would be expected, at least nine of the compounds in Table 2.A-6 (Appendix A) are proppants and many are solvents, crosslinkers, gels, and surfactants. Since the compounds listed in Table 2.A-6 (Appendix A) are used in significant amounts, they are considered to be priority compounds that warrant further investigation.

2.4.4.4. Characterization by Environmental Toxicity

For assessing environmental toxicity, aquatic species are typically exposed to varying concentrations of chemicals under controlled conditions and, after a specified time, the test species are examined for acute or chronic effects (U.S. EPA, 1994; OECD, 2013). Toxicity to the environment is inferred from tests against a variety of aquatic species that fall into the categories of fish, crustaceans, and aquatic plants, usually represented by algae. In these studies, the test animal is exposed to high concentrations of the test chemical, and the survival or health of the animals as a function of the exposure is determined, with the most common acute metric being the concentration at which 50% of the test population is expected to be adversely effected or dies, if the endpoint is lethality (see methods section). Since aquatic toxicity tests are highly standardized, the results can be used to compare and contrast industrial chemicals (Stringfellow et al., 2014).

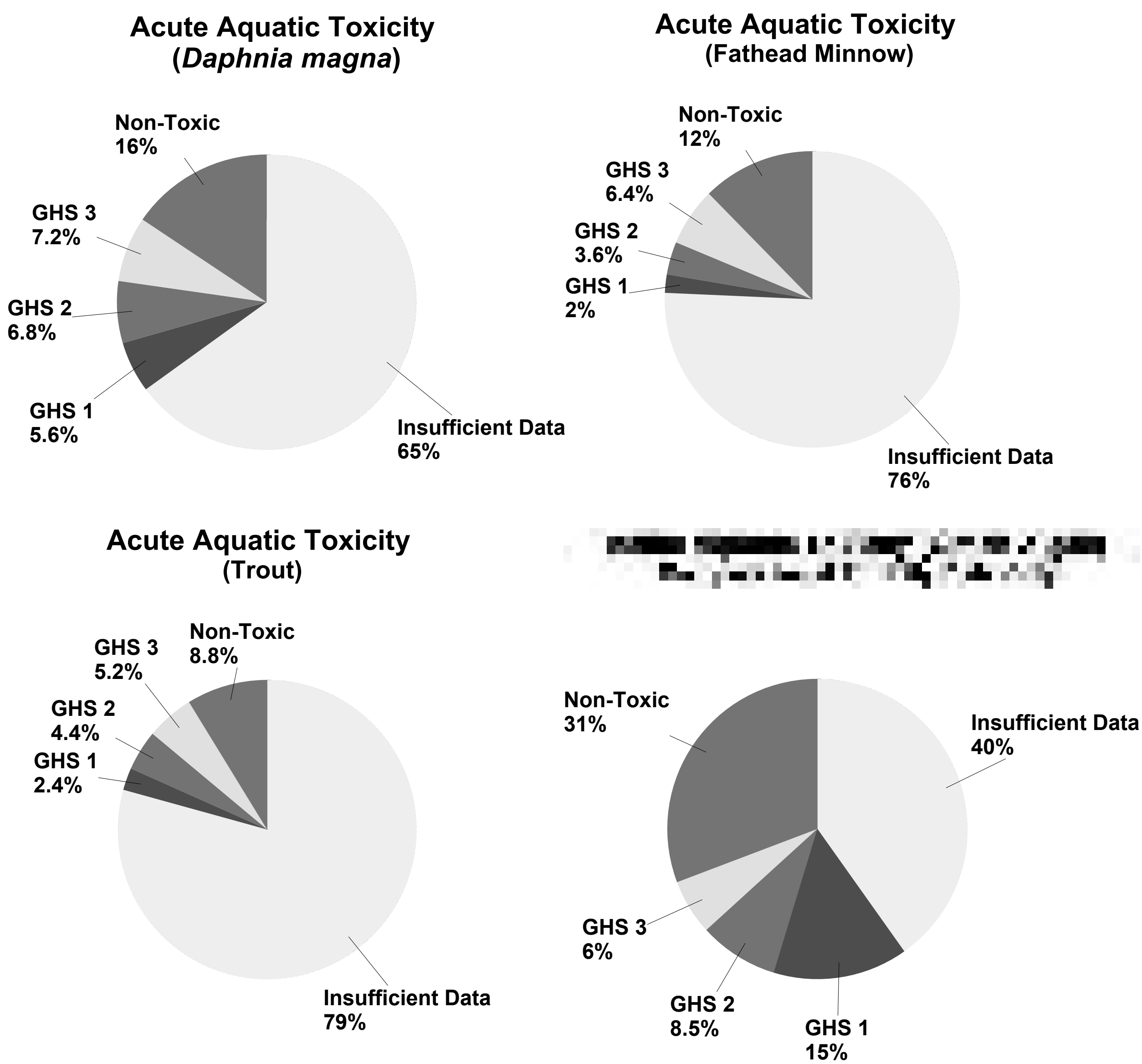


Figure 2.4-2. Aquatic toxicity data for all hydraulic fracturing and acid treatment chemicals. Chemical toxicity was categorized according to United Nations standards in the Globally Harmonized System of Classification and Labeling of Chemicals (GHS), which classifies acute toxicity for aquatic species on a scale of 1 to 3, with 3 being the least toxic.

An overview analysis of the experimental results for acute aquatic toxicity tests are presented in Figure 2.4-2. Thirty-three chemicals have a GHS ranking of 1 or 2 for at least one aquatic species (Table 2.A-7), indicating they are hazardous to aquatic species and could present a risk to the environment if released. Species for which toxicity data were collected are *Daphnia magna*, fathead minnows, and trout. The most toxic chemical additives for these aquatic organisms are shown in Table 2.4-6. Significant data gaps exist for aquatic species testing. *Daphnia magna* toxicity data are missing for 65% of the chemical additives identified by CASRN, fathead minnow toxicity data are missing for 76%, and trout data are missing for 79% of chemicals (Figure 2.4-2). EPI Suite estimations for green algae toxicity are missing for 40% of the chemicals (Figure 2.4-2).

Table 2.4-6. The most toxic hydraulic fracturing chemical additives used in California with respect to acute aquatic toxicity, based on the United Nations Globally Harmonized System (GHS) of Classification and Labeling of Chemicals system. Lower numbers indicate higher toxicity, with a designation of "1" indicating the most toxic compounds. Results are only shown for chemicals with GHS rating of 1 for any of the aquatic organisms in the analysis (Daphnia magna, fathead minnows, and trout).

Chemical Name	CASRN	GHS rating
2-Propenoic acid, ammonium salt (1:1), polymer with 2-propenamide	26100-47-0	1
2,2-dibromo-3-nitrilopropionamide	10222-01-2	1
2-Methyl-3(2H)-isothiazolone	2682-20-4	1
5-Chloro-2-methyl-3(2H)-isothiazolone	26172-55-4	1
Alcohols, C10-16, ethoxylated	68002-97-1	1
Alcohols, C12-13, ethoxylated	66455-14-9	1
Alkyl dimethylbenzyl ammonium chloride	68424-85-1	1
Chlorous acid, sodium salt (1:1)	7758-19-2	1
Ethoxylated C14-15 alcohols	68951-67-7	1
Glutaraldehyde	111-30-8	1
Hydrochloric acid	7647-01-0	1
Naphthalene	91-20-3	1
Quaternary ammonium chloride, benzylcoco alkyldimethyl, chlorides	61789-71-7	1
Solvent naphtha, petroleum, heavy arom.	64742-94-5	1

It is important to note that acute toxicity levels of many compounds from EPA standard tests of *Pimephales promelus* (fathead minnow) should be interpreted with caution, since they may differ from the sensitivity of California species. We examined relative toxicity (mortality) of a common well stimulation additive in a comparison between California freshwater fish and *Daphnia* and minnow species (Table 2.4-7). Several observations were made, including that (1) toxicity can vary by more than an order of magnitude among fish species, and (2) in almost all cases, fathead minnow was more resistant to the QAC than other California resident species. These data underscore the need to perform standardized toxicity tests with individual well stimulation chemicals and mixtures of well stimulation chemicals against California species, as well as standard test organisms. Additionally, toxicity will differ by life history stage, and many embryos or larvae may show much higher sensitivity to chemicals than adults, further illustrating that standard acute toxicity tests are just a first step in a more complete evaluation of chemicals (U.S. EPA, 2011).

The aquatic toxicity tests reviewed in this report describe the effects that varying concentrations of pure chemicals have on aquatic species, and are most applicable to effluents and other discharges released directly to surface waters. In the context of normal operations during well stimulation treatments, chemicals are injected in the subsurface, where they can interact with subsurface minerals and otherwise undergo chemical reactions before potentially contacting groundwater or surface water. For example, acids injected into formation rock react rapidly, and the acidity of the injected fluid diminishes quickly. Therefore, any comparison made between the concentrations assessed in toxicity tests and the concentrations reported in well stimulation fluids need to account for the fact that well stimulation fluids will typically be diluted and altered prior to any potential contact with either groundwater or surface water. Further study is required to understand how well stimulation fluids are altered as they interact with surrounding formation rock, and gaining knowledge of these chemical transformations needs to be an essential component of future risk assessment studies for unconventional oil and gas development.

Table 2.4-7. Comparison of results between standard test organisms and California native and resident species. Shown is a comparison of the lethal concentration to 50% of test organisms (LC_{50}) values across different aquatic species towards a common quaternary ammonium compound (QAC) used in hydraulic fracturing fluids. If different LC_{50} values for the same experimental conditions were present in the EPA's Pesticide Ecotoxicity Database, a range of test concentrations was noted. In addition, some experiments had different exposure duration when the effect was observed, leading to lower LC_{50} values with increasing exposure duration e.g., for the striped bass. (U.S. EPA and Office of Pesticide Programs, 2013; Bills et al., 1993; Krzeminski et al., 1977).

Species	Alkyl dimethylbenzyl ammonium chloride CASRN 68424-85-1 LC_{50} or EC_{50} ($\mu g\ L^{-1}$)
Water Flea (*Daphnia magna*)[1]	37–158
Fathead Minnow (*Pimephales promelas*)	280–1,400
Bluegill (*Lepomis macrochirus*)	68–5,300
Rainbow Trout (*Oncorhynchus mykiss*)[2]	64–7,690
Brown Bullhead (*Ameiurus nebulosus*)	1,590
Green Sunfish (*Lepomis cyanellus*)	2,250
Redear Sunfish (*Lepomis microlophus*)	740
Smallmouth Bass (*Micropterus dolomieu*)	1,370
Largemouth Bass (*Micropterus salmoides*)	1,130
Striped Bass (*Morone saxatilis*)	2,820–14,200
Channel Catfish (*Ictalurus punctatus*)	980
Brown Trout (*Salmo trutta*)	1,950
Lake Trout (*Salvelinus namaycush*)	420
Goldfish (*Carassius auratus*)	1,490

[1]*In the case of Daphnia magna, results are reported as effective concentration where 50% of the test population is immobilized at the indicated concentration (EC_{50}). For all other species, the results are measured as mortality (LC_{50}).*
[2]*Native California species, all other fish are non-native resident species.*

2.4.5. Selection of Priority Chemicals for Evaluation Based on Use and Environmental Toxicity

Identification of priority chemical additives for further investigation is an important step toward a complete understanding of the potential direct impacts of hydraulic fracturing and other well stimulation treatments. Using the information and analysis discussed above, we can develop a proposed list of priority chemical additives, based on toxicity and mass used (Appendix A, Table 2.A-8). Chemicals on this list were ranked and given a "Tox Code," representing the highest toxicity ranking the compound received under the GHS system for any environmental toxicity test using aquatic species. The Tox Code was combined with the analysis of the mass of chemical used per well stimulation treatment to allow better synthesis of information (Appendix A, Table 2.A-8). In Chapter 6, a similar approach is taken for the ranking of chemicals in the context of public health and expanded to create a human-health-hazard screening index and includes other impact factors in addition to toxicity and mass of chemical used.

The chemicals list in Table 2.A-8 represent the "known knowns," namely chemicals for which we have a CASRN and some level of toxicity information. In addition to the evaluation of these chemicals, we need to consider the "known unknowns," that for the majority of chemicals identified by CASRN we do not have sufficient toxicological information for characterization (Figures 2.4-2, Appendix B, 2.B-1, and 2.B-2). In addition, there are the "unknown unknowns," represented by the large number of chemicals (discussed below) that are not identified by CASRN (Appendix A, Table 2.A-9) and the large number of well stimulation treatments for which no information was reported under the voluntary disclosure system.

2.4.6. Chemical Additives with Insufficient Information to be Fully Characterized

Over 100 of the materials listed in Table 2.A-1 (see Appendix A) are identified by non-specific names and are reported as trade secrets, confidential business information, or proprietary information (Appendix A, Table 2.A-9). These materials cannot be evaluated for hazard, risk, and environmental impact without more specific identification. Chemical additives that are not identified by CASRN cannot be conclusively identified and cannot be fully evaluated. As can be seen from Tables 2.A-1 and 2.A-6, many of these unidentified or poorly identified compounds are used frequently or in significant amounts for well stimulation. Without complete identifying information, it is not possible to know if more than one chemical (a chemical mixture) is being reported using the same common name. Therefore, 100 chemicals could be the minimum number of completely unknown materials. Additives that were not identified by CASRN were not included in the hazard analysis discussed below.

Undefined chemicals should not be ignored, and some hazard information can be inferred from the reported common names. For example, the common names "oxyalkylated amine quat," "oxyalkylated amine," "quaternary amine," and "quaternary ammonium compound" all indicate that these additives fall into the category of quaternary ammonium compounds (QACs). Similarly, many of the general names suggest that the proprietary additives are surfactants (e.g., “ethoxylated alcohol,” “surfactant mixture,” etc.) that are widely used in the industry. Surfactants and QACs have broad application in both industry and household use, and QACs can be used as biocides (Kreuzinger et al., 2007; Sarkar et al., 2010). The environmental hazard associated with an individual surfactant or QAC is highly variable, and some QACs can be persistent in the environment (e.g., Garcia et al., 2001; U.S. EPA, 2006a; 2006b; Davis et al., 1992; Arugonda, 1999). In other disclosures, surfactants and QACs used for well stimulation are identified by CASRN, and evaluation of those chemicals can be used to give insight into the hazard associated with proprietary chemicals used for the same purpose.

2.4.7. Other Environmental Hazards of Well Stimulation Fluid Additives

In this report, we performed a hazard assessment of chemicals for which adequate information was available. A hazard is any biological, chemical, mechanical, environmental, or physical agent that is reasonably likely to cause harm or damage to humans, other organisms, or the environment in the absence of its control (Sperber, 2001). A chemical can be considered a hazard if it can potentially cause harm or danger to humans, property, or the environment because of its intrinsic properties (Jones, 1992).

The identification of hazards (or the lack thereof) is the first step in performing risk assessments. Once the hazards are established or defined, then the more involved process of risk assessment can begin. In contrast to hazard, risk includes the probability of a given hazard to cause a particular loss or damage (Alexander, 2000). It is important to note that it was beyond the scope of this study to perform a risk assessment, and that there are extensive data gaps on the chemical mixtures and environmental exposures that need to be addressed to enable future risk assessments. In addition, many of the materials listed in Appendix A, Table 2.A-1 are reactive and are expected to react with one another and/or other materials within the well and mineral formation. These byproducts could be more or less hazardous than the parent compounds examined here. Byproducts are not measured or reported, and thus could not be evaluated here.

2.4.7.1. Chronic and Sublethal Effects of Chemicals

In this chapter, the analysis of potential impacts from chemicals used in well stimulation fluids has focused on acute lethality to aquatic organisms. However, sublethal impacts from acute or chronic exposures are often related to individual survival potential and population viability (U.S. EPA, 1998). Impacts on reproduction and development are directly linked to population viability. Physiological status, disease or debilitation, avoidance behavior, and migratory behavior are identified as important to population viability in the U.S. Environmental Protection Agency’s Generic Ecological Assessment Endpoints (U.S. EPA, 2003).

Lack of data on chronic and sublethal impacts of chemicals used in well stimulation treatments represents a critical data gap in the analysis of potential ecological impacts of unconventional oil and gas development in California. However, the limited data available indicate that sublethal impacts may occur. Exposure to the biocide 2,2-dibromo-3-nitrilopropionamide (DBNPA) negatively impacts aquatic organisms at concentrations well below lethal levels. Growth of juvenile trout was impaired after 14 days exposure to 0.04 mg L^{-1} DBNPA (Chen, 2012). The same study showed impaired reproduction in aquatic invertebrates at 0.05 mg L^{-1} (Daphnia magna). *Xenopus laevis* tadpoles exposed to sublethal concentrations of the biocide methylisothiazolinone (MIT) during development showed several neurological deficits affecting behavior and susceptibility to seizures (Spawn and Aizenman, 2012). Chronic sublethal exposure to the surfactants linear alkylbenzene sulfonates (e.g., dodecylbenzene sulfonic acid) can impact the gills and olfactory system of fish (Zeni and Stagni, 2002; Asok et al., 2012) and decrease reproduction in invertebrates (da Silva Coelho and Rocha, 2010). More information is needed to assess the potential chronic and/or sublethal impacts of well stimulation fluids on aquatic species.

2.4.7.2. Environmental Persistence

The risk associated with a given chemical depends on how long the chemical persists in the environment. A toxic compound released into the environment that decays rapidly presents less chance for exposure to occur, damage to be inflicted, and risk to be accumulated. The list of chemicals used in hydraulic fracturing (Table 2.A-1) includes some compounds that could be environmentally persistent. For example, many of the chemical additives are surfactants and related compounds such as QACs. Persistence of surfactants and QACs is directly related to hydrocarbon chain length and other structural properties, with high molecular weight constituents likely to be the least volatile and most slowly degraded by microbes (Garcia et al., 2001; Kreuzinger et al., 2007; HERA, 2009; Li and Brownawell, 2010; Sarkar et al., 2010; Jing et al., 2012). Other compounds that may persist in the environment include the halogenated biocides DNBPA and MBNPA (2-bromo-3-nitrilopropionamide) and copper-EDTA (ethylenediaminetetraacetic acid).

A complete investigation of persistent pollutants found in well stimulation fluid is beyond the scope of this study, but this preliminary analysis suggests that potentially persistent pollutants and the reaction products of well stimulation fluid should be evaluated. Baseline measurements for current environmental levels of these compounds, including concentrations in biota as appropriate, are needed in order to determine whether or not these levels are altered by future exposure to well stimulation fluid.

A major mechanism for environmental attenuation of chemicals is biodegradation. Biodegradation in nature or in engineered treatment facilities removes chemicals from environmental systems. Biodegradable materials do not typically persist in the environment, regardless of whether they are released by accident or on purpose.

Standardized methods to measure the biodegradation potential allow the comparison and ranking of chemicals (OECD, 2013; U.S. EPA, 2011). Biodegradation tests only apply to organic compounds. The percentages of chemicals, which have been tested under standardized OECD test conditions and found to be biodegradable, not biodegradable, or for which biodegradation information is unknown, are shown in Figure 2.4-3. The "biodegradable" category includes all chemicals that are ranked as inherently or readily biodegradable by OECD protocols (OECD, 2013). The majority of chemicals that have been tested are biodegradable and therefore are not expected to persist in the environment (Figure 2.4-3). However, approximately one-half of the organic compounds identified by CASRN have not been tested for biodegradation by standardized methods, and many more compounds not identified by CASRN cannot be evaluated. Additionally, standardized biodegradation tests do not take into account chemical interactions that may occur, such as how the presence of biocides may affect the degradation of otherwise biodegradable compounds. Overall, it can be concluded that there is insufficient information to predict how these chemical mixtures will persist in the environment.

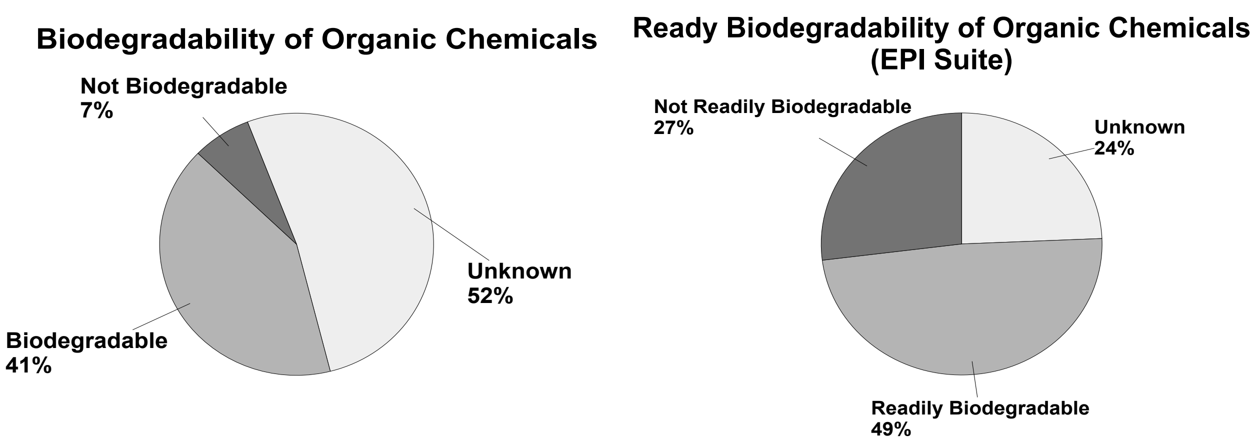


Figure 2.4-3. Biodegradability of chemicals. For pie charts containing both experimental and computational biodegradability data, the experimental data was used as the value for that chemical in the creation of the pie chart. If only computational data was available, the computational value was used. Computational results are generated for the U.S. EPA BIOWIN program which are not considered as reliable or accurate as experimental results (U.S. EPA, 2012b).

2.4.7.3. Bioaccumulation

Given the large numbers of compounds used in well-stimulation treatments, it is possible that some compounds or reaction products of those chemicals will persist in the environment. Compounds that persist in the environment present a greater risk, if released, than readily degradable compounds. Some persistent compounds may have the potential to "bioaccumulate" or become more concentrated in organisms than in the environment. This is particularly important for organisms higher up on the trophic food chain, such as humans. Trophic transfer of chemicals that bioaccumulate in exposed organisms to higher concentrations of a chemical, or its transformation products, than are found in the environment are an important exposure mechanism in ecological systems (Currie et al., 1997; Clements and Newman, 2006; Maul et al., 2006; Wallberg et al., 2001; Zhang et al., 2011).

Bioaccumulation is driven by contaminant uptake, distribution, metabolism, storage, and excretion (Connell, 1988; Mackay and Fraser, 2000). The potential for a chemical to bioaccumulate can be indicated by its physiochemical characteristics, such as the octanol-to-water partition coefficient (K_{ow}), which indicates the degree of lipophilicity. However, some chemicals may bioaccumulate despite physiochemical characteristics that indicate otherwise. Active transport of chemicals (Buesen et al., 2003) or the inhibition of efflux transporters (Smital and Kurelec, 1998) can also result in bioaccumulation. An analysis of all chemicals identified in this study indicated that characterization of octanol-to-water partition coefficients for these compounds has not been completed (Figure 2.4-4). Measurement of octanol-to-water partition coefficients and other basic physical and chemical characteristics, such as Henry's constants and sorption coefficients, are needed for development of a complete environmental profile of a chemical (Stringfellow et al. 2014; U.S. EPA 2011).

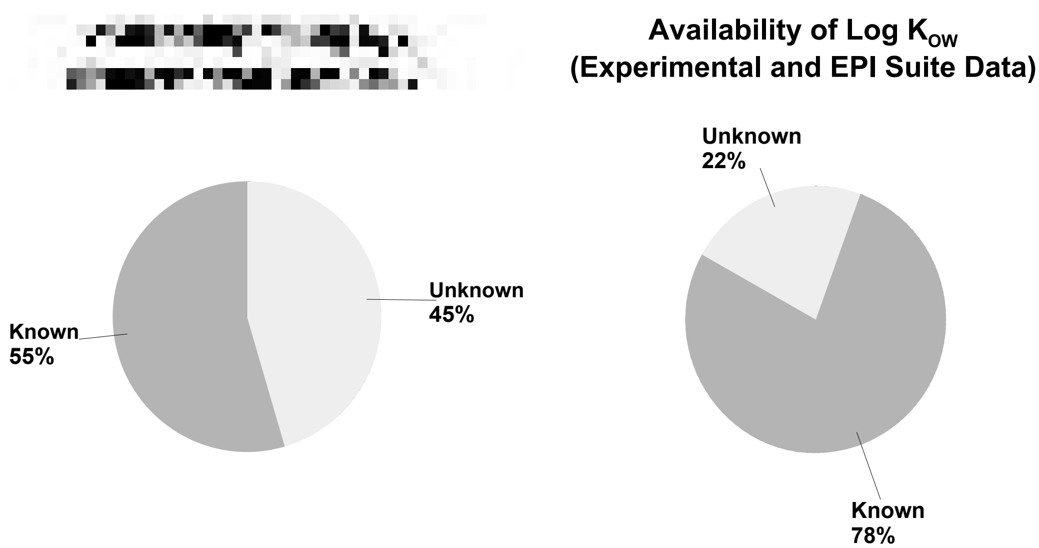
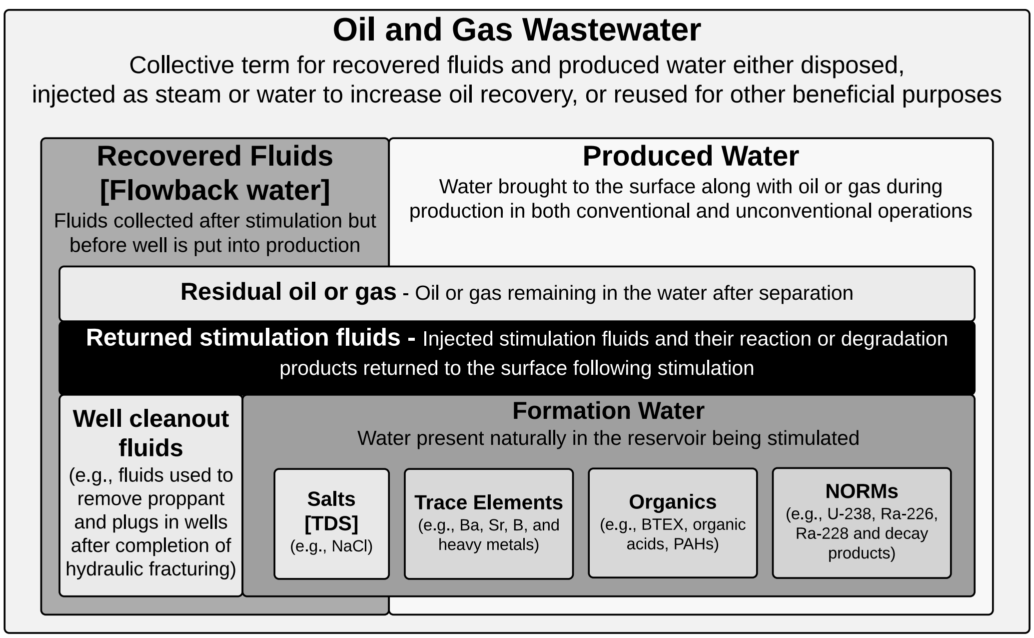


Figure 2.4-4. Availability of octanol-water partitioning measurements for hydraulic fracturing and acid treatment chemicals. The potential for a chemical to bioaccumulate can be indicated by its physiochemical characteristics, such as the octanol-to-water partition coefficient (K_{ow}) which indicates the degree of lipophilicity. Physical data such as octanol-to-water partition coefficients are needed to create a complete environmental profile on a chemical. Computational results are not considered as reliable or accurate as experimental results.

2.5. Wastewater Characterization and Management

2.5.1. Overview of Oil and Gas Wastewaters

Both stimulated and non-stimulated wells generate water as part of oil and gas production over the lifetime of the wells. This water byproduct is referred to as "produced water," which consists of formation water mixed with oil and gas that is brought to the surface during production. For stimulated wells, the additional term "flowback" is commonly used to describe the fluids recovered after the well pressure is reduced following stimulation, but *before* the well is put into production (Pavley, 2013; U.S. EPA, 2012a; Vidic et al., 2013). New California regulations introduce another term, "recovered fluids," which is defined as the water returned "following the well stimulation treatment that is not otherwise reported as produced water" (DOGGR, 2014e). The U.S. EPA (U.S. EPA, 2012a) and others use the term "wastewater" to refer to all fluids that return to the surface along with the oil and gas, including recovered fluids, flowback, and produced water. Figure 2.5-1 illustrates the complex nature of wastewater from unconventional oil and gas development.



*** Boxes are not drawn to scale and are separated for visual clarity.**

Figure 2.5-1. The water returned from stimulated wells in California consists of recovered fluids (i.e., flowback water) and produced water, which can be disposed of as wastewater or beneficially reused. The recovered fluids in California are typically generated in small quantities and can contain returned stimulation fluids, well cleanout fluids and formation water. The produced water consists primarily of formation water (also referred to as formation brines due to its high salt content), as well as some residual oil or gas, and an unknown amount of returned stimulation fluids. The concentrations and composition of the returned stimulation fluids in both the recovered fluids and produced water is currently unknown. Note that the boxes are not drawn to scale and are separated for visual clarity.

Wastewater from well stimulation operations can contain a variety of constituents, including (1) the additives pumped into the well during well stimulation; (2) compounds that formed due to transformation or degradation of the additives, or to chemical reactions between the additives; (3) dissolved substances from waters naturally present in the target geological formation; (4) substances mobilized from the target geological formation; and (5) some residual oil and gas (NYSDEC, 2011; Stepan et al., 2010). It is expected that the amount of stimulation fluids returned is highest immediately following well stimulation, with a decrease in concentration over time (Barbot et al., 2013; Clark et al., 2013; Haluszczak et al., 2013; King, 2012). The period during which returned stimulation fluids come to the surface following stimulation varies between and within a region, but can range from a few hours to several weeks in shale producing natural gas (Barbot et al., 2013; Hayes, 2009; Stepan et al., 2010; Warner et al., 2013b). Studies have not been conducted to determine the return period for simulation fluids used for oil production in diatomite, as found in California. It is likely that, in California, stimulation fluids, chemical additives, and their reaction byproducts will be present in the water returned to the surface after the well is put into production, and thus will be present in produced water.

New California monitoring and reporting requirements focus on testing and management of recovered fluids and do not require extensive measurement or monitoring of produced water, which is likely to contain some of the stimulation fluids and their degradation byproducts. A recent white paper from DOGGR notes “When well stimulation occurs, most of the fluid used in the stimulation is pumped to the surface along with the produced water, making separation of the stimulation fluids from the produced water impossible. The stimulation fluid is then co-disposed with the produced water” (DOGGR, 2013). The combined handling of wastewaters generated during unconventional oil and gas production makes collection of better data and full characterization of wastewaters over time an important component of understanding the environmental impacts of hydraulic fracturing. The lack of studies on these wastewaters is identified as a major day gap.

In this section, we summarize data available on the quantities and characteristics of wastewater generated from stimulated wells in California. In our analysis, we evaluate the following questions:

- What are the quantities of recovered and produced water generated from stimulated wells within the first few months following stimulation, and are these volumes different from the quantities of produced water generated by non-stimulated wells in California?

- What are the chemical compositions of recovered fluids and produced water from stimulated wells? Is produced water from stimulated wells compositionally different than produced water from non-stimulated wells?

- How are recovered and produced waters from stimulated wells managed, i.e., how are they handled onsite, treated, reused and/or disposed?

2.5.2. Recovered Fluids Generated from Stimulated Wells in California

Recovered fluids are the fluids that are returned to the surface *before* production commences. According to one California operator, the recovered fluids can be a mixture of water from the formation, returned stimulation fluids, and well clean-out fluids (pers. comm., Nick Besich, Aera Energy). Operators are required to disclose "the source, volume, and specific composition and disposition" of the recovered fluids in well completion reports submitted to DOGGR within 60 days following stimulation.

2.5.2.1. Quantities of Recovered Fluids

We determined the quantities of recovered fluids from 506 completion reports filed and posted as of December 15, 2014, for 499 hydraulic fracturing and seven matrix acidizing treatments (DOGGR, 2014a). We first compared the volume of recovered fluid from each well to the corresponding volume of injected stimulation fluids to estimate the maximum recovery of stimulation fluids during the initial phase of wastewater production. One well where the injected volume was reported as zero was excluded from this analysis. Actual recoveries are likely to be lower, but could not be calculated, since the concentrations or masses of stimulation fluid constituents in the recovered fluids are not measured. We also compared the volumes of recovered fluids to the produced water generated during the first month of production, for records where matching production data were available in the DOGGR Production database, to put the recovered fluid volumes in the context of total wastewater generated immediately after stimulation. Wells for which the production volume for the first month or the volume of recovered fluid were reported as zero have been excluded from this analysis.

The volumes of recovered fluids collected from both hydraulic fracturing and acid matrix treatments range from 0 to 1,600 m^3 (9,900 barrels) (Table 2.5-1). The recovered fluid volumes are small (mostly less than 5%) compared to the injected fluid volumes for hydraulic fracturing treatments (Figure 2.5-2). There were eighteen hydraulic fracturing treatments for which the recovered fluid volumes were reported as zero, which could either be errors or indicate that fluids were directly diverted into the production pipeline without capturing any recovered fluid. Hence, the recovered fluid is conclusively a small portion of the fluids injected as part of a hydraulic fracturing treatment. In contrast, the recovered fluids from matrix acidizing potentially represent a much larger fraction (50–70%) of the stimulated fluids for the matrix acidizing operations (Table 2.5-1). The actual recovery of returned stimulation fluids has not been investigated and would require chemical analysis to differentiate between returning well stimulation fluids and connate water. However, the actual recovery of returned well stimulation fluids is likely to be lesser than the reported volumes of recovered fluid, since the recovered fluids can also contain well cleanout fluids and formation water (Section 2.5.2.2).

Table 2.5-1. A comparison of total recovered fluid and injected fluid volumes for stimulated wells located throughout California, as reported in DOGGR completion reports as of Dec 15, 2014 (N=505). All numbers are rounded to two significant figures. St. dev. = standard deviation; min. = minimum, max. = maximum.

	Matrix Acidizing (N=7)			**Hydraulic fracturing (N=498)**	
	Recovered Volume m^3 (barrels)	Injected Volume m^3 (barrels)		Recovered Volume m^3 (barrels)	Injected Volume m^3 (barrels)
Median	150 (970)	240 (1,500)		11 (72)	300 (1,900)
Average	170 (1,100)	270 (1,700)		77 (480)	410 (2,600)
St. Dev.	71 (450)	100 (650)		240 (1,500)	420 (2,600)
Min.	84 (530)	150 (960)		0 (0)	37 (230)
Max.	290 (1,800)	430 (2,700)		1,600 (9,900)	2,600 (16,000)

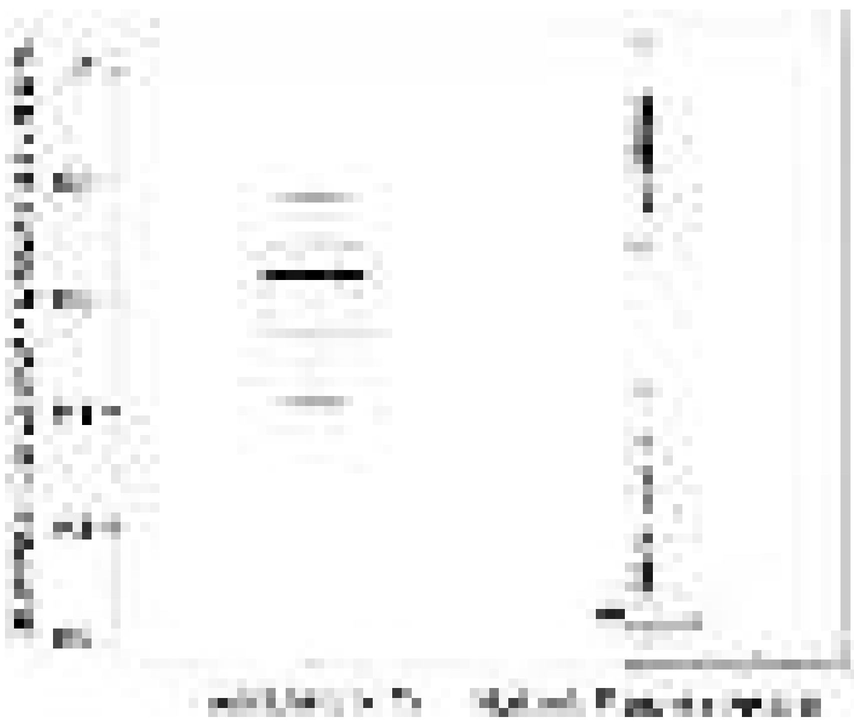


Figure 2.5-2. The fraction of recovered fluid volumes compared to the injected stimulation fluid volumes was significantly higher for acid matrix treatments (50-70%), when compared to hydraulic fracturing treatments. Typically, hydraulic fracturing treatments had very small recoveries (<5%), though there were many cases in which the recovered fluid volumes were much higher. Boxes show the 25th to 75th percentiles of the data, and the central lines show the median. Whiskers extend to 1.5 times the interquartile range from the box. The circles represent the outliers in the data. Data Source: DOGGR Completion reports as of Dec 15, 2015.

The recovered fluids were an extremely small fraction of wastewater generated within just the first month of production (Figure 2.5-3). The volume of produced water in the first month of operations was also substantially larger than the volumes of injected stimulation fluids for both hydraulic fracture and acid matrix treatments.

These analyses show that for hydraulic fracturing operations, the recovered fluids are a fraction of the amount of fluid injected, suggesting that produced water will likely contain some amount of fracturing fluids. Operators are currently required to only report chemical analysis results for the recovered fluids (Section 2.5.2.2), but there is no data available or reported about the masses of stimulation fluids (or their degradation byproducts) present in produced waters. The amount and fate of the injected fracturing fluids that is left behind in the subsurface is unknown.

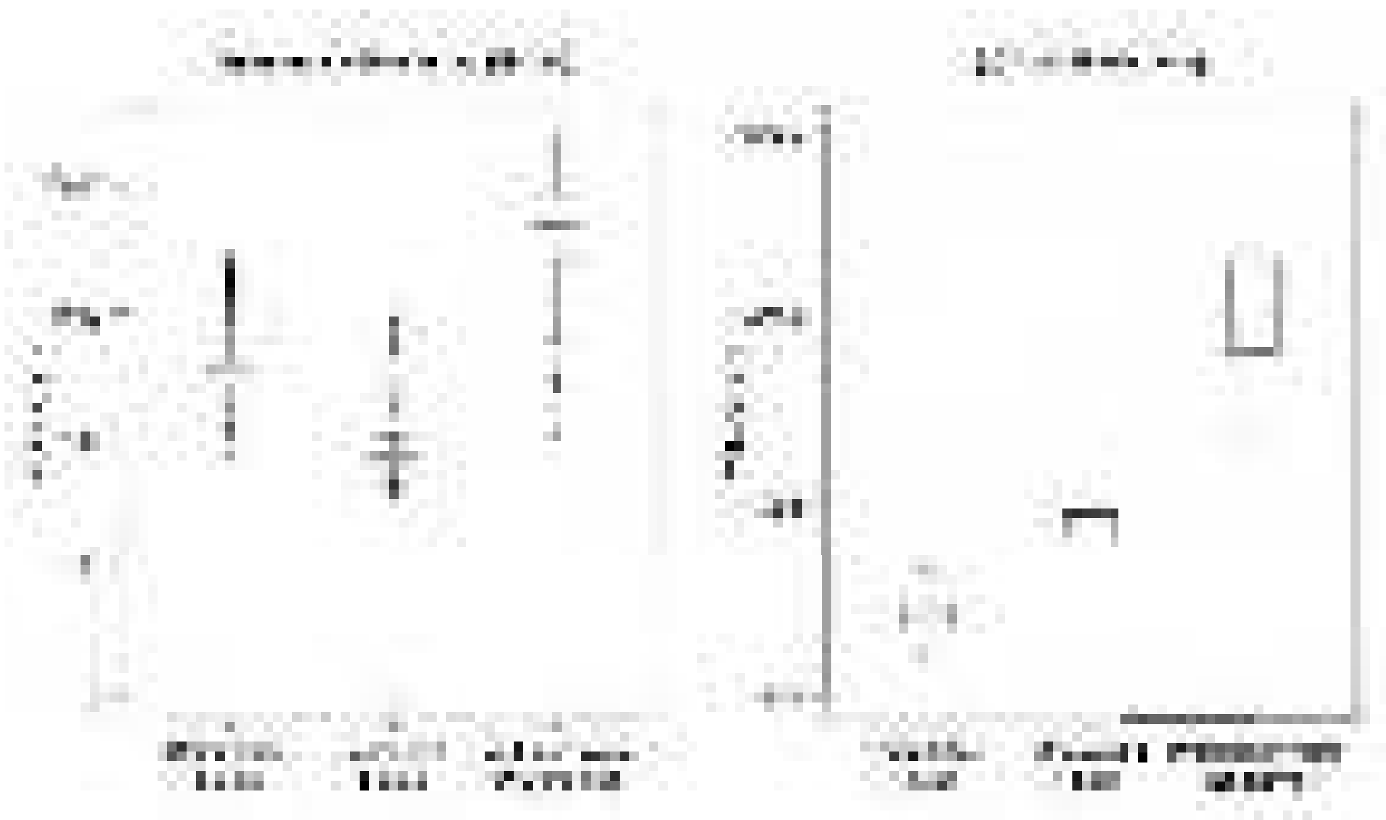


Figure 2.5-3. Volumes (log-scale) of injected fluids, recovered fluids, and produced water in the first month of production for (a) hydraulically fractured and (b) matrix acidizing treatments for wells that were reported in the DOGGR completion reports as of Dec 15, 2014 that had matching records in the DOGGR Production database. Wells that did not have any production within the first month were not considered in this analysis. Boxes show the 25th to 75th percentiles of the data, and the central lines show the median. Whiskers extend to 1.5 times the interquartile range from the box. The circles represent the outliers in the data.

Under new regulations, recovered fluids are now being characterized before disposal, and the results are included in well completion reports submitted to DOGGR. We investigated 45 laboratory chemical analyses that were submitted for onshore stimulated wells as of July, 2014. These data were made available as PDF files and represent waters recovered from operations in two fields (North and South Belridge) by one operator (Aera Energy). Operators are not required to report when the samples were collected after stimulation. According to the operator, the sample "is collected somewhere in the middle of recovery, but operationally that does not always happen." (Aera Energy, Appendix 2.F). Analyses include total carbohydrate, because the carbohydrate guar is a commonly used gelling agent in well stimulation, but this is the only stimulation additive for which a specific measurement was made. Other constituents that were measured include TDS, trace metals, organics, and naturally occurring radioactive materials (NORM) (Table 2.5-2).

Carbohydrates were detected in some of the recovered fluids, suggesting that there may be other stimulation chemicals present as well (Table 2.5-2). Some of the recovered fluids contained high concentrations of TDS, some trace elements (arsenic, selenium and barium), NORM, and hydrocarbons (Table 2.5-2). TDS levels were as high as 260,000 mg L^{-1}. Observed concentrations of the measured parameters were highly variable across wells, even though samples were limited to one operator and two fields. These results confirm that the recovered fluids represent multiple wastewater sources, including formation water and returned stimulation fluids, as was described by the operator (Aera Energy, Appendix 2.F).

The new regulations that go into effect July 2015, are more specific about when the samples for recovered fluids should be collected, and will also require an additional sample for produced water. The new regulations state that the operators must report the "composition of water recovered from the well following the well stimulation treatment, sampled after a calculated wellbore volume has been produced back but before three calculated wellbore volumes have been produced back, and then sampled a second time after 30 days of production after the first sample is taken, with both samples taken prior to being placed in a storage tank or being aggregated with fluid from other wells" (DOGGR, 2014d).

Table 2.5-2. Chemical analyses reported for recovered fluids collected from stimulated wells in North and South Belridge. Measured constituents include salts (TDS), trace metals, organics, NORM and guar (total carbohydrate). Constituents below the detection limit are marked as "ND." A limited amount of data is also available for concentrations of chemical constituents in produced water samples collected (before 1980) from conventional wells across California. All numbers are rounded to two significant digits.

Parameter	Recovered Fluids [a]	Conventional Oil and Gas [b]
General		
Total Dissolved Solids @180 C (mg L^{-1})	430 - 260,000	1,000 – 85,000
Conductivity (μmhos cm^{-1})	240 - 77,000	
pH	6.4 - 9.4	2.6 - 12
Temperature (degrees F)	64 - 130	
Bicarbonate Alkalinity as $CaCO_3$ (mg L^{-1})	ND - 2,900	
Bicarbonate (mg L^{-1})		0 – 13,000
Carbonate Alkalinity as $CaCO_3$ (mg L^{-1})	ND - 470	
Hydroxide Alkalinity as $CaCO_3$ (mg L^{-1})	ND - 0	
Total Alkalinity as $CaCO_3$ (mg L^{-1})	69 - 2,900	0 - 2,100
Major Cations		
Calcium (mg L^{-1})	10 - 13,000	0 – 14,000
Magnesium (mg L^{-1})	7.5 - 700	0 - 2,300
Sodium (mg L^{-1})	93 - 130,000	0 – 100,000
Potassium (mg L^{-1})	2.1 - 66,000	0 – 8,000
Aluminium (mg L^{-1})		0 - 250
Major Anions		
Bromide (mg L^{-1})	ND - 150	1 - 200
Chloride (mg L^{-1})	130 - 190,000	0 - 160,000
Fluoride (mg L^{-1})	ND - 3	
Nitrate as NO3 (mg L^{-1})	ND - 26	0 - 18
Sulfate (mg L^{-1})	28 - 1,900	0 - 15,000
Trace Elements		
Hexavlent Chromium (μg L^{-1})	ND - 9.5	
Antimony (μg L^{-1})	ND - 240	
Arsenic (μg L^{-1})	ND - 1,300	
Barium (μg L^{-1})	ND - 13,000	0 - 170
Beryllium (μg L^{-1})	ND - 50	
Boron (mg L^{-1})	0.26 - 110	0 - 600
Cadmium (μg L^{-1})	ND - 83	
Chromium (μg L^{-1})	ND - 160	0 - 200
Cobalt (μg L^{-1})	ND - 130	
Copper (μg L^{-1})	ND - 1,300	0 - 100
Iron (mg L^{-1})		0 - 540

Parameter	**Recovered Fluids [a]**	**Conventional Oil and Gas [b]**
Lead (μg L^{-1})	ND - 88	
Lithum (mg L^{-1})	ND - 41	
Manganese (mg L^{-1})		0 - 50
Mercury (μg L^{-1})	ND - 0.3	
Molybdenum (μg L^{-1})	ND - 500	
Nickel (μg L^{-1})	ND - 260	0 - 30
Selenium (μg L^{-1})	ND - 510	
Silver (μg L^{-1})	ND - 42	
Strontium (mg L^{-1})	0.25 - 230	0 - 600
Thallium (μg L^{-1})	ND - 0	
Vanadium (μg L^{-1})	ND - 220	
Zinc (μg L^{-1})	ND - 1,600	
Radioactivity/NORM		
Recoverable Uranium (pCi L^{-1})	ND - 95	
Gross Alpha (pCi L^{-1})	ND - 220	
Radium 226 (pCi L^{-1})	0.230 - 86	
Radium 228 (pCi L^{-1})	0-52	
Organics (VOCs)		
Benzene (μg L^{-1})	ND - 1,300	
Ethylbenzene (μg L^{-1})	ND - 470	
Toluene (μg L^{-1})	ND - 3,400	
Total Xylenes (μg L^{-1})	ND - 3,600	
p&m Xylenes (μg L^{-1})	ND - 2,500	
o-Xylene (μg L^{-1})	ND - 1,100	
Organics (PAHs)		
Acenaphthene (μg L^{-1})	ND - 86	
Acenaphthylene (μg L^{-1})	ND - 9.8	
Anthracene (μg L^{-1})	ND - 6.5	
Benzo[a]anthracene (μg L^{-1})	ND - 9.8	
Benzo[b]fluoranthene (μg L^{-1})	ND - 3.3	
Benzo[k]fluoranthene (μg L^{-1})	ND - 4.9	
Benzo[a]pyrene (μg L^{-1})	ND - 15	
Benzo[g,h,i]perylene (μg L^{-1})	ND - 0.56	
Chrysene (μg L^{-1})	ND - 20	
Dibenzo[a,h]anthracene (μg L^{-1})	ND - 0	
Fluoranthene (μg L^{-1})	ND - 4.1	
Fluorene (μg L^{-1})	ND - 140	
Indeno[1,2,3-cd]pyrene (μg L^{-1})	ND - 0.85	
Naphthalene (μg L^{-1})	ND - 730	
Phenanthrene (μg L^{-1})	ND - 180	
Pyrene (μg L^{-1})	ND - 6.1	

Parameter	Recovered Fluids [a]	Conventional Oil and Gas [b]
Oil and Gas		
Total Petroleum Hydrocarbons -Crude Oil (μg L^{-1})	ND - 6,700,000	
Methane (mg L^{-1})	ND - 5.4	
Stimulation Fluid Constituents		
Total Carbohydrates (μg L^{-1}) - Guar Indicator	0 - 3,700,000	

[a] *From DOGGR Completion Reports. (N=45, submitted from January 2014 to July 2014).*

[b] *Compiled for this report from the USGS Produced Water Database 2.0 (USGS, 2014b). (N=800).*

3.5.2.3. Management of Recovered Fluids

Recovered fluids are typically stored in tanks at the well site prior to disposal or reuse. According to well completion reports, more than 99% of these fluids are injected into Class II disposal wells. A small amount (0.2%) of recovered fluids are recycled, for example, in future well cleanout operations (Aera Energy, Appendix 2.F).

2.5.3. Produced Water Generated from Stimulated Wells in California

The majority of wastewater from stimulation operations is generated after the well is put into production. Data on produced water volumes and disposition are maintained in DOGGR's production database (DOGGR, 2014c). In California on average, approximately ten barrels of produced water are generated for every barrel of oil extracted (Clark and Veil, 2009). In California, well stimulation typically occurs in oil and gas fields that had long-term conventional production (CCST et al., 2014; Volume I). The produced water streams from stimulated wells are combined with those from conventional wells and treated as one waste stream. Operators are required to submit monthly reports to DOGGR on the volume of oil, gas, and water produced from their wells and the disposition method. These data include produced water disposal, as well as reuse in subsequent oil and gas operations or other beneficial uses.

2.5.3.1. Quantities of Produced Water

We compared the volumes of produced water from stimulated and non-stimulated wells to determine if they were different. Monthly produced water volumes for the first six months of oil production from DOGGR's production database were used for this analysis. The records used from the database were for wells in stimulated and non-stimulated pools in Kern County, which had oil production between January 1, 2011 and September 30, 2013. Only wells with at least 10 months of production data were included. Limiting the data to wells in Kern County focused the analysis on wells located where most well stimulation is occurring. Data on non-stimulated wells in other counties were not included, because of possible regional differences in wastewater production. Multiple stimulation events at individual wells were excluded from the analysis, in order to prevent bias in the results. In this analysis, volumes of produced waters were evaluated for 1,414 stimulated and 3,247 non-stimulated wells.

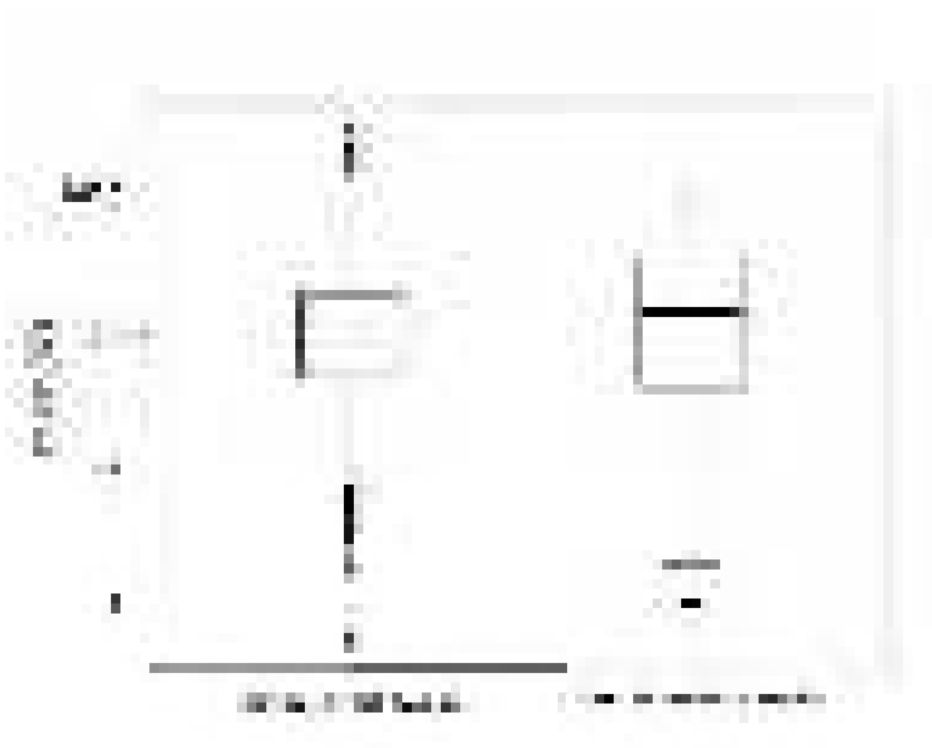


Figure 2.5-4. A comparison of quantities of produced water generated in the first 6 months of oil production from stimulated (N=1,414) and non-stimulated wells (N=3,247) in Kern County. Only wells that had oil production between January 1, 2011 and September 30, 2013 for which there were 10 months of continuous production data were included in the analysis. Note the log-scale in the Y-axis. Boxes show the 25th to 75th percentiles of the data, and the central lines show the median. Whiskers extend to 1.5 times the interquartile range from the box. The circles represent the outliers in the data.

The data do not show substantive differences between the volumes of produced water generated in the first six months from stimulated wells and non-stimulated wells (Figure 2.5-4), even though their distributions were different (Figure 2.5-5).

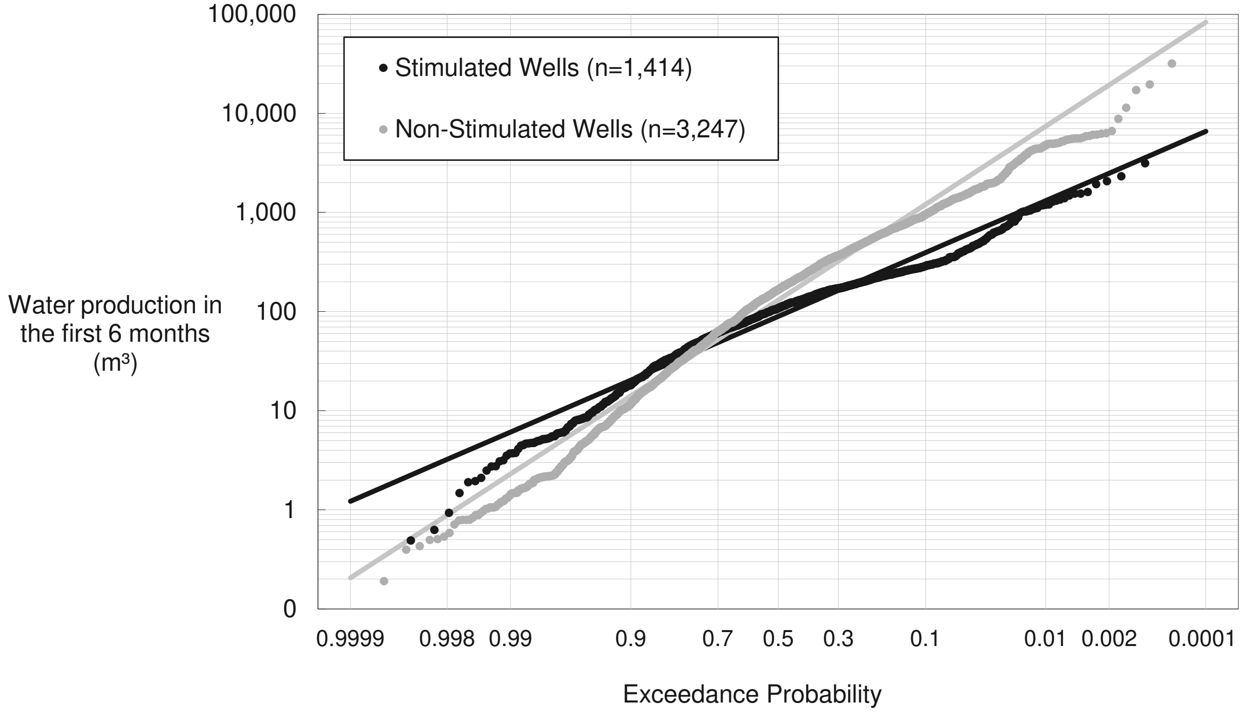


Figure 2.5-5. Probability plot comparing the distributions of produced water volumes for stimulated and non-stimulated wells. The X-axis represents an exceedance probability - i.e., the probability that the produced water generated will exceed a certain value. The Y-axis is on a log scale and has observations of the volumes of produced water in the first 6 months of production, in m^3, for oil and gas wells in Kern County, California with at least 10 months of production data from January 1, 2011 to September 30, 2013. For example, there is a 90% probability that the volume of produced water will exceed 10 m^3 (~60 barrels) for stimulated wells vs. 15 m^3 (~95 barrels) for non-stimulated wells, and a 10% probability that the volume of produced water will exceed ~ 300 m^3 (~1,900 barrels) for stimulated wells vs. ~900 m^3 (5,700 barrels) for non-stimulated wells. These data show that both of the distributions are different, and that there may be a few cases where the non-stimulated wells produce more water than stimulated wells and vice-versa.

2.5.3.2. Chemical Constituents of Produced Water

There are no published studies that have characterized the chemical constituents of produced water from stimulated wells in California. Operators are not required to report the composition of produced water from stimulated wells. New regulations that take effect July, 2015, will require collection of one produced water sample initially and then another "after 30 days of production after the first sample is taken." This is an inadequate sampling regime to characterize how, or if, well-stimulation-fluid additives or their reaction products are returning with produced water.

Since data on produced water specifically from stimulated wells were not available at the time of writing this report, we identified potential constituents that could be present in produced water from stimulated wells based on (1) studies that have analyzed the compositions of produced water from conventional oil and gas wells in California (e.g., Benko and Drewes, 2008), and (2) a few published studies that have characterized produced water from stimulated wells in other regions (CCST et al., 2014 and references therein). Some historical data on produced water composition in California are available in the USGS produced water database (USGS, 2014b), but data for several constituents are not available (Table 2.5-2). Additionally, the produced water can contain returned stimulation fluids, as discussed above.

Produced water from conventional wells primarily consists of water from the targeted formation. Formation water can contain naturally occurring dissolved constituents, such as salts (measured as total dissolved solids or TDS), trace elements, organic compounds, and naturally occurring radioactive materials (NORM). The most concentrated constituents measured in produced water from both conventional and unconventional wells are typically salts, i.e., sodium and chloride (Barbot et al., 2013; Blauch et al., 2009; CCST et al., 2014; Haluszczak et al., 2013; Warner et al., 2012a; 2012b). Magnesium and calcium can also be present at high levels and can contribute to increased water hardness. The TDS concentrations of produced water from conventional wells in California are typically around 10,000–30,000 mg L^{-1} (CCST et al., 2014), although concentrations can be as high as 85,000 mg L^{-1} (Table 2.5-2).

Formation brines can contain high concentrations of trace elements, such as boron, barium, strontium, and heavy metals, which may be brought up to the surface in the produced water (Table 2.5-2). For example, several studies report measuring high levels of trace elements such as barium, strontium, iron, arsenic, and selenium in the waters recovered from fracturing operations in the Marcellus Shale (e.g., Balaba and Smart, 2012; Barbot et al., 2013; Haluszczak et al., 2013; Hayes et al., 2009). Produced waters from oil and gas operations, including those in California, also contain many organic substances, e.g., organic acids, polycyclic aromatic hydrocarbons (PAHs), phenols, benzene, toluene, ethylbenzene, xylenes, and naphthalene (e.g., Fisher and Boles, 1990; Higashi and Jones, 1997; Veil et al., 2004).

Wastewaters from some shale formations have been found to contain high levels of NORM that were several hundred times U.S. drinking water standards (Barbot et al., 2013; Haluszczak et al., 2013; NYSDEC, 2009; Rowan et al., 2011). In 1996, a study of NORM in produced waters in California conducted by DOGGR (DOGGR, 1996) measured bulk radioactivity and some NORM elements (K-40, U-238, U-235, Ra-226, Ra-228 and Cs-137) in both solid and liquid samples. The study found several produced water samples containing elevated levels of radium greater than 25 pCi g^{-1}, but DOGGR did not consider radium to constitute a public health hazard at the time because “produced waters are not used as a source of drinking water.” However, there are several mechanisms by which produced water can be released into surface and groundwater resources (Section 2.6), and hence elevated levels of potentially contaminating constituents, including NORM, that occur in produced water should be included in future assessments.

More study is needed on produced water in California, particularly characterization of produced water from stimulated wells. Historical (pre-1980) data available on the composition of produced water from conventional wells in California may not be relevant to stimulated wells. The fraction of injected chemicals that return to the surface, and the time period over which they return, are unknown. In addition, the fundamental biogeochemical processes affecting stimulation fluids under reservoir temperature and pressure conditions in the presence of formation minerals have not been investigated. However, it is known that chemical additives are degraded, transformed, sorbed, and otherwise modified in the subsurface, since both specific and non-specific reactions, including strong acid and oxidation reactions, are part of the stimulation process (King, 2012). Other processes, such as biological degradation or transformation of stimulation chemicals, as well as mobilization of formation constituents, can also occur and influence the composition of produced water (Piceno et al., 2014). More data on produced water composition from stimulated and conventional wells in California are needed to assess whether stimulation could affect the produced water chemistry.

2.5.3.3. Management of Produced Water

2.5.3.3.1. Produced Water from Onshore Oil and Gas Operations

As described above, produced water from stimulated wells may contain well-stimulation-chemical additives. Monthly data (1977 to the present) on disposal of produced water are available in DOGGR’s Monthly Production database. An analysis was conducted on 2,018 documented well stimulation events which took place between 2011 and 2014 (Volume I, Appendix O) and it was found that data on produced water disposition were available from DOGGR’s Monthly Production database for 1,657 wells. For each well for which data was available, we examined disposition during (1) the first full month after stimulation occurred, and (2) from the date of initial well stimulation through June 2014. These results are presented in Table 2.5-3 and Figure 2.5-6.

Table 2.5-3. Produced water disposition during the first full month after stimulation and post stimulation to the present, January 1, 2011-June 30, 2014. Data from the DOGGR Monthly Production database.

	Number of Wells		Total Volume (First Full Month After Stimulation)			Total Volume (Stimulation to June 2014)		
			(m^3)	(acre-feet)	%	(m^3)	(acre-feet)	%
Evaporation–percolation	890	54%	720,000	580	57%	11,000,000	9,200	58%
Subsurface injection	470	28%	330,000	260	26%	4,100,000	3,300	21%
Other	130	8%	180,000	140	14%	3,400,000	2,700	17%
Not reported	150	9%	31,000	25	3%	510,000	410	3%
Surface body of water	2	0.1%	2,100	1.7	0.2%	95,000	77	0%
Unknown	14	1%	-	-	-	-	-	-
Sewer system	-	-	-	-	-	-	-	-
Evaporation - lined pits	-	-	-	-	-	-	-	-
Total	**1,700**	**100%**	**1,300,000**	**1,000**	**100%**	**19,000,000**	**15,769**	**100%**

Note: All numbers rounded to two significant figures. Numbers may not add up due to rounding. Subsurface injection includes injection into Class II disposal wells as well as injection for enhanced oil recovery, i.e., water flooding and steam flooding.

Data Source: DOGGR Monthly Production database

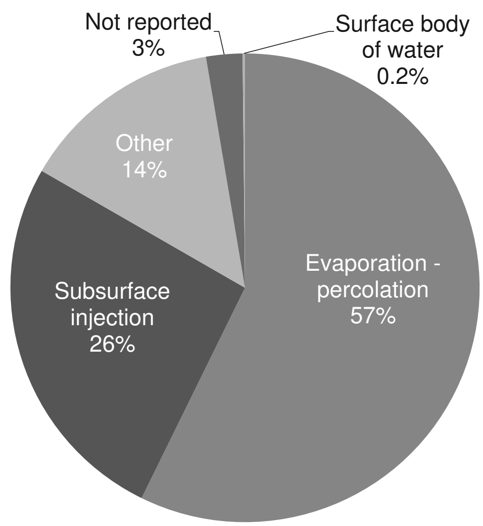


Figure 2.5-6. Produced water disposition during the first full month after stimulation. Data for stimulated wells throughout California were evaluated for the time period 2011-2014. Data from the DOGGR Monthly Production database.

Note: Subsurface injection includes injection into Class II disposal wells as well as injection for enhanced oil recovery, i.e., water flooding and steam flooding.

Between January 2011 and June 2014, these 1,657 stimulated wells generated a total of 1.3 million m^3 (1,000 acre-feet) of produced water during the first full month following stimulation. Evaporation-percolation in unlined surface impoundments (also referred to as percolation pits, ponds, or sumps) was reported to be the most common disposition method for these stimulated wells. According to California records, nearly 60% of the produced water from stimulated wells, or 720,000 m^3 (580 acre-feet), was disposed to unlined pits for evaporation and percolation during the first full month after stimulation. While produced water disposal in percolation pits has been reported in several California counties (e.g., Fresno, Monterey, and Tulare counties), disposal of produced water from stimulated wells in percolation pits was limited to Kern County and was associated with wells in Elk Hills (65%), South Belridge (27%), North Belridge (5.5%), Lost Hills (2.5%), and Buena Vista (<1%) (Table 2.5-4). Overall, use of percolation pits is common in production areas where well stimulation is applied and an estimated 40% of all produced water from stimulated oil pools is discharged to percolation pits for disposal. There were no reports of discharge to lined surface impoundments for evaporation only as a disposal method.

It is of note that operators have suggested that the information supplied to DOGGR specifying disposal practices for produced water may not be accurate. Chevron, for example, says that it ceased disposing produced water from its Lost Hills operation in unlined pits in 2008 (Appendix 2.E), although DOGGR records indicate this practice was continuing in 2014. Likewise, Occidental Petroleum (now California Energy Resources) says it has used subsurface injection for all produced water in Elk Hills (Nelson, 2014, personal communication). Our analysis is reliant on official data reported to DOGGR, which shows that these and other operators sent the majority of their produced water to unlined pits for evaporation and percolation, but the reports from industry suggest that more produced water may be disposed of in injection wells and less to percolation pits now, than in the past. Further investigation is needed to substantiate current wastewater management practices—particularly in relation to produced water from stimulated wells that may contain hydraulic fracturing fluids—and determine legacy effects from past disposal practices.

Table 2.5-4. Produced water disposition by evaporation-percolation during the first full month after stimulation by field, January 1, 2011 – June 20, 2104. Data from the DOGGR Monthly Production database.

Field	**Water volume (m^3)**	**Water volume (acre-feet)**	**Percent**
Elk Hills	460,000	380	65%
South Belridge	190,000	160	27%
North Belridge	39,000	32	5.5%
Lost Hills	18,000	14	2.5%
Buena Vista	2,000	2	0.27%
Total	**720,000**	**580**	**100%**

Note: All figures rounded to two significant figures. Numbers may not add up due to rounding.

Subsurface injection into Class II wells is the second most commonly reported disposition method for stimulated wells in California. Class II wells include saltwater disposal wells, enhanced recovery wells, and hydrocarbon storage wells (U.S. EPA, 2014). With enhanced oil recovery, reinjection of produced water serves multiple purposes, including enhancing product recovery, preventing subsidence, and disposing of produced water generated during production. About one-quarter of the produced water from stimulated wells, or about 330,000 m^3 (260 acre-feet), was injected into Class II wells for disposal or enhanced recovery (Table 2.5-3, Figure 2.5-6). While much of this occurred in Kern County, subsurface injection was the only disposition method reported in several counties, including Colusa, Fresno, Glenn, Ventura, and Orange County (Table 2.5-5).

Table 2.5-5. Produced water disposition by subsurface injection during the first full month after stimulation, by county, January 1, 2011 – June 30, 2014. Data from the DOGGR Monthly Production database.

County	Water volume (m^3)	Water volume (acre-feet)	Percent
Colusa	47	0.04	0.014%
Fresno	1,900	2	0.59%
Glenn	7.6	0.01	0.0023%
Kern	270,000	216	82%
Los Angeles Offshore	52,000	42	16%
Orange	1,700	1	0.52%
Sutter	430	0	0.13%
Ventura	3,500	3	1.1%
Total	**330,000**	**260**	**100%**

Note: All figures rounded to two significant figures. Numbers may not add up due to rounding.

As shown in Table 2.5-3, very few operators discharge produced water from stimulated wells into creeks or streams, with only two wells reported to be discharging a total of 2,100 m^3 (1.7 acre-feet) of produced water into surface water bodies during the first full month following stimulation. There were no reports of produced water from stimulated wells being disposed of in sewer systems.

The disposition method for 17% of the produced water from stimulated wells is either not known or not reported. “Other” was the third most common disposition method reported by operators—accounting for 14% of the produced water from stimulated wells. Similarly, the disposition method for 3% of the produced water was not reported. DOGGR staff confirmed that some operators are using the “other” category to describe disposition that is, in fact, included in some of the other categories, e.g., subsurface injection, surface body of water, sewer disposal, etc. (Fields, 2014). Some disposition methods, however, are not explicitly covered in these categories, such as reuse for irrigation, well stimulation, or other beneficial purposes, although there is anecdotal evidence that reuse for these purposes is occurring in California (for more information, see Section 2.6). These results suggest a need to improve data collection and better understand wastewater management practices in California.

2.5.3.3.2. Produced water from Offshore Oil and Gas Operations

California has four offshore oil platforms (Esther, Eva, Emmy, and Holly) and several man-made islands (Long Beach Unit, Rincon Island) operating in state waters. Well stimulation operations have been reported on Platforms Esther, Eva, and on the Long Beach Unit (THUMS Islands). There are also 23 oil platforms operating in federal waters off the coast of California, of which well stimulation operations have been reported on Platforms Gail, Gilda, and Hidalgo. Well stimulation accounts for a small fraction of offshore oil and gas production. It is estimated that approximately 12 hydraulic fracturing operations occur per year in state waters, and less than 10% of wells are hydraulically fractured in federal waters (Volume I, Chapter 3).

Options for the management and treatment of produced water on offshore oil platforms and islands are limited by treatment technology footprint, transportation costs, storage capacity, effluent limitations, and disposal options. Operations in state waters typically treat produced water to meet requirements for re-injection for enhanced oil recovery, and operations in federal waters treat produced water for discharge. Permitted disposal options vary as platforms located in federal waters are regulated under a general NPDES permit issued by U.S. Environmental Protection Agency (U.S. EPA), Region 9 (U.S. EPA, 2013a), while operations in California state waters are regulated under individual NPDES permits issued by regional water quality control boards.

On Platforms Esther and Eva, oil, gas, and produced water are separated using three-phase separators. The produced water then goes through a series of treatment processes to remove residual oil and suspended solids (California State Lands Commission, 2010a; 2010b).[2] Once treated, produced water is typically re-injected into the producing formation for enhanced oil recovery. On the Long Beach Unit, a portion of the produced water is reused as base fluid for well stimulation (Garner, 2014, personal communication).

Platforms operating in federal waters off the coast of California are permitted to discharge produced water that has been treated, as stipulated under a general NPDES permit.[3] When well stimulation fluids co-mingle with produced water, the mixture is managed, treated, and discharged according to produced water stipulations. Each of the 23 platforms has a maximum annual allowable produced water discharge volume, which ranges from

2. There is no evidence of a separate treatment system for managing wastewaters from well stimulation operations on Platform Esther. It is expected that wastewaters from well stimulation operations on Platforms Esther and Eva are subject to the same treatment processes and fate as produced water.

3. NPDES permit No. CAG280000

0.25 million to 8.9 million m^3 (206 to 7,192 acre-feet) per platform (U.S. EPA, 2013a). Platforms Gail and Hidalgo are allowed to discharge 0.7 million m^3 (560 acre-feet) and 2.9 million m^3 (2,350 acre-feet), respectively. Platform Gilda's discharge allowance is combined with that for Platform Gina at 4 million m^3 (3,300 acre-feet).

For a permitted discharge, oil and grease levels are measured weekly and must be lower than 29 mg L^{-1} monthly average and 42 mg L^{-1} daily maximum in discharged wastewater, according to effluent limitations in Subpart A of 40 CFR Part 435 in the Clean Water Act. The permit does not allow discharge of free oil, where free oil is defined as oil which will cause a film, sheen, or discoloration to the water surface upon discharge (U.S. EPA, 2014). Fourteen platforms, including Platforms Gail, Gilda, and Hidalgo, have specific monitoring and effluent requirements for produced water discharge, with measurements typically occurring on an annual or monthly basis. Platforms Gail, Gilda, and Hidalgo must also monitor for various aromatic hydrocarbons, but only have effluent limits for undissociated sulfide. All other platforms must monitor 26 constituents of concern.[4] These data are submitted to the EPA. The number of constituents sampled is based on previous studies where constituents present at concentrations above or near the water quality standards were identified and listed in the permits (U.S. EPA, 2013b). Sampling frequency depends on the frequency of discharge; however, constituents must be sampled "at least once during the last two years" of the permit (U.S. EPA, 2013a). Discharges are not monitored for constituents specific to or indicative of hydraulic fracturing, and the timing of sampling is unlikely to coincide with or measure any potential impacts from well stimulation treatments.

2.6. Contaminant Release Mechanisms, Transport Pathways, and Driving Forces

2.6.1. Overview of Contaminant Release Pathways

Well stimulation and associated activities can result in the release of contaminants into the environment, including into surface water and groundwater resources. Releases can occur during chemical transport, storage, mixing, well stimulation, well operation and production, and wastewater storage, treatment, and disposal. The term "release mechanism" refers to the way in which a contaminant migrates from its intended containment (natural or manmade) into the surrounding environment. Once released, contaminants can be transported through various mechanisms (e.g., percolation into soil, transport into groundwater, runoff to local streams) or transformed through physical, chemical, and biological processes. A *physical connection*, either natural or induced, between the release location and the impacted surface or groundwater body is referred to as a "transport pathway." A *driving force* (e.g., differences in hydraulic head or pressure) is required for contaminant migration into the connected surface or groundwater body.

4. Where the California Ocean Plan also contains criteria for a select constituent, then the more stringent of the two is used, as the California Ocean Plan can regulate "discharge outside the territorial waters of the State [that] could affect the quality of the waters of the State" (SWRCB, 2012).

The extent to which water resources are affected by releases of well stimulation chemicals or wastewaters depends on the amount and type of contaminant(s) released, existence of transport pathways and corresponding driving forces, and the transformations occurring during transport. Other factors that impact the probability of contaminant migration include reservoir depth, physical and hydrological properties of the formation, production strategies, drilling and casing practices, and the unique geologies of each oil and gas-producing region.

Release mechanisms and transport pathways can occur at the surface or in the subsurface, and are associated with a variety of activities during the production process (e.g., well stimulation, wastewater management and disposal, and well operation). Surface releases are typically easier to identify and associate with a particular activity. Subsurface releases are generally more difficult to detect, associate with a particular release mechanism, and mitigate. Reservoir and stimulation fluids can migrate through the subsurface if (1) surface releases eventually percolate into groundwater; (2) produced water is directly injected into protected groundwater; or (3) if transport pathways out of the reservoir being fractured (out-of-zone) have been created through stimulation operations, either through direct fracturing into overlying aquifers or via out-of-zone connection to a preexisting pathway (e.g., a preexisting fracture network, a fault, or some other permeable feature). While transport through preexisting or induced subsurface pathways has been documented in conventional oil and gas operations, it is not known whether stimulation increases the frequency of occurrence of such pathways. Regardless of the uncertainty whether stimulation increases the frequency of leakage pathways, stimulation introduces a new set of water quality concerns for leakage, through pathways documented from conventional oil and gas operations, due to the use of stimulation chemicals and the commingling of produced water and returned stimulation fluids.

2.6.2. Potential Release Mechanisms to Water in California

In this section, we identify potential release mechanisms specific to well stimulation activities that can (1) form transport pathways (natural, induced, or a combination) to water resources and (2) allow stimulation or reservoir fluids to migrate into water resources if the appropriate driving forces are present. We examined several plausible release mechanisms for surface and groundwater contamination associated with onshore well stimulation, based on an exhaustive literature review of release events and hazards that have been reported in the U.S. (Table 2.6-1). While release mechanisms and transport pathways that occur during post-stimulation and wastewater management apply to all oil and gas development in California, they are relevant to stimulated wells because produced water from stimulated wells may contain hazardous chemicals from well stimulation fluids.

Table 2.6-1. Activities and associated release mechanisms for the different stages of well stimulation

Activities	**Release Mechanisms and Transport Pathways**	**Releases**
Preparation: Site development, well drilling, construction and completion	• Erosion and surface runoff* • Well blowout resulting from failure to control well pressure and improper well installation* • Release of drilling fluids and waste during handling, storage and disposal* • Migration through existing or induced pathways or other subsurface features (such as faults, fractures, or permeable adjacent formations)*	• Soil/particulate matter in stormwater runoff • Drilling fluids and wastes • Oil and gas • Formation water
Well stimulation	• Transportation accident • Equipment failure • Leakage from onsite chemical storage • Spills during chemical mixing • Pipe failure (both above and below ground) • Well failure due to stimulation • Problems related to drilling, completion, or well design errors (e.g., poor cementing, wrong perforation depth) • Migration via other pathways intercepted by fractures (including plugged, deteriorated, or abandoned wells) • Fractures or other permeable pathways directly intercepting groundwater resources	• Additives • Stimulation fluids • Oil and gas • Formation water
Post-stimulation: Well cleanout and production	• Pipe failure (both above and below ground) • Well failure due to drilling, completion or well design errors (e.g., leakage through compromised casing and cement) • Migration via other pathways intercepted by fractures (including plugged, deteriorated, or abandoned wells, faults, fractures, permeable adjacent formations)	• Well cleanout fluids • Wastewaters • Oil and gas • Formation water
Wastewater management: Handling, storage, reuse, and disposal	• Spills and leaks during storage and handling • Transportation accident • Pipe failure (both above and below ground) • Overflow from storage reservoir • Percolation (from storage or disposal pits) • Reuse of produced water for beneficial purposes (e.g., irrigation) • Disposal of produced water into sewer system (and subsequent disposal of treatment residuals) • Improper siting of disposal wells (into aquifer or protected groundwater) • Failure of disposal well (e.g., leakage through casing or cement) • Migration through existing pathways during subsurface disposal (e.g., faults, fractures, permeable overburden) • Illegal discharge	• Wastewaters • Oil and gas • Treatment residuals (including disinfection byproducts)

*Note: * Release mechanisms that are not within the scope of this assessment since they are part of routine oil and gas development and there are no unique impacts associated with well stimulation. While release mechanisms and transport pathways that occur during post-stimulation and wastewater management apply to all oil and gas development in California, they are of particular relevance for stimulated wells (and are included in this study) because (1) produced water from stimulated wells may contain returned stimulation fluids, and (2) the quality of formation water from stimulated reservoirs may differ from that of conventional reservoirs.*

We narrowed the broad set of possible release mechanisms to a subset that is most relevant for California (Figures 2.6-1 and 2.6-2, Table 2.6-2). In the following sections, we list several incidents of contamination that have occurred in California or other oil and gas producing regions, to show that these release mechanisms are viable, and relevant for California.

The California-specific release mechanisms are classified as normal, accidental, and intentional (Table 2.6-2). "Normal" release mechanisms result from practices that are part of routine operations in the California oil and gas industry, and include disposal of produced water in unlined pits, injection of produced water into potentially protected groundwater, reuse of produced water for irrigation, and disposal of produced water in sewer systems. "Accidental" release mechanisms can be of several types, including errors in design and execution of the stimulation operation—such as out-of-zone fracturing, leakage through degraded or impaired wells, leakage through natural subsurface features, surface spills and leaks, or consequences of natural disasters such as earthquakes and floods. It should be noted that in California, where fracturing depths are much shallower than in other parts of the country, fractures induced by hydraulic fracturing could potentially form direct transport pathways to groundwater. Nationally, several incidents have been caused by leakage through degraded abandoned wells and leakage of stray gas from production or other wells into groundwater. Surface releases caused by spills or leaks have been conclusively linked to stimulation operations. "Intentional" release mechanisms are unauthorized or unpermitted releases such as illegal discharges.

Finally, we assigned a priority for each release mechanism based on the release type (e.g., all releases that are part of normal operations are considered high priority), and direct or indirect evidence indicating their likelihood of occurrence in California (Table 2.6-2). We focus on release mechanisms and transport pathways from hydraulic fracturing operations, and assume that this covers concerns associated with matrix acidizing operations, given that the latter follow a similar process as hydraulic fracturing operations, albeit using less equipment, lower injection pressures, and no proppant (Volume I, Chapter 2).

Table 2.6-2. Assessment of release mechanisms associated with stimulation operations for their potential to impact surface and groundwater quality in California. Considerations for the priority ranking include whether the releases occur due to activities that are part of normal operations, and the likelihood of the occurrence in California. References for this table are provided in the text.

Release Mechanism	**Release type***	**Has occurred in California?**	**Has occurred in other places?**	**Evidence associating release to hydraulic fracturing?**	**Priority**
Percolation of produced water from unlined pits	Normal	Yes	Yes	Yes	High
Injection of recovered fluids and produced water into potentially protected groundwater via Class II wells	Normal	Yes	No	Unknown	High
Reuse of produced water for irrigation	Normal	Yes	No	Unknown but likely	High
Disposal of produced water in sewer systems	Normal	Yes	Yes	Unknown in California, yes in other states	High
Leakage through hydraulically induced fractures	Accidental	Unknown	Unknown	Unknown	Medium
Leakage through failed inactive wells (abandoned, buried, idle or orphaned)	Accidental	Unknown	Yes	Unknown	Medium
Leakage through active wells (production, disposal or other wells)	Accidental	Unknown	Yes	Yes	Medium
Leakage through other subsurface pathways (preexisting natural fractures, faults, or other permeable features)	Accidental	Yes	Unknown	Unknown	Medium
Surface and near-surface spills and leaks	Accidental	Yes	Yes	Unknown	Medium
Operator error	Accidental	Unknown	Yes	None in California, yes elsewhere	Low
Illegal discharges of wastewater from oil and gas operations	Intentional	Yes	Yes	Yes	Low

**The type of activity leading to the release. Categories are*

Normal: Activity is part of normal operations, and release occurs by design.

Accidental: Release was caused due to an accident, but can be prevented by following proper design and protocols

Intentional: Release was intentional despite being unauthorized, and can be prevented by proper oversight and monitoring.

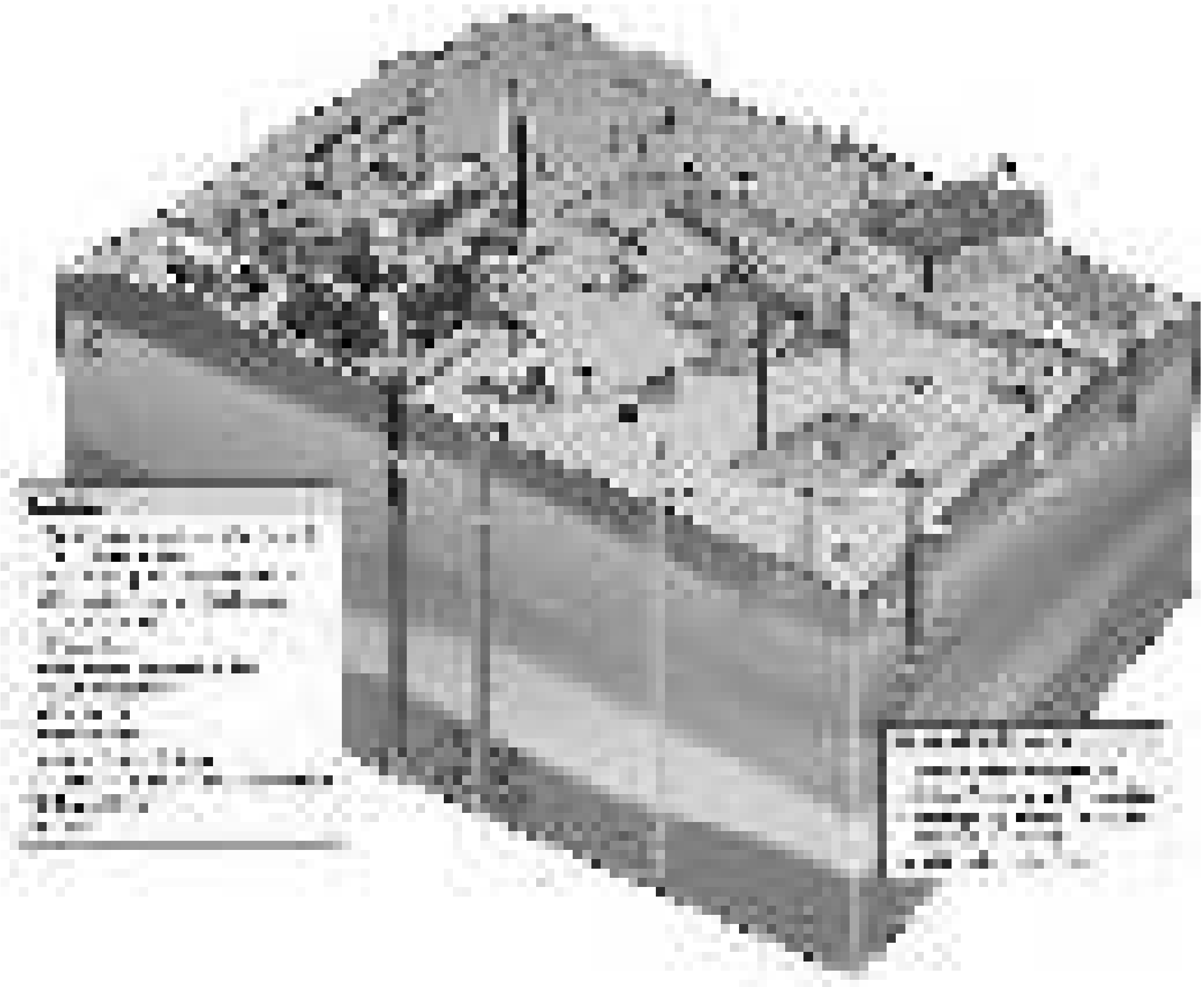


Figure 2.6-1. Potential contaminant release mechanisms that originate at the surface related to stimulation, production, and wastewater management and disposal activities in California. The diagram is not drawn to scale.

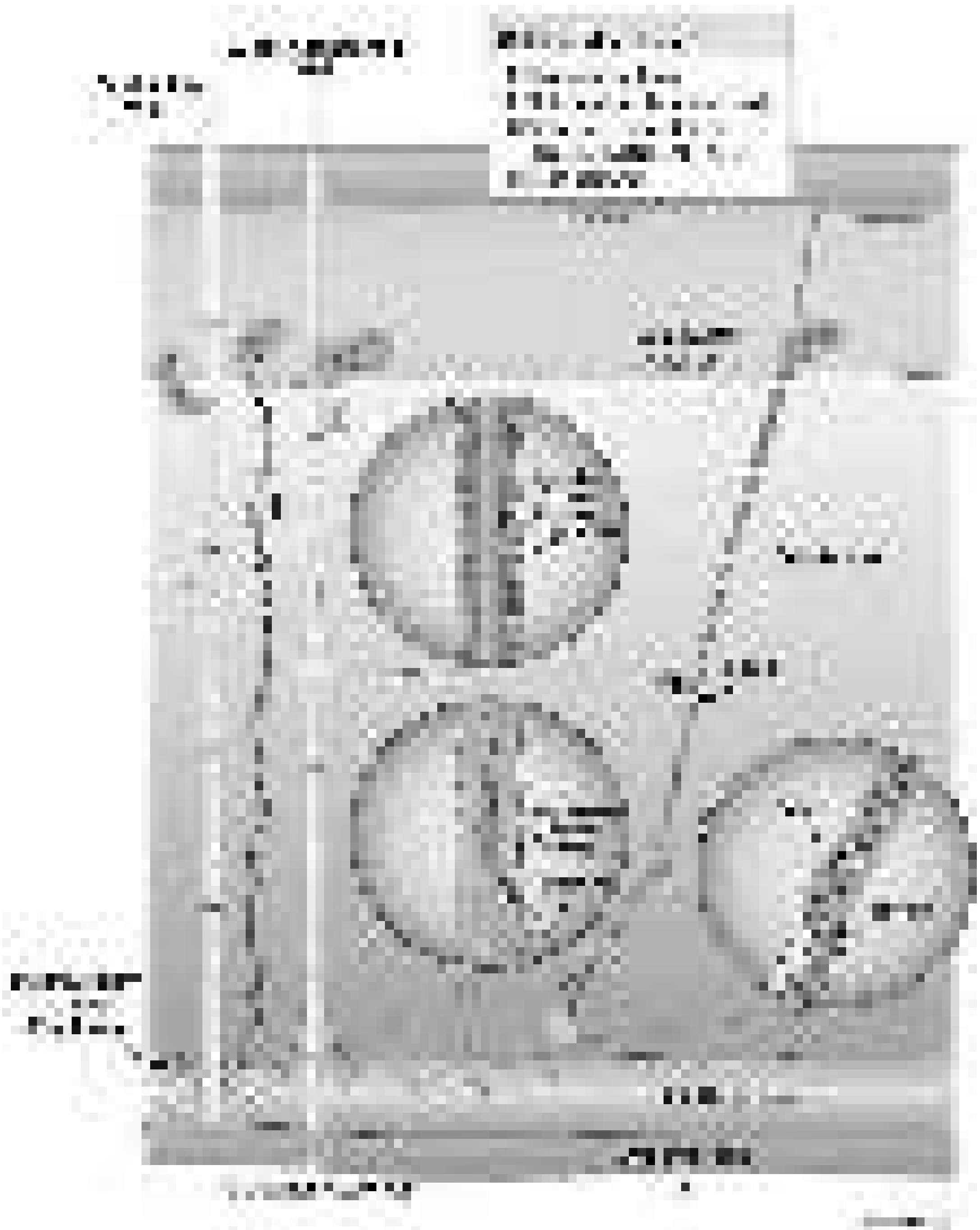


Figure 2.6-2. Potential release mechanisms and transport pathways in California that could originate in the subsurface. These include leakage through failed (production, abandoned or disposal) wells, migration through intercepted fractures and fault activation. The diagram is not drawn to scale.

2.6.2.1. Use of Unlined Pits for Produced Water Disposal

As described above, evaporation-percolation in unlined surface impoundments (percolation pits) is a common disposition method for produced water from stimulated wells in California (Section 2.5). Because the primary intent of unlined pits is to percolate water into the ground, this practice provides a direct pathway for the transport of produced water constituents, including returned stimulation fluids, into groundwater. Some states, including Kentucky, Texas, and Ohio, have phased out the use of unlined pits for disposal (Kell, 2011; 401 KAR 5:090 Section 9(5)(b)(1)).

The state's nine Regional Water Quality Control Boards have primary authority to regulate disposal pits in California.[5] Most of the instances of discharge into percolation pits occurred in the region under the authority of the Central Valley Regional Water Quality Control Board. Within that region, disposal of produced water in percolation pits overlying groundwater with existing and future beneficial uses has been allowed if the wastewater meets certain salinity, chloride, and boron thresholds.[6] Produced water that exceeds the salinity thresholds may also be discharged in "unlined sumps, stream channels, or surface water if the discharger successfully demonstrates to the Regional Water Board in a public hearing that the proposed discharge will not substantially affect water quality nor cause a violation of water quality objectives" (CVRWQCB, 2004). There was previously no testing required, nor thresholds specified, for other contaminants, including chemicals used for well stimulation or other routine oilfield activities. The Central Valley Regional Water Quality Control Board implemented an order on April 1, 2015 requiring operators to conduct a chemical analysis of wastewater disposed in active produced water disposal ponds in the Central Valley; however, the list of constituents to be analyzed does not include any indicators for stimulation fluid constituents (CVWQCB, 2015).

Figure 2.6-3 shows active and inactive unlined pits and ponds in the Central Valley and along the Central Coast. Presumably, the pits are largely used to deliberately percolate wastewater for the purpose of disposal. Active pits are primarily found on the east and west side of the southern San Joaquin Valley, although a small number of active pits can also be found in Monterey and Santa Barbara Counties. The Central Valley Regional Board is currently conducting an inventory of unlined pits in the Central Valley. As of April 2015, a total of 933 pits have been identified, of which 62% are active and 38% are inactive. An estimated 36% of the active unlined pits are operating without the necessary permits from the Central Valley Regional Board (Holcomb, 2015). Central Valley Regional Board

5. Local Air Districts also regulate some aspects of oilfield pits, e.g., volatile organic carbon (VOC) emissions.

6. According to the Water Quality Control for the Tulare Basin, which was developed by the Central Valley Regional Water Quality Control Board, tdisposal of oil field wastewater in pits overlying groundwater with existing and future beneficial uses is permitted if the salinity of the wastewater is less than or equal to 1,000 micromhos per centimeter (μmhos cm^{-1}) electrical conductivity (EC), 200 milligrams per liter (mg L^{-1}) chlorides, and 1 mg L^{-1} boron (CVRWQCB 2004).

staff expects to issue 180 enforcement orders for facilities that are not permitted or are operating with outdated permits by the end of 2015. Cease and desist orders have been issued for some facilities operating with outdated permits (Holcomb, 2015). An analysis of groundwater quality near these pits can be found in Section 2.7.

There is not one centralized location for reporting and tracking locations of unlined or disposal pits in California, so any list of disposal pits must be considered approximate. The Central Valley Regional Board, which recently launched an investigation into unlined pits, found that more than one-third of the pits located in their jurisdiction were functioning without the proper permits, indicating that there may be additional pits of which the state is unaware (Holcomb, 2015). The DOGGR production database indicates that produced water is sent to evaporation-percolation disposal ponds in counties where there are no reported pit locations, suggesting that there may be unreported pits in those counties. For example, according to the production database, 47,000 m^3 (38 acre-feet) were sent to evaporation-percolation ponds in Ventura County in 2013 (DOGGR, 2014c), despite there being no reported pit locations within or near the borders of that county (Holcomb, 2015).

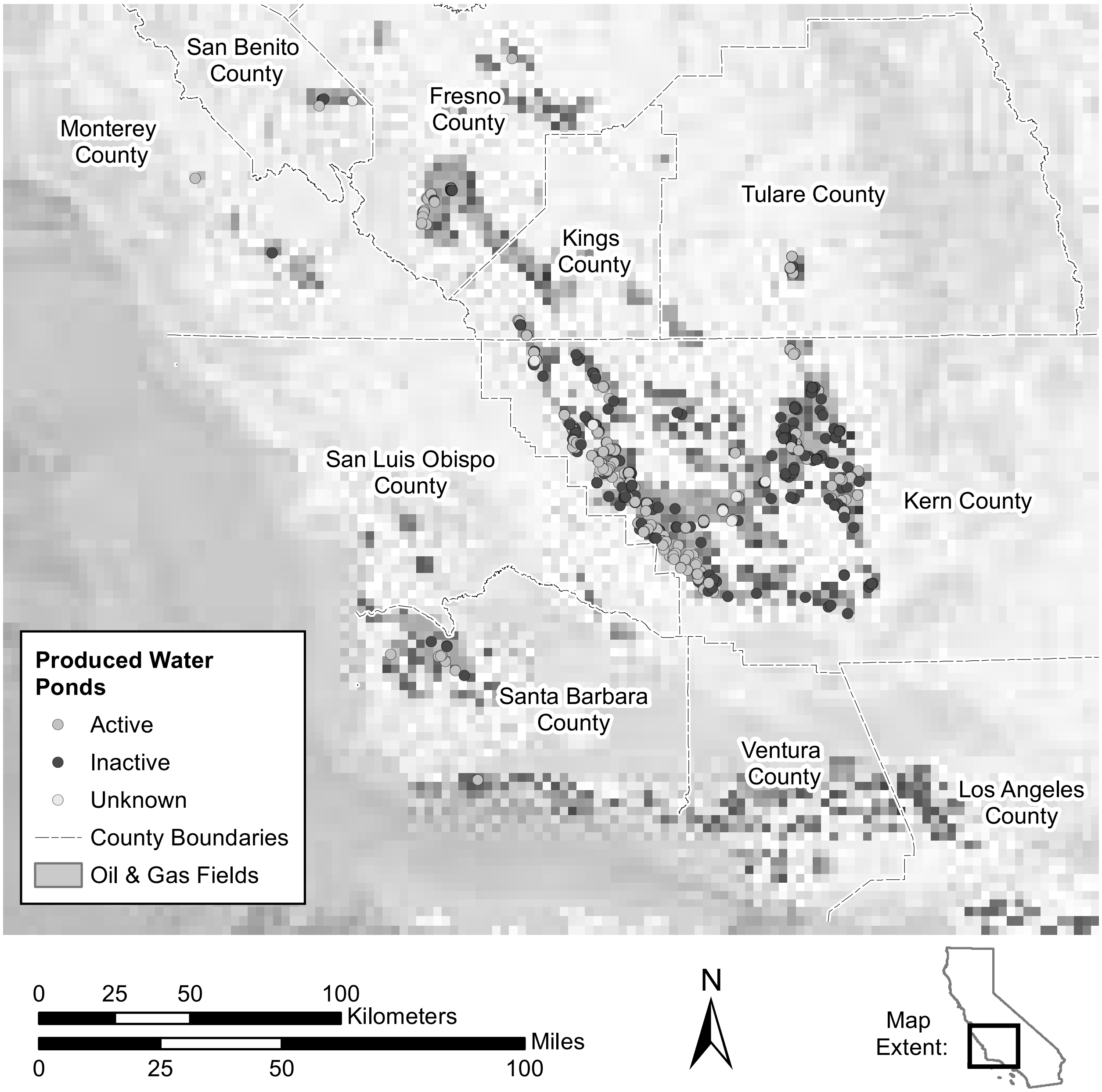


Figure 2.6-3. Unlined pits used for produced water disposal in the Central Valley and the Central Coast, 2015. Data from CVRWQCB 2015; Borkovich 2015a; 2015b (Appendix 2.G).

There is ample evidence of groundwater contamination from percolation pits in California and other states (e.g. CVRWQCB, 2015; Holcomb, 2015; Kell 2011). For example, in California, the Central Valley Regional Water Quality Control Board determined that several percolation pits in Lost Hills and North and South Belridge had impacted groundwater, and ordered their closure (CVRWQCB, 2015). In these cases, monitored natural attenuation rather than active remediation was selected as the method for corrective action for improving the groundwater quality. Groundwater contamination has also been associated with unlined pits in other states. Kell (2011) reviewed incidents of groundwater contamination caused by oil field activities in Texas between 1993 and 2008 and in Ohio between 1983 and 2007. Of the 211 incidents in Texas over the 16-year study period, 27% were associated with unlined infiltration pits, which have been phased out in Texas starting in 1969 (Kell, 2011). Of the 185 groundwater-contamination incidents in Ohio over a 25-year period, 5% (or 10 incidents) were associated with the failure of unlined pits. Like Texas, unlined disposal pits are no longer used in Ohio, and no incidents have been reported since the mid-1980s (Kell, 2011). While these studies and others linking wastewater percolation and unlined pits to groundwater contamination are not specific to well stimulation fluids, they are illustrative of the implications of this disposal method. Moreover, the presence of stimulation fluids in the produced water is likely to increase the risk of groundwater contamination.

A case in Pavillion, WY, raises additional concerns about the use of unlined pits for produced water disposal. According to the U.S. EPA draft report, released in 2011, high concentrations of hydraulic fracturing chemicals found in shallow monitoring wells near surface pits "indicate that pits represent a source of shallow ground water contamination in the area" (Digiulio et al., 2011). At least 33 unlined pits were used to store or dispose of drilling muds, flowback, and produced water in the area. Neither the company responsible for the natural gas wells, nor the other stakeholders contested these findings (Folger et al., 2012).

2.6.2.2. Injection of Produced Water into Protected Groundwater via Class II Wells

Subsurface injection was the second most common disposal method for produced water from stimulated wells (Section 2.5). Studies show that with proper siting, construction, and maintenance, subsurface injection is less likely to result in groundwater contamination than disposal in unlined surface impoundments (Kell, 2011). However, there are significant concerns about whether California's Class II underground injection control (UIC) program is adequately protective of underground sources of drinking water (USDWs) – defined as groundwater aquifers that are used for water supply or could one day supply water for human consumption.[7]

In 2011, at the request of EPA Region 9, an independent consultant reviewed California's UIC Program and found inconsistencies in how USDWs are defined (Walker, 2011). Specifically, the DOGGR program description refers to the protection of freshwater containing 3,000 mg L^{-1} or less TDS. Current federal regulation, however, defines USDWs as containing less than 10,000 mg L^{-1} TDS. This suggests that USDWs in California containing between 3,000 and 10,000 mg L^{-1} TDS are not adequately protected. More recently, DOGGR acknowledged that it has approved UIC projects in zones with aquifers

7. The UIC program was developed as a result of the 1974 Safe Drinking Water Act and was intended to protect USDWs.

lacking exemptions, even though those zones would likely qualify for an exemption under current regulations.[8] Additionally, new information has indicated that, for several decades, injection activities have been allowed in 11 other aquifers that were thought to be exempt; however, the geologic basis for those exemptions is “now in question” (Bohlen and Bishop, 2015).

In response to these issues, DOGGR is reviewing more than 30,000 of the state’s 50,000 Class II wells, and is expected to complete that review in early 2016. Given their mutual role in protecting water resources, DOGGR and the State Water Board are working together on this review. In 2014, DOGGR ordered the immediate closure of 11 disposal wells in Kern County that potentially present health or environmental risks, and State Water Board staff identified 108 water supply wells located within a one-mile radius of these wells.[9] Subsequent sampling found no sign of contamination from oil and gas operations (SWRCB, 2014b). Currently, 140 active wells are under immediate review by the State Water Board, because they are operating in aquifers that lack hydrocarbons and contain water with less than 3,000 mg L^{-1} TDS. These wells are being reviewed for “proximity to water supply wells or any other indication of risk of impact to drinking water and other beneficial uses” (Bohlen and Bishop, 2015). The State Water Board is reviewing 150 injection wells per month and expects to be done with its review in May 2015. Going forward, DOGGR has proposed a schedule and process to the U.S. EPA to bring California’s UIC program into compliance with federal regulations. Further analysis on this subject can be found in Volume III, Chapter 5.

2.6.2.3. Reuse of Produced Water for Irrigated Agriculture

Produced water is commonly reused for beneficial purposes, including steam flooding, irrigation, and industrial cooling. In some cases, the produced water is treated prior to reuse, but in others it is simply blended with freshwater to bring the levels of salts and other constituents down to an acceptable range. In California, in in particular the San Joaquin Valley, there is growing interest in expanding the beneficial reuse of produced water for agriculture, particularly for irrigation, due to the co-location of oil, gas, and agricultural operations and ongoing water scarcity concerns in these areas. The use of produced water from unconventional production raises specific or unique concerns, because of the variety of chemicals used during well stimulation that may end up mingled with produced water and the unknowns concerning the toxicity and environmental profile of those chemicals (discussed in the characterization of chemicals section, above).

8. An “exempt aquifer” is an aquifer that meets the criteria for protection but that protection has been waived because it is not currently being used — and will not be used in the future — as a drinking water source, or it is not reasonably expected to supply a public water system due to a high total dissolved solids content.

9. Since review, two of the 11 wastewater disposal wells have been authorized to resume operations.

It is not known if produced water from stimulated wells is or has been used for irrigation in California. According to data from the Central Valley Regional Board, there are currently five fields (Deer Creek, Jasmin, Kern River, Kern Front and Mount Poso) where produced water is reused to irrigate crops. Of these fields, well stimulations have only been reported in Kern River and Mount Poso. In Mount Poso, the last reported hydraulic fracture was in 2003. Although hydraulic fracturing was reported as recently as 2014 in the Kern River, only three hydraulic fracturing operations have been reported since 2012.

Produced water from the Kern River oil field irrigates the Cawelo Water District, a service area covering 182 km^2 (45,000 acres), of which roughly 82% of crops are permanent crops, including citrus, nuts, and grapes (Cawelo Water District, 2014). The water is treated at the Kern Front No. 2 Treatment Plant before it is delivered to the water district (CVRWQCB, 2012). The Cawelo Water District sets water quality goals that comply with requirements established by the CVRWQCB in the Tulare Lake Basin Plan. However, these requirements do not include monitoring for constituents specific to, or indicative of, hydraulic fracturing (CVRWQCB, 2012).

2.6.2.4. Treatment and Reuse of Oil and Gas Industry Wastewater

Comprehensive data on current practices applied in California for the treatment of produced water before beneficial reuse are not available. However, in general, the treatment of produced water has been the subject of intensive investigation and standard treatment practices have evolved for the reuse of produced water (e.g., Federal Remediation Technologies Roundtable, 2007). Treatment of constituents commonly found in produced water (e.g., oil and grease, dissolved solids, suspended particles, bacteria, etc.) is generally well documented (Arthur et al., 2005; Drewes, 2009; Fakhru'l-Razi et al., 2009; Igunnu and Chen, 2012; M-I SWACO, 2012). We are unaware of any studies that examine whether commonly used produced water treatment systems would effectively remove hydraulic fracturing chemicals (particularly organic chemicals) that might be found in produced water from stimulated wells.

We evaluated the potential effectiveness of various chemical, physical, and biological treatment technologies commonly used for produced water treatment in California for removing well stimulation chemicals (Appendix 2-C). Results of the analysis indicate that there is no one treatment technology that can independently treat all categories of well-stimulation-fluid additives, but that treatment trains (systems of combined processes in series) could probably be developed to treat most stimulation chemicals known to be used in California. For example, the San Ardo Oil Field Water Management Facility, located in the upper Salinas Valley in Monterey County, treats produced water through several pretreatment processes, followed by a two-pass reverse osmosis (RO) system before use for environmental purposes and groundwater recharge (Figure 2.C-1)—whereas the Kern Front No. 2 Treatment Plant in northern Kern County treats produced water by gravity separation, followed by air flotation with coagulants and mechanical agitation for use in irrigation (Figure 2.C-2). Based on the analysis in Appendix C, the treatment train at

San Ardo would be expected to effectively remove all well stimulation chemicals from influent streams, while the Kern Front No. 2 Treatment Plant would not be expected to remove most chemicals associated with well stimulation operations. In summary, the most common simple treatment trains, for example oil separation followed by filtration, are not expected to be effective at removing most well stimulation chemicals, but more complex treatment trains, potentially including RO, may be effective.

Reuse of produced water for irrigated agriculture, groundwater recharge, or environmental flows is an attractive idea, especially in the face of drought. For a successful reuse program, it will be necessary to identify beneficial uses for reclaimed wastewater from oil and gas production, identify the water quality objectives to support that use, and identify what parameters of the produced waters exceed these water quality objectives. Treatment and reuse of produced water from fields with stimulated wells should consider the presence of well-stimulation-fluid chemicals and their breakdown products as part of this evaluation.

2.6.2.5. Disposal of Produced Water in Sanitary Sewer Systems

There is no evidence that produced water from stimulated wells in California is currently being disposed of in sanitary sewer systems. Statewide, however, an estimated 7 million m^3 and 4 million m^3 (5,700 and 3,200 acre-feet) of produced water was disposed of in sanitary sewer systems in 2012 and 2013, respectively, and some of this has occurred in fields where wells have been stimulated (e.g., Wilmington Oil Field in Los Angeles County and a small amount from the Lost Hills Oil Field and Midway-Sunset Oil Field in Kern County). Oil and gas well operators that discharge produced water into sanitary sewers are required by the sanitation districts to obtain pretreatment permits. Pretreatment of produced water is typically minimal—consisting primarily of oil and water separators, followed by clarification and sometimes air stripping or flotation—and does not remove most chemicals associated with well-stimulation operations.

Additionally, sewage treatment plants are not typically equipped to handle produced water, potentially disrupting the treatment process and discharging salt and other contaminants into the environment. In Pennsylvania, for example, the high salt content of oil and gas wastewater resulted in increased salt loading to Pennsylvania rivers (Brantley et al., 2014; Kargbo et al., 2010; Vidic et al., 2013; Wilson and VanBriesen, 2012). Ferrar et al. (2013) identified concentrations of some chemicals, including barium, strontium, bromides, chlorides, total dissolved solids, and benzene, in treated effluent that exceeded drinking water quality criteria. Similarly, Warner et al. (2013a) studied the effluent from a brine treatment facility in Pennsylvania and found that TDS from the effluent led to an increase in salts downstream, despite significant reduction in concentrations due to the treatment process and dilution from the river. Moreover, radium activities in the stream sediments near the point of discharge were 200 times higher than in upstream and background sediments, and were above radioactive waste disposal thresholds. State regulators in Pennsylvania subsequently discouraged the practice of discharging waters recovered from fracturing operations to sanitary sewer systems due to water quality concerns, although some discharge into these facilities has continued. Much of the research on disposal to these systems has focused on the produced water constituents and has not specifically addressed the fate of stimulation chemicals commingled with produced water.

2.6.2.6. Leakage through Hydraulic Fractures

One concern related to subsurface leakage through hydraulic fractures is the degree to which induced fractures may extend beyond the target formation to connect to overlying protected groundwater, or to other natural or man-made pathways such as faults, natural fractures, or abandoned wells. Many studies, which are discussed in detail below, reference stimulation activities conducted at significant depth, and thus it has been generally assumed that fractures cannot directly intercept groundwater resources. The situation in California is notably different, due to the shallow depths of fracturing (Volume I, Chapter 3). Additional data about fracture geometry and depths are starting to emerge from the well completion reports that are now being submitted to DOGGR by operators.

The completion reports have data for the horizontal and vertical extent of stimulation, which are reported as "Stimulation Length" and "Stimulation Height." For this assessment, we analyzed the reported stimulation length and height, and calculated the depth (from the surface) to the top of the stimulation using data reported for 499 hydraulic fracturing treatments from a total of 506 well completion reports that were available as of December 15, 2014. The depth from the surface to the top of the stimulation was calculated as:

$$\frac{\text{TVD Wellbore Start+TVD Wellbore End}}{2} - \frac{\text{Stimulation Height}}{2}$$

where "TVD Wellbore Start" and "TVD Wellbore End" refer to the true vertical depths at the top and bottom of the treatment interval in the well, respectively.

This calculation is based on the assumption that the reported stimulation geometries are accurate. It is also assumed that stimulation propagates equally in both vertical directions from the midpoint of the treatment interval, and so does not account for asymmetrical vertical growth relative to the well interval treated. We also assume that the midpoint of the stimulation height occurs at the midpoint of the true vertical depth of the treated wellbore interval. The original dataset had to be modified to create consistent data formats. Only hydraulic fracturing treatments were considered; data for the seven acid matrix treatments were excluded. The distribution of these depths is shown in Figure 2.6-4.

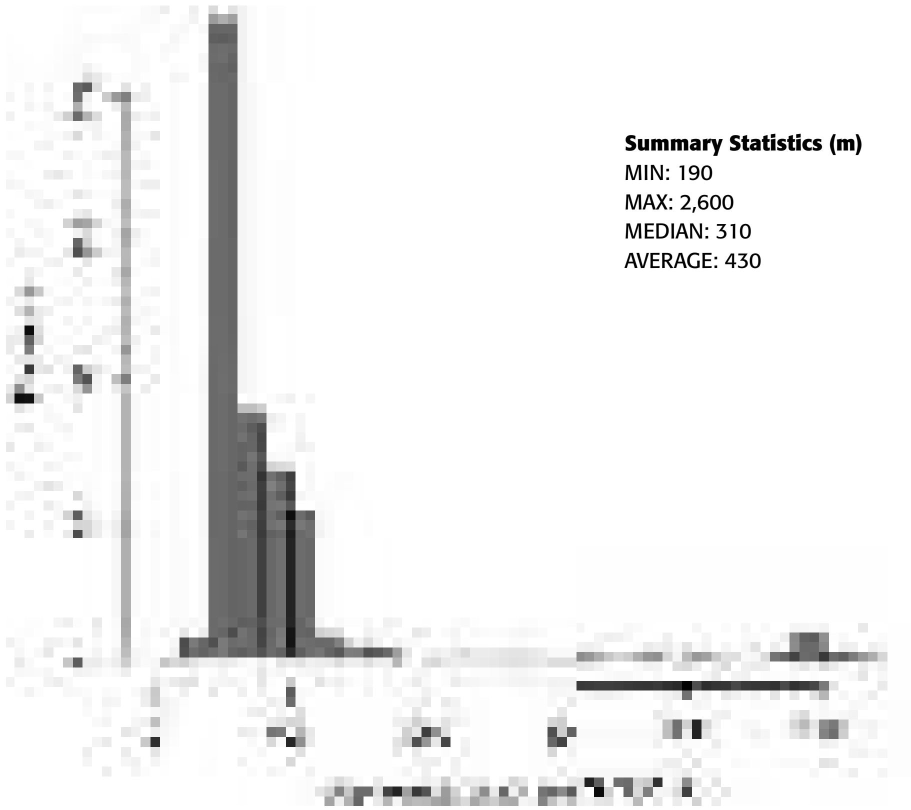
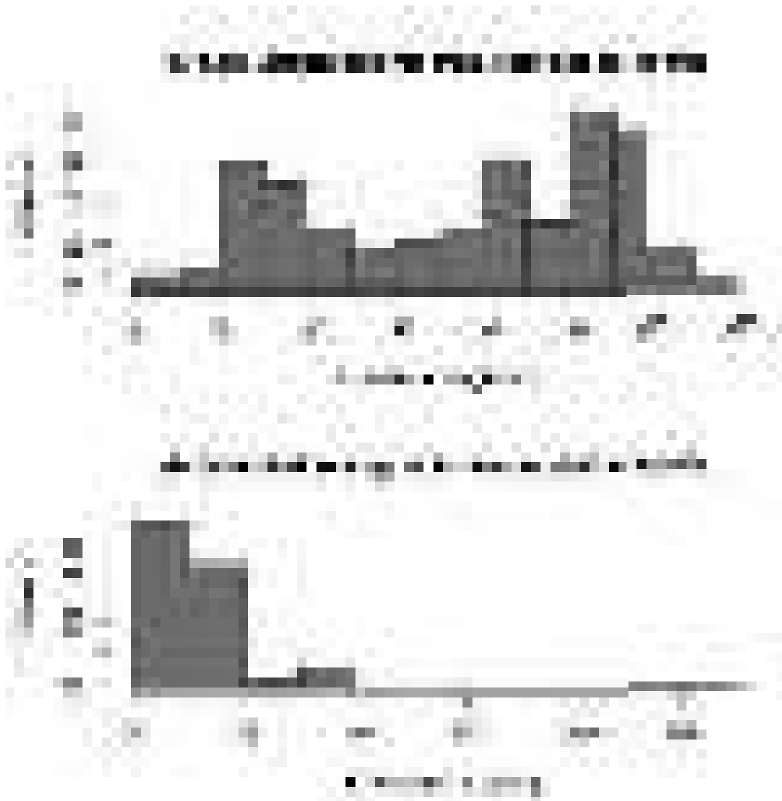


Figure 2.6-4. The approximate depth (from the surface) of the top of the hydraulic fracturing stimulations, (calculated by subtracting half the stimulation height from the midpoint of the wellbore treatment interval). Data source: Completion reports submitted to DOGGR as of Dec 15, 2014.

The data show that the true vertical depths to the top of the producing horizon in which the fracturing is induced are mostly shallow, ranging from 200 to 300 m (650 to 1,000 ft), and that in approximately half the operations, fracturing can extend to depths less than 300 m (1,000 ft) from the surface. This result is consistent with an earlier analysis that found the top of the fracturing interval in about half the operations to be less than 300 m (1,000 ft) deep (Volume I, Chapter 3). The shallow depths of fracturing raise concern about the possibility that out-of-zone fractures may directly intercept protected groundwater resources. Additional research is needed to determine how often this occurs, if at all, and the consequences if it does occur.

Most of the reported stimulation heights are between 50 m and 300 m (165 ft and 1,000 ft), while stimulation lengths in lateral directions are typically less than 50 m (165 ft) (Figure 2.6-5); however, the data for stimulation dimensions are inferred from unsubstantiated industry calculations. Based on the data submitted to DOGGR, it appears as though stimulations due to fracturing are oriented more vertically than horizontally (Figure 2.6-6).



Summary statistics (m)
MIN: 16
MAX: 336
MEDIAN: 211

Summary statistics (m)
MIN: 15
MAX: 266
MEDIAN: 23

Figure 2.6-5. Distribution of (a) stimulation heights and (b) stimulation lengths in California. Data source: Completion reports submitted to DOGGR as of December 15, 2014.

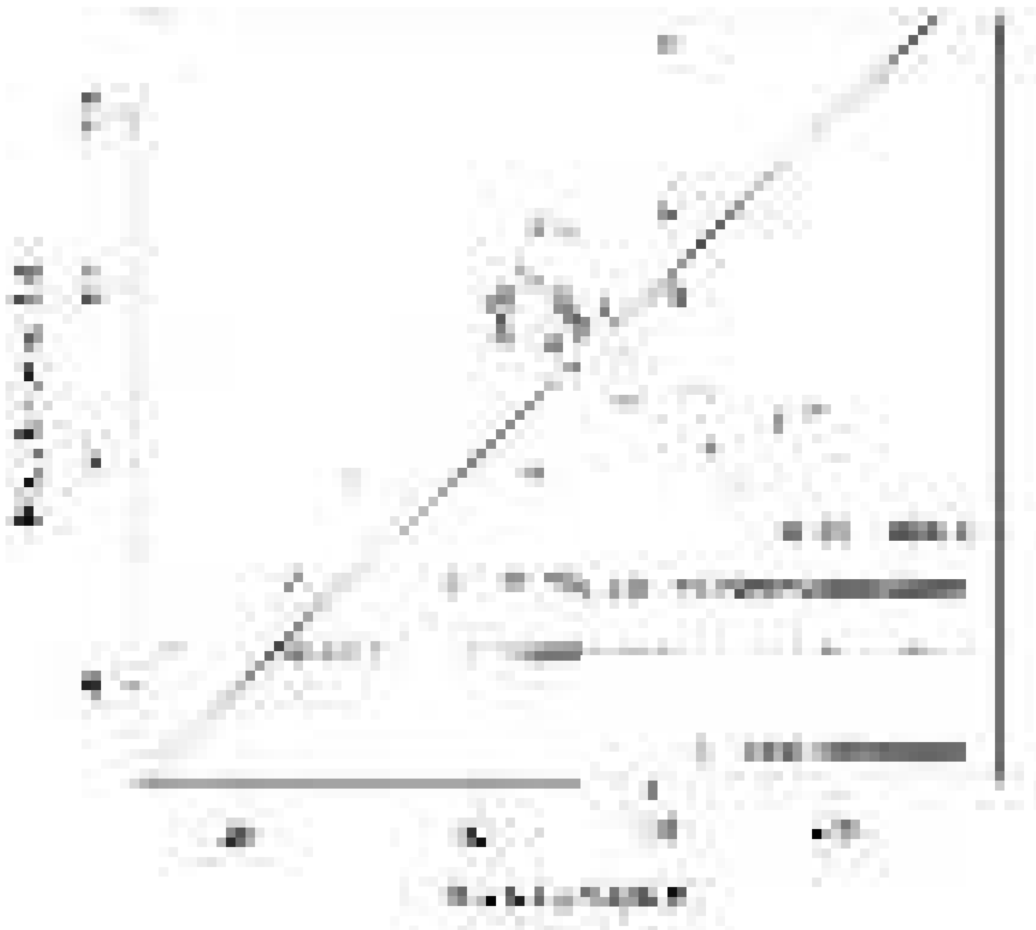


Figure 2.6-6. Comparison of stimulation heights with stimulation lengths for fracturing operations show that stimulations extend more vertically than horizontally. The solid line represents the 1:1 relationship; axes are in log scales. Data source: Completion reports submitted to DOGGR as of December 15, 2014.

The accuracy of the reported data on fracture geometries is unknown, given that operators do not report the methods for calculating the stimulation height and length. Furthermore, examination of hundreds of the well records that record hydraulic fracturing operations indicates operations consisting of only one stage are less than one quarter of all operations. However, all the completion reports indicate only one stage per well. It is unlikely such a substantial change in practice occurred at the same time that mandatory reporting commenced. It is more likely operators are reporting all fracturing stages within a well as one stimulation, and misreporting the number of stages in the well. Consequently, it is not possible to draw definite conclusions from these data regarding the length versus height, and consequently orientation, of fractures from individual stages. However, four-fifths of the reports list a stimulation height that is the same or less than the vertical height of the treatment interval in the well, suggesting almost all fracturing in California is horizontal. This is at odds with the other data submitted by operators (Figure

2.6-6), and with the predominance of vertical fracturing reported in literature regarding the reservoirs in the San Joaquin Basin, where most hydraulic fracturing occurs (Volume III, Chapter 5).

Basic work on understanding induced fractures spans decades (Hubbert and Willis, 1972; Nordgren, 1972; Perkins and Kern, 1961), but literature on studies conducted in California is limited. Emanuele et al. (1998) measured the orientation of fractures resulting from tens of stages in three horizontal wells in the Lost Hills field at a depth of approximately 600 m (2,000 ft) using surface tiltmeter measurements, along with some subsurface tiltmeter measurements. The orientation of all the fractures was within 10 degrees of vertical. Allan et al. (2010) reported on testing of longitudinal versus transverse fracturing in horizontal wells at a depth of approximately 300 m (1,000 ft) in the South Belridge field, reporting that the fractures were likely vertical as indicated by surface and downhole tiltmeter measurements.

However, both fracture *orientation* and fracture *extent* must be evaluated. In work performed outside of California, where fracturing occurs generally much deeper and with less injection volumes, fracture orientations have been different. Flewelling and Sharma (2014) observed that shallow formations are more likely to fracture horizontally rather than vertically, regardless of fracture extent, and capped potential fracture vertical extent at 600 m (2,000 ft) or less. Fisher and Warpinski (2012) compared microseismic data on fracture extent and found that fractures in shallower formations (<1,200 m, or 3,900 ft) have a greater horizontal component, and that deep hydraulic fractures should not be vertically extensive such as to contact shallow aquifers. This paper, however, also stated that earlier work found orientations dependent on the unique stress profiles and rock fabric of a given location (Walker et al., 2002). Coupled flow-geomechanical modeling (Kim and Moridis, 2012) found inherent physical limitations to the extent of fracture propagation—for example, the presence of overlying confining formations may slow or stop fracture growth in the vertical direction, thus containing fractures within the reservoir (Kim et al., 2014). Likewise, Davies et al. (2012) find that the majority of induced fractures (with data focused on high-volume fracturing operations in the Barnett Shale in Texas) range from less than 100 m (330 ft) to about 600 m (2,000 ft) in vertical extent, with approximately a 1% probability of a fracture extending 350 m (1,100 ft) vertically. This leads to a suggested minimum separation of 600 m (2,000 ft) between shale reservoirs and overlying groundwater resources for high-volume fracturing operations conducted in deeper formations elsewhere in the country (King, 2012). For comparison, completion reports show that the fractures in California can be as shallow as 200 m (650 ft) from the surface, which is much less than this suggested minimum, and thus a predominantly vertical fracture orientation increases the likelihood of encountering protected groundwater. More studies are needed to evaluate the fracturing behavior, fracture propagation, and the orientation of fractures relative to reservoir depth in typical hydraulic fracturing operations in California.

2.6.2.7. Leakage through Failed Inactive (Abandoned, Buried, Idle or Orphaned) Wells

Oilfield gas and formation water may reach the surface through degraded and leaking wellbores. Regions with a history of oil and gas production such as California have a large number of inactive (abandoned, buried, idle or orphaned) wells, many of which may be undocumented, unknown, and either degraded, improperly abandoned, or substandard in construction. Fractures created during hydraulic fracturing can create connectivity to inactive wells, particular in high-density fields such as found in the Kern County in California. However, the inactive wells have to fail (for example, due to degradation of cement or casings), and sufficient driving forces must be present for leakage of gas or formation water to occur through inactive wells.

In California, there are more inactive than active wells. Of a total of about 221,000 wells listed in the DOGGR GIS wells file, nearly 116,000 wells have been plugged and abandoned according to state standards. Nearly 1,800 wells are "buried," i.e., older wells which have not been abandoned to standards and whose location is approximate. Finally, the status of 388 wells is unknown, i.e., these are pre-1976 wells whose status is only on a hard copy file. Approximately 53% of the abandoned wells are located in Kern County. DOGGR also has an idle and orphan well program.[10] An idle well is defined as "a well that has not produced oil and/or gas or has not been used for fluid injection for six consecutive months during the last five years". An orphaned well is an abandoned well that has no owner. The DOGGR idle wells inventory lists, as of December 2014, a total of 21,347 idle wells, although this number differs from the number of idle wells reported in the GIS wells file (13,450 wells). DOGGR also lists 110 currently orphaned wells in California and an additional 1,307 hazardous orphaned wells were plugged by DOGGR between 1977 and 2010.

The accuracy of the locations of inactive wells listed in the DOGGR GIS wells file has not been independently verified, and the actual counts of buried wells may be underestimated, since there could be historical wells whose location is unknown. The conditions of the abandoned, plugged, and buried wells are unknown. Under SB 4, operators are required to identify plugged and abandoned wells that may be impacted by the stimulation operation while applying for a permit, but are not required to test their condition. Idle wells are required to be tested periodically to ensure that they are not impacting surface and groundwater by the DOGGR Idle and Orphan well program. The type of testing required is not specified, and can be as simple as a fluid-level survey or may be a more complicated well-casing mechanical integrity test.

Old and inactive wells are a problem in many other states. For example, in Pennsylvania, there are thousands of wells from previous oil and gas booms, with 200,000 dating from before formal record-keeping began and 100,000 that are essentially unknown (Vidic

10. See http://www.conservation.ca.gov/dog/idle_well/Pages/idle_well.aspx

et al., 2013), and increasing attention has been given to assessing these as transport pathways. Abandoned wells have also been attributed as causes for contamination of groundwater in Ohio and Texas and programs to locate, assess, and cap previously abandoned wells have been subsequently initiated in those states (Kell, 2011). Chilingar and Endres (2005) documents a California incident in 1985, where well corrosion at shallow depths led to casing failure of a producing well and the subsequent migration of gas via a combination of abandoned wells and fault pathways to a Los Angeles department store basement, resulting in an explosion. The paper also documents multiple cases of gas leakage from active oil fields and natural gas storage fields in the Los Angeles Basin and elsewhere, with the most common pathway being gas migration through faulted and fractured rocks penetrated by abandoned and leaking wellbores, many of which predate modern well-casing practice and are undocumented or hidden by more recent urban development. While stimulation technologies are not implicated in these events, they illustrate the real possibility of degraded abandoned wells as pathways.

The hazards of degraded abandoned wells are not just limited to their proximity to stimulated wells, but are also relevant to the issue of disposal of wastewater from stimulated wells by injection into Class II wells. A 1989 U.S. Government Accountability Office (GAO) study of Class II wells across the United States (U.S. GAO, 1989) found that one-third of contamination incidents were caused by communication with an improperly plugged abandoned oil and gas well. Current UIC program permitting requirements require a search for abandoned wells within a quarter mile of a new injection wellbore, and plugging and remediation of any suspect wellbores (40 CFR 144.31, 146.24). However, Class II wells operating prior to 1976 are exempt from this requirement. Thus, 70% of the disposal wells reviewed were pre-existing, grandfathered into the program, and allowed to operate without investigating nearby abandoned wells (U.S. GAO, 1989).

2.6.2.8. Failure of Active Production, Class II, and Other Wells

Operating wells (whether used for production or injection) can serve as leakage pathways for subsurface migration. Pathways can be formed due to inadequate design, imposed stresses unique to stimulation operations, or other forms of human error. Class II deep injection wells with casing or cement inadequacies would also have similar potential for contamination as a failed production well or a well that fails due to stimulation pressures. Examples of potential subsurface releases through wells are illustrated in Figure 2.6-2.

Stimulated wells may be subject to greater stresses than non-stimulated wells, due to the high-pressure stimulation process and the drilling practices used to create deviated (often horizontal) wells (Ingraffea et al., 2014). During hydraulic fracturing operations, multiple stages of high-pressure injection may result in the expansion and contraction of the steel casing (Carey et al., 2013). This could lead to radial fracturing and/or shear failure at the steel-concrete or concrete-rock interfaces, or even separation between the casing and the cement. These gaps or channels could serve as pathways, or (as a worst-case) create connectivity between the reservoir and overlying aquifers. Current practice does not typically use the innermost casing as the direct carrier of stimulation fluids (or produced fluids and gases). Additional tubing (injection tubing or production tubing) is run down the innermost well casing without being cemented into place, and thus carries the stresses associated with injection. However, less complex stimulation treatments, such as some California operations, may not require such additional steps, and some fracturing operations may use the innermost casing to carry the fracturing fluids and the pressures associated with the fracturing operation.

In addition, several mechanisms—such as surface subsidence, reservoir compaction or heaving, or even earthquakes—can lead to well impairment due to casing shear (Dussealt et al., 2001). The diatomite formations in Kern County are highly porous and compressible, and hence are particularly susceptible to depletion-induced compaction. For example, several wells failed in the 1980s in Belridge (at a peak rate of 160 wells per year) following years of active production enabled by stimulation, which led to reservoir depletion and subsidence (Fredrich et al., 1996; Dussealt et al., 2001). Waterflood programs were then initiated to counter the subsidence, which led to much lower rates of well failure in the late 1990s of around 2–5% of active wells per year or approximately 20 wells per year (De Rouffignac et al., 1995; Fredrich et al., 1996; Dussealt et al., 2001). The current situation with groundwater overdraft in the southern San Joaquin Valley may pose an added risk to wells in the region due to subsidence. Earthquakes can also lead to casing shear; for example, hundreds of oil well casings were sheared in the Wilmington oil field in Los Angeles during five or six earthquakes of relatively low magnitude (M2 to M4) during a period of maximum subsidence in the 1950s (Dussealt et al., 2001).

Failures in well design and construction may allow migration of gas and fluids from the reservoir, or from shallower gas and fluid-bearing formations intersected by the wellbore. Wells can thus serve as pathways for gas migration to overlying aquifers or even to the surface (Brufatto et al., 2003; Watson and Bachu, 2009). Multiple factors over the operating life of a well may lead to failure (Bonett and Pafitis, 1996; Brufatto et al., 2003; Carey et al., 2013; Chilingar and Endres, 2005; Dusseault et al., 2000; Watson and Bachu, 2009); however, the most important mechanism leading to gas and fluid migration is poor well construction or exposed (or uncemented) casing (Watson and Bachu, 2009). A surface casing may not protect shallow aquifers, particularly if the surface casing does not extend to a sufficient depth below the aquifer (Harrison, 1983; 1985).

Watson and Bachu (2008) also noted that deviated wellbores, defined as "any well with total depth greater than true vertical depth," show a higher occurrence of gas migration than vertical wells, likely due to the challenges of deviated well construction increasing the likelihood of gaps, bonding problems, or thin regions in the cement that could create connectivity to other formations. In a review of the regulatory record, Vidic et al. (2013) noted a 3.4% rate of cement and casing problems in Pennsylvania shale-gas wells (that all had some degree of deviation) based on filed notices of violation. Pennsylvania inspection records, however, show a large number of wells with indications of cement/casing impairments for which violations were never noted, suggesting that the actual rate of occurrence could be higher than reported (Ingraffea et al., 2014; Vidic et al., 2013).

The bulk of the peer-reviewed work on contaminant migration associated with stimulation focuses on the Marcellus Shale gas plays of Pennsylvania, West Virginia, Ohio, and New York. This literature features a number of competing studies that focus on fracturing-derived pathways, but also provides a robust debate on the role of deteriorated or poorly constructed wells. A sampling study by Osborn et al. (2011a) and Jackson et al. (2013a) noted that methane concentrations in wells increased with increasing proximity to gas wells, and that the sampled gas was similar in composition to gas from nearby production wells in some cases. Follow-up work by Davies (2011) and Schon (2011) found that leakage through well casings was a better explanation than other fracturing-related processes (also see Vidic et al., 2013). Most recently, other sampling studies (Darrah et al., 2014; Molofsky et al., 2013) found gas compositions in wells with higher methane, ethane, and propane concentrations sometimes match Marcellus gas, likely through leaks in well casings; in other instances, they do not match the gas compositions in the Marcellus Shale, suggesting that intermediate formations are providing the source for the additional methane, probably due to insufficient cementing in poorly constructed wells. The Darrah et al. (2014) study in particular identifies eight locations in the Marcellus (and also for one additional case in the Barnett Shale in Texas) where annular migration through/around poorly constructed wells is considered the most plausible mechanism for measured methane contamination of groundwater.

In California, a 2011 report that studied the over 24,000 active and 6,900 inactive injection wells in the state found that, while procedures were in place to protect freshwater resources, other water resources (with higher levels of dissolved components, but not considered saline) may be at risk due to deficiencies in required well-construction practices (Walker, 2011). In California, there has been little to no investigation to quantify the incidence and cumulative hazard or indicators of wellbore impairment. However, studies from other oil- and gas-producing regions indicate that wellbores have the potential to serve as leakage pathways in California, and need to be investigated.

2.6.2.9. Leakage through Other Subsurface Pathways (Natural Fractures, Faults or Permeable Formations)

Several modeling studies have attempted to elucidate mechanisms of subsurface transport in fractured formations through numerical simulation, although in all cases some simplification of subsurface properties was necessary, since subsurface heterogeneity is both difficult to quantify and to represent in a model. A well-publicized study by Myers (2012) found potential transport between fractured reservoirs and an overlying aquifer, but did so using a highly simplified flow model regarded as unrepresentative (Vidic et al., 2013). Two recent studies modeled higher-permeability pathways intersecting reservoir boundaries. Modeling work by Kissinger et al. (2013) suggests that transport of liquids, fracturing fluids, or gas is not an inevitable outcome of fracturing into connecting pathways. Modeling work by Gassiat et al. (2013) found that migration of fluids from a fractured formation is possible for high-permeability fractures and faults, and for permeable bounding formations, but on 1,000-year timeframes. Flewelling and Sharma

(2014) conclude that upward migration through permeable bounding formations, if possible at all, is likely an even slower process operating at much longer timescales (in their estimate, ~1,000,000 years). Additional modeling studies on gas transport through fractures in shale formations, suggest gas escape is likely to be limited in duration and scope for hydrostatic reservoirs (Reagan et al., 2015; U.S. EPA 2015b). Such studies require corroborating field and monitoring studies to provide a complete view of the possible mechanisms and outcomes.

Sampling and field studies have also sought evidence of migration via fractures, but the bulk of the peer-reviewed work focuses on the Marcellus Shale, and no such studies have been conducted in California. A key conclusion is that pathways and mechanisms are difficult to characterize, and the role of fracturing or transport through fractures has not been clearly established. Methane concentrations in wells increase with proximity to gas wells, and the gas is similar in composition to gas produced nearby (Jackson et al., 2013a; Osborn et al., 2011a), but evidence of contamination from brines or stimulation fluids was not found (Jackson et al., 2011; 2013a; Osborn et al., 2011b), suggesting that gas and liquid migration may not be driven by the same processes. The most recent sampling studies (Darrah et al., 2014; Molofsky et al., 2013) conclude that migration through poorly constructed wells is a more likely scenario than fracture-related pathways. Work on the properties of gas shales (Engelder et al., 2014) proposed that a "capillary seal" would restrict the ability of fluids to migrate out of the shale, but many reservoirs in California contain more mobile water, reducing this possibility.

Fault activation resulting in the formation of fluid pathways is an additional concern when stimulation operations occur in faulted geologies, such as in California (Volume II, Chapter 4). Fault activation is a remote possibility for faults that can admit stimulation fluids during injection (Rutqvist et al., 2013), possibly increasing the permeability of previously sealed faults or creating new subsurface pathways analogous to induced fractures (possibly on a larger scale). Fault activation could also give rise to (small) micro-seismic events, but fault movement is limited to centimeter scales across fault lengths of 10 to 100 m (33 to 330 ft) (Rutqvist et al., 2013). Chilingar and Endres (2005) document a California incident in which the migration of gas via permeable faults (among other pathways) created a gas pocket below a populated area in Los Angeles and resulted in an explosion. While the incident was not related to stimulation operations, it shows how naturally faulted geologies can provide pathways for migration of gas and fluids.

2.6.2.10. Spills and Leaks

Oil and gas production involves some risk of surface or groundwater contamination from spills and leaks. Well stimulation, however, raises additional concerns, owing to the use of chemicals during the stimulation process, the generation of wastewaters that contain these chemical additives (as well as formation brines with potentially different compositions from conventional produced waters), and the increased transportation requirements to haul these materials to the well and disposal sites.

Surface spills and leaks can occur at any time in the stimulation or production process. Spills and leaks can occur during chemical or fluid transport, pre-stimulation mixing, during stimulation, and after stimulation during wastewater disposal. In addition, storage containers used for chemicals and well stimulation fluids can leak (Figure 2.6-1). Releases can result from tank ruptures, piping failures, blowouts, other equipment failures and defects, overfills, fires, vandalism, accidents, or improper operations (NYSDEC, 2011). Additionally, natural disasters (e.g., floods or earthquakes) may damage storage and disposal sites or cause them to overflow. For example, major flooding in 2013 damaged oil and gas operations in northeast Colorado, spilling an estimated 180 m^3 (48,000 gal) of oil and 160 m^3 (43,000 gal) of produced water (COGCC, 2013). Once released, these materials can run off into surface water bodies and/or seep into groundwater aquifers.

In California, any significant or threatened release of hazardous substances must be reported to California Office of Emergency Services (OES) (19 CCR 2703(a)). According to California state law, the reporting threshold for chemical spills varies by chemical. There is no specific reporting threshold for produced water, although any release must still be reported to the appropriate DOGGR district office (Cal. Code Regs. tit. 14, § 1722(i)). All spills into or on state waters must also be reported to OES. OES maintains a database with information on the location, size, and composition of the spill; whether the spill impacted a waterway; and the cause of the spill. OES then conveys information on spills originating from or associated with an oil or gas operation to DOGGR, and DOGGR staff enters these data into the California Well Information Management System (CalWIMS) database. In some cases, DOGGR works with companies after a spill has occurred to obtain additional information and, as a result, some of the data within DOGGR and OES spills databases are inconsistent. For this analysis, we relied on the OES database; however, we discuss the need to standardize these databases in Section 2.9. It is of note that operators are not required to report whether a spill was associated with well stimulation, nor do the reports contain an American Petroleum Institute (API) number, which could be used to link the spill to stimulation records.

Between January 2009 and December 2014, a total of 575 produced water spills were reported to OES, or an average of about 99 spills annually. The majority (55%) of these spills occurred in Kern County, followed by Los Angeles (16%), Santa Barbara (13%), Ventura (6%), Orange (3%), Monterey (2%), and San Luis Obispo (1%), and Sutter (1%) counties. Nearly 18% of these spills impacted waterways.

Chemical spills were also reported in California oil fields, including spills of chemicals typically used in well stimulation fluids, e.g., hydrochloric, hydrofluoric, and sulfuric acids. Between January 2009 and December 2014, a total of 31 chemical spills were reported to OES. Forty-two percent of these spills were in Kern County, followed by Los Angeles (16%), Sonoma (16%), and Lake (3%) counties. Chemical spills represent about 2% of all reported spills attributed to oil and gas development during that period. None of the reported spills contained chemicals used for hydraulic fracturing in California.

Nine of the chemical spills were of acid. This suggests that acid spills are relatively infrequent, representing less than 1% of all reported spills attributed to oil and gas development during that period. Among these was a storage tank at a soft water treatment plant containing 20 m^3 (5,500 gal) of hydrochloric acid in the Midway-Sunset Oil Field in Kern County that ruptured violently, releasing the acid beyond a secondary containment wall. No injuries or deaths were associated with this or any other acid spill. While 10% of the chemical spills were reported to enter a waterway, none of the acid spills was reported to enter a waterway.

2.6.2.11. Operator Error During Stimulation

Human error during the well completion, stimulation, or production processes could also lead to contamination of groundwater. Operator error could create connectivity to other formations that could serve as transport pathways. For example, poor monitoring or control of the fracturing operation could lead to creation of fractures beyond the confines of the reservoir, or increase the extent of fractures beyond desired limits. Such errors, if not found and corrected, could lead to unexpected migration of fluids, or in the case of the high-density well siting often found in California, connectivity between wells that impacts production activities themselves. Fracturing beyond the reservoir bounds due to operator error may also be of particular concern in the case of the shallower fracturing operations that may occur in California.

An example of operator error during stimulation is a 2011 incident in Alberta, Canada (ERCB, 2012), where an overlying formation was inadvertently fractured due to misreading of well fluid pressures, and stimulated fluids were injected into a water-bearing strata below an aquifer. Immediate flowback of fracturing fluids recovered most of the injected volume, and monitoring wells were installed into the aquifer and an overlying sandstone layer. A hydraulic connection between the fractured interval and the overlying aquifer was not observed, but groundwater samples contained elevated levels of chloride, benzene, toluene, ethylbenzene, and xylenes (BTEX), petroleum hydrocarbons and other chemicals. The Energy Resources Conservation Board (ERCB) finding states that the incident presented “insignificant” risk to drinking water resources, but criticized the onsite crew’s risk management, noting there were multiple opportunities to recognize abnormal well behavior before the misplaced perforation.

2.6.2.12. Illegal Discharges

Illegal discharges of wastewater from oil and gas production have been noted in California for disposal in both unlined pits and via subsurface injection. For example, in July 2013, the CVRWQB issued a $60,000 fine to Vintage Production California, LLC, for periodically discharging saline water, formation fluids, and hydraulic fracturing fluid to an unlined pit in an area with good-quality groundwater (CVRWQCB, 2013). In a follow-up survey on disposal practices of drilling fluids and well completion fluids, the CVRWQCB identified several other illegal discharge incidents between January 2012 and December 2013 and fined the responsible operators (CVRWQCB, 2014). In a recent GAO review of the UIC programs in eight states, California agencies reported 9 and 12 instances of alleged contamination in 2009 and 2010, respectively, resulting from one operator injecting fluids illegally into multiple wells (U.S. GAO, 2014).

2.7. Impacts of Well Stimulation to Surface and Ground Water Quality

In this section, we review the potential impacts of well stimulation on water quality by examining results from the few sampling studies that have been conducted near hydraulic fracturing operations in the United States. Only one sampling study has been conducted near a hydraulic fracturing site in California (in Inglewood). Thus, we considered studies conducted in other regions of the United States where stimulation operations have occurred, including Pennsylvania, Texas, Ohio, Montana and North Dakota, to (1) examine incidents where water has been potentially contaminated due to oil and gas activities, to determine viable contaminant release mechanisms, and assess whether they apply to well stimulation activities in California; and (2) identify considerations for future sampling studies and monitoring programs in California, based on lessons learned from other states.

While some of the sampling studies have shown no evidence of water contamination associated with well stimulation, other studies found detectable impacts that were associated with, and allegedly caused by, well stimulation operations. A recently released draft report by the U.S. EPA did not find evidence of widespread, systemic impacts on drinking water resources in the United States, but found specific instances of impacts on drinking water resources, including contamination of drinking water wells. (U.S. EPA 2015b).

Notably, most groundwater sampling studies do not even measure stimulation chemicals, partly because their full chemical composition and reaction products were unknown. It should be noted that detecting groundwater contamination is more difficult than detecting surface water contamination because (1) the effects of contamination, the release mechanisms, and the transport pathways are less visible than at the surface; (2) there are many possible pathways and sources for contaminants to be present in groundwater, and definitively attributing contamination to well stimulation is difficult; and (3) impacts on groundwater may not be detected on relatively short time scales because of slow transport processes. These difficulties are compounded by the lack of baseline water quality data and monitoring to detect problems, as well as the lack of knowledge about the full composition of stimulation fluids and standard analytical methods to detect the chemical additives and their degradation products.

2.7.1. Studies that Found Evidence of Potential Water Contamination near Stimulation Operations

Several studies have found evidence of contamination due to stimulation, which were primarily attributed to surface spills or leaks of fluids used in hydraulic fracturing, or improper wastewater disposal (Table 2.7-1). For example, in 2007, flowback fluids overflowed retention pits in Knox County, KY, killing or displacing all fish (including Blackside Dace, a federally threatened species), invertebrates, and other biota for months over a 2.7 km (1.7 mi) section of a local waterway (Papoulias and Velasco, 2013). In a study examining the effect of spills, the presence of known or suspected endocrine-disrupting chemicals used for hydraulic fracturing were measured at higher levels in surface and groundwater samples in drilling-dense areas of Garfield County, Colorado compared to nearby background sites with limited or no drilling activity (Kassotis et al., 2013). Surface water samples were collected from five distinct sites that contained from 43 to 136 natural gas wells within 1.6 km (1 mi) and had a spill or incident related to unconventional natural gas extraction within the previous six years.

There have been far fewer reports of groundwater contamination caused by subsurface release mechanisms, such as leakage through wells or leakage through hydraulic fractures or other natural permeable pathways. Most of the problems reported were due to the presence of methane gas or other formation water constituents in drinking water wells, and only three reports involve the possibility of contamination by hydraulic fracturing fluids. A recent study in Pennsylvania investigates an incident of contamination by natural gas in potable groundwater, where well waters were also observed to foam (Llewellyn et al., 2015). The authors used 2-D gas chromatography coupled to time-of-flight mass spectrometry (GCxGC-TOFMS) to identify an unresolved complex mixture of organic compounds in the aquifer that had similar signatures to flowback water from Marcellus shale-gas wells. The organic compounds were not present in nearby wells that were outside of the affected area. One compound in particular, 2nbutoxyethanol, which is not a natural constituent of water in the region, was identified in both the foaming waters and flowback water, although the study mentions that it could have also been used in drilling fluids. The authors conclude that, although they were not able to unambiguously prove a direct connection between shale gas operations and the detected organic chemicals in household waters, the timing and presence of similar compounds in "flowback/produced" waters suggest that the hydraulic fracture operations were a likely source (Llewellyn et al., 2015). The contaminant release mechanisms suggested by the authors include surface spills or subsurface leakage and transport through shallow fractures. The study also suggests that the most likely release mechanism for the natural gas was leakage through wells due to excessive annular pressures and lack of proper annular cement (Llewellyn et al., 2015).

There are two other unconfirmed potential groundwater contamination incidents attributed to subsurface leakage of hydraulic fracturing fluid within the United States (DiGiulio et al., 2011; U.S. EPA, 1987), but neither of them has been documented in a peer-reviewed publication (Brantley et al., 2014; Vidic et al., 2013). The first study is a U.S. EPA investigation in Pavilion, Wyoming, where surface storage and disposal of wastewaters was implicated in contamination of shallow surface water as discussed in Section 2.6. Initial results published in a draft report (DiGiulio et al., 2011) suggested that groundwater wells had been contaminated with various fracturing-fluid chemicals (glycols and alcohols) as well as methane, via flow from the stimulated reservoir to groundwater. However, a follow-up study by the USGS involving resampling of the wells could not confirm some of these findings (Wright et al., 2012). The U.S. EPA is no longer working on this study, which is now being led by the State of Wyoming. The second reported incident of groundwater contamination is based on a U.S. EPA study focusing on operations in Ripley, West Virginia. In this case, a gel used as a constituent in fracturing fluids was reported to have contaminated a local water well located less than 330 m (1,000 ft) from a vertical gas well (U.S. EPA, 1987). Contaminant transport could have either occurred through four abandoned wells located near the vertical gas well during the fracturing process, or by contamination from the flush fluid used to remove loose rock cuttings prior to cementing (Brantley et al., 2014).

Several other studies note the presence of elevated levels of other contaminants in groundwater near stimulation operations. Some studies were unable to attribute the cause to stimulation, while others had to conduct several follow-on investigations to identify the contaminant release mechanisms. For example, some sampling studies found high concentrations of methane and other hydrocarbons in drinking-water wells in Pennsylvania, particularly those near hydraulic fracturing operations. Methane concentrations in the wells increased with increasing proximity to gas wells, but evidence of contamination from brines or fracturing fluids was not found (Dyck and Dunn, 1986; Jackson et al., 2011; 2013a; Osborn et al., 2011a; 2011b). There was significant debate about whether the high methane concentrations were naturally present, or a result of hydraulic fracturing operations. Additional sampling work (Jackson et al., 2013a) found ethane and propane, as well as methane, in water wells near Marcellus production locations. The studies determined that the methane was formed by thermogenic processes at depth (as would be expected for shale gas), and that the isotopic ratios of methane were found to be more consistent with non-Marcellus gas (Molofsky et al., 2013). The most recent sampling study (Darrah et al., 2014) again found isotopic and noble gas compositions inconsistent with a Marcellus (and thus a stimulation-derived) source, and identified eight locations where wells are considered the most plausible mechanism for measured methane contamination of groundwater—including incidents of migration through annulus cement (four cases), through production casings (three cases), and due to underground well failure primarily. In another study in the Marcellus, radon concentrations obtained from previously measured public data were found to increase in proximity to unconventional wells (Casey et al., 2015). Radon is a radioactive decay product of radium, and can dissolve and be transported through groundwater. The researchers also noted that concentrations increased in 2004 from previously fluctuating measurements, just preceding the Marcellus boom in 2005. However, the study had several shortcomings, including the lack of any detailed statistical measures for spatial association of radon with hydraulic fracturing operations, the lack of evidence showing any pathway that could cause an increase in radon concentrations, the reliance on unverified public data that were not necessarily submitted by accredited professionals, and other limitations that led to an acknowledgement by the authors stating that the study was exploratory.

Another study conducted in the Barnett Shale also illustrates the difficulty in tracing the source of the contaminants detected in groundwater near well stimulation operations shale (Fontenot et al., 2013), despite having historical and background water quality data. This study sampled 100 groundwater wells located in aquifers overlying the Barnett, and found that TDS concentrations exceeded the U.S. EPA Secondary Maximum Contaminant Level (MCL) of 500 mg L^{-1} in 50 out of 91 samples located within 3 km (1.9 mi) of gas wells, and that the maximum values of TDS near the wells were over three times higher than those from background wells located in areas that were unimpacted by fracturing enabled oil and gas development. Similarly, trace elements such as arsenic, barium, selenium, and strontium were found to be present at much higher levels compared to background or historical concentrations, and organics (methanol and ethanol) were detected in 29% of samples in private drinking-water wells. However, it was not possible to determine if hydraulic fracturing was the cause of the high TDS, trace element or organic concentrations, since historical, regional, and background values of these constituents were also high.

An extensive review of groundwater-contamination claims and existing data can be found in a report for the Ground Water Protection Council, focusing on Ohio and Texas groundwater-investigation findings during a 16-year study period from 1983 through 2008 (Kell, 2011). The study area and time period included the development of 16,000 horizontal shale gas wells with multistage fracturing operations in Texas and one horizontal shale gas well in Ohio. The report notes that, for the study period, no contamination incidents were found involving any stimulation activities including “site preparation, drilling, well construction, completion, hydraulic fracturing stimulation, or production operations at any of these horizontal shale gas wells.” However, there were a total of 211 reported groundwater contamination incidents in Texas caused by other oil and gas activities. Seventy-five of these were caused by wastewater management and disposal activities, including 57 incidents due to improper storage of wastewater in surface containment pits. This practice has mostly been replaced by disposal via Class II injection wells that have a significantly better record of protecting groundwater resources than unlined pits (as discussed in Section 2.6). Other contamination incidents were related to orphaned wells (30 incidents, most of which were caused by inadequately sealed boreholes) and production activities (56 incidents that include 35 releases from storage tanks, 12 releases from flow lines or wellheads, 7 releases from historic clay-lined storage pits, and 2 releases related to well construction including an incident caused by a short surface casing that did not adequately isolate all groundwater). In Ohio, a total of 185 groundwater-contamination incidents were reported from other oil and gas activities, most of which occurred prior to 1993. Of these, 41 incidents were related to orphaned wells in abandoned sites, 39 incidents were caused by production-related activities (including 17 incidents of leaks from storage tanks or lines; 10 incidents caused by onsite produced water storage pits; 12 incidents caused due to well construction issues), and 26 incidents caused due to waste management and disposal activities. The report concludes that, although no documented links have been found implicating the fracturing process itself to contamination incidents, a regulatory focus on activities that could be linked to contamination is critical, along with documentation of hydraulic fracturing operations such that regulators can determine which processes put groundwater at risk.

Table 2.7-1. Examples of release mechanisms and contamination incidents associated with oil and gas activities in the United States.

Year	**Location**	**Media Impacted**	**Contaminant**	**Attributed to Well Stimulation?**	**Evidence of Water Contamination**	**Release Mechanism**	**Operator**	**Source**
1982	Jackson County, WV	Groundwater	Gelatinous material (fracturing fluid) and white fibers	Disputed	Fluid and fibers were found in the water sample from a well located <1,000 ft from a vertical gas well.	Unknown; 4 abandoned gas wells drilled in the 1940s are present within 1,700 ft of the new gas well and may have served as conduits for contamination.	Kaiser Gas Co.	Brantley, 2014; U.S. EPA, 1987
2007	Knox County, KY	Surface water	Flowback fluids	Yes	Flowback fluids were released directly into Acorn Fork. The incident killed or displaced all fish, invertebrates, and other biota for months over a 2.7 km section of the creek.	Retention pits overflowed.	Not known	Papoulias and Velasco, 2013
2007	Bainbridge Township, Geauga County, OH	Groundwater	Natural Gas	Yes	Natural gas seeped into an aquifer	Defective cement job in the well casing, compounded by operator error.	Ohio Valley Energy Systems Corp.	Ohio DNR, 2008
2009	Hopewell Township, Washington County, PA	Surface water	Wastewater	Yes	Fluid overflowed the impoundment's banks and ran over the ground and into a tributary of Dunkle Run.	Wastewater pit failure.	Atlas Resources LLC	PA DEP, 2010
2009	Dimock Township, Susquehanna County, PA	Surface water	8,000 gallons of water/liquid gel mixture used in hydraulic fracturing	Yes	Pollution in Stevens Creek and a nearby wetland resulted in a fish die-off.	Unknown	Cabot Oil & Gas	PA DEP, 2009
2011	Alberta, Canada	Groundwater	Fracturing fluids	Yes	Groundwater samples from monitoring wells found elevated levels of chloride, BTEX, petroleum hydrocarbons, and other chemicals.	Inadvertent fracturing of an overlying formation and injection of fluids into water-bearing strata below an aquifer.	Crew Energy Inc.	ERCB, 2012
2013	Colorado	Surface water	48,000 gallons of oil and 43,000 gallons of produced water	Yes		Spill during major flooding in 2013 damaged oil and gas operations.	Multiple	COGCC, 2013

Year	Location	Media Impacted	Contaminant	Attributed to Well Stimulation?	Evidence of Water Contamination	Release Mechanism	Operator	Source
2013	Kern County, CA		Saline water, formation fluids, and hydraulic fracturing fluid	Yes	None	Illegal discharge to an unlined pit.	Vintage	CVRWQCB, 2013
1983-2007	Ohio	Groundwater	Drilling contaminants- e.g., drill cuttings, crude oil, flowback, and produced water	Not known	The Ohio Division of Mines and Reclamation documented 185 groundwater contamination incidents caused by historic or regulated oilfield activities over a 25 year period	41 of the 185 incidents were caused by orphaned wells. 144 were caused by violations at permitted or regulated activities, including drilling & completion; production, on-lease transport, & storage; waste management & disposal; and plugging & site reclamation.	Not known	Kell, 2011
1993-2008	Texas	Groundwater	Multiple - e.g., drill cuttings, crude oil, flowback, and produced water	Not known	The Texas Railroad Commission documented 211 incidents of groundwater contamination caused by historic or regulated oilfield activities over the 16 year period.	75 incidents resulted from waste management and disposal activities, including 57 legacy incidents caused by produced water disposal pits that were phased out starting in 1969 and closed by 1984. 56 incidents related to releases that occurred during production phase activities including storage tank or flow line leaks. 30 incidents were caused by orphaned wells or sites.	Not known	Kell, 2011
2010-2011	Weld County, CO	Groundwater	BTEX	Not known	77 reported surface spills impacting the groundwater	Spills due to 1. Equipment failure (47 spills); 2. corrosion/equipment failure (10 spills); 3. historical impact (i.e., discovery of a spill during inspection) (15 spills); 4. human error (3 spills); 5. Multiple leaks in dump line system (1 spill); and 6. unknown (1 spill).	Not known	Gross et al., 2013
2015	Pennsylvania	Groundwater	Methane, unresolved mixture of organic compounds (including 2-butoxyethanol)	Likely (but not unambiguously) caused by stimulation	Wells that had been previously contaminated with natural gas were observed to be foaming	Suggested release mechanisms are surface spills or release/transport through shallow fractures for the organic compounds. Methane was probably released through a different mechanism (through well casing)	Not known	Llewelyn et al., 2015

Year	Location	Media Impacted	Contaminant	Attributed to Well Stimulation?	Evidence of Water Contamination	Release Mechanism	Operator	Source
Not known	Kansas, Kentucky, Michigan, Mississippi, New Mexico, Oklahoma, and Texas.	Groundwater	Brine	Not known	A majority of the 23 cases were identified by users of the groundwater. The rest were identified by operators or EPA staff during monitoring operations or while reviewing injection records.	Improperly plugged oil and gas wells in the vicinity of Class II injection wells; leaks in Class II injection well casing; and Class II injection into a USDW.	Multiple	U.S. GAO, 1989
Not known	Pavillion, WY	Groundwater	Benzene, xylenes, gasoline-range organics, and diesel-range organics	Yes	High concentrations of hydraulic fracturing chemicals were found in shallow monitoring wells near surface pits.	Infiltration from storage/disposal pits.	Encana Oil and Gas	DiGiulio et al., 2011; Folger et al., 2012
Not known	Pavillion, WY	Groundwater	Elevated concentrations of well stimulation and drilling chemicals	Disputed	Contaminants were detected in deep monitoring wells.	Thought to be related to gas production; however, the gas company disputes this claim.	Encana Oil and Gas	Folger et al., 2012
Not known	Saskatchewan, Canada	Groundwater	Methane	Not known	Elevated levels of methane found in groundwater associated with oil and gas fields.	Unknown; article suggests leakage along the exploration holes or migration through natural fractures.	Not known	Van Stempvoort et al., 2005
Not known	Texas	Surface water	Sediment	Not known	Study shows a strong correlation between shale-well density and stream turbidity.	Sediment runoff from wellpads.	Not known	Williams et al., 2008
Not known	Garfield County, CO	Surface and Groundwater	Endocrine-disrupting chemicals	Not known	Data suggest elevated endocrine-disrupting chemical activity in surface water and groundwater close to unconventional natural gas drilling operations.	Unknown	Not known	Kassotis et al., 2013
Not known	Northeastern PA and upstate NY	Groundwater	Methane	Not known	Study shows that, in active gas-extraction areas, average and maximum methane concentrations in drinking-water wells increased with proximity to the nearest gas well.	Unknown	Not known	Osborn et al., 2011a

2.7.2. Studies that Found No Evidence of Water Contamination Near Stimulation Operations

There are a few sampling surveys that have been conducted near stimulation operations in the United States. Many of these studies found no evidence of water contamination near stimulation operations, including the only sampling study conducted in California (Cardno ENTRIX, 2012).

The California study reviewed ten years of oil and gas production, including two years of well stimulation operations, at the Inglewood field in Los Angeles County. During this period, conventional hydraulic fracturing was conducted on 21 wells and high-volume hydraulic fracturing was conducted on two wells.[11] The Inglewood field is located in a populated area and underlies a freshwater formation that is regulated and monitored for water quality (Cardno ENTRIX, 2012). The study sampled the groundwater for pH, total petroleum hydrocarbons (TPH), benzene, methyl tertiary butyl ether (MTBE), total recoverable petroleum hydrocarbons (TRPH), total dissolved solids (TDS), nitrate, nitrite, metals, and biological oxygen demand (BOD), none of which is a specific analysis for chemicals used in hydraulic fracturing. The study concluded that there were no detectable impacts to groundwater quality due to the production or stimulation activities (Cardno ENTRIX, 2012). There was no evidence of migration of stimulation fluids, formation fluids, or methane gas during the study's timeframe, even though the formation contained faults and fractures connecting shallow formations to deeper formations (Cardno ENTRIX, 2012). Monitoring found no significant differences in pre-drilling and post-stimulation TDS levels. Trace metals were also sampled; arsenic was the only trace element that exceeded drinking water standards. However, the study mentions that arsenic is naturally present at high levels in Southern California, and concentrations were high in the monitoring wells before drilling (Cardno ENTRIX, 2012). Microseismic monitoring in the study indicated that fractures were contained within the hydrocarbon reservoir zone, extending to within no more than 2,350 m (7,700 ft) of the base of the freshwater zone (Cardno ENTRIX, 2012).

Outside of California, a few other studies have sampled water quality near hydraulically fractured wells in several regions, including the Marcellus Shale, Pennsylvania (e.g., Boyer et al., 2011; Brantley et al., 2014 and references therein; Siegel et al., 2015), the Fayetteville Shale, Arkansas (Warner et al., 2013b), the Barnett Shale, Texas (Fontenot et al., 2013), and the Bakken Shale, Montana/North Dakota (McMahon et al., 2015). Many of these studies, which largely examined groundwater quality, did not find statistically significant changes to the water quality of nearby groundwater wells after fracturing, when compared to baseline trends. The baseline trends were determined from samples

11. Conventional hydraulic fracturing uses water, sand, and additives to stimulate up to several hundred feet from the well and is typically applied in sandstone, limestone, or dolomite formations. High-volume hydraulic fracturing, by contrast, uses more fluids and is generally applied to shales rather than sandstones.

collected before drilling (if available) or alternatively from background sites with comparable geology and geochemistry that were considered to be relatively un-impacted by hydraulic fracturing operations.

In an extensive review, Brantley et al. (2014) found that stimulation in Pennsylvania has never been conclusively tied to an incident of water contamination, and that this could indicate that incidents are rare, and that contaminant release was diluted quickly. However, the review notes that it was not possible to draw firm conclusions due to several challenges, including (1) variable background concentrations of constituents in the groundwater and little knowledge of pre-existing contaminant concentrations; (2) lack of information about the timing and locations of drilling and production incidents; (3) withholding of water quality data from specific incidents due to liability concerns; (4) limited sample and sensor data for the constituents of concern; (5) possibility of sensor malfunction or drift. An extensive field study in the Marcellus Shale in southwest Pennsylvania was recently completed, but has not been peer-reviewed (NETL, 2014). The study combined microseismic monitoring of fracture propagation with sampling of produced gas and water from overlying conventional reservoirs. They found no evidence of gas, brine, or tracer migration into the monitored wells. A more recent study by Siegel et al. (2015) that examined an extensive industry dataset in the Marcellus Shale concluded that there was no correlation between the methane concentrations in domestic groundwater wells and hydraulic fracturing operations. However, the findings are questionable, due to the sampling strategy and techniques used (the samples were provided by the operator, Chesapeake Energy) and the lack of true baseline measurements.

In another study, 127 drinking water wells in the Fayettesville Shale were sampled and analyzed for major ions, trace metals, CH_4 gas content and its C isotopes ($\delta 13C_{CH4}$), and select isotope tracers (δ11B, Sr87/Sr86, δD, δ18O, $\delta 13C_{DIC}$). The data were compared to the composition of flowback samples directly from Fayetteville Shale gas wells. Methane was detected in 63% of the drinking-water wells, but only six wells had concentrations greater than 0.5 mg CH_4 L^{-1}. No spatial relationship was found between CH_4 and salinity occurrences in shallow drinking water wells with proximity to shale-gas drilling sites. They concluded, based on the analyses of geochemical and isotope data, that there was no direct evidence of contamination in shallow drinking-water aquifers associated with nearby stimulation operations (Warner et al., 2013b).

Another recent study conducted in the Bakken Shale sampled 30 domestic wells for major ions, nutrients, trace elements, 23 volatile organic compounds (VOCs); methane and ethane; and hydrocarbon-gas chemical (C1–C6) and isotopic (δ2H and δ13C in methane) compositions in 2013 (McMahon et al., 2015). This study also concluded that there had been no discernible effects of energy-development activities on groundwater quality, but also mentioned that the results had to be considered in the context of groundwater age and velocity. The groundwater age of the domestic wells ranged from <1,000 years to >30,000 years, based on ^{14}C measurements, and thus it was suggested that domestic wells may not be as well suited for detecting contamination from recent surface spills compared to shallower wells screened near the water table. The horizontal groundwater velocities, also calculated from ^{14}C measurements, implied that the contaminants would only have travelled ~0.5 km (0.3 mi) from the source, and thus a more long-term monitoring plan was suggested to truly assess the effects of energy development in the area.

In general, it is difficult to detect groundwater contamination, especially in situations where there has not been adequate baseline water quality data or monitoring. In cases where some monitoring has been conducted, potential contaminant release may not have been detected for a number of reasons, such as inappropriate locations for testing, slow transport of contaminants, and high analyte detection limits.

2.7.3. Quality of Groundwater Near Stimulated Oil Fields in California

In order to know if poor groundwater quality is due to oil and gas development activities, the natural quality (background quality) of the groundwater needs to be understood. Contaminants associated with oil and gas development wastewaters, including TDS, trace elements, and NORM, occur naturally in California groundwater, and regional surveys are needed to establish background concentrations in areas of oil and gas development in order to determine how this activity is impacting groundwater. Elevated levels of trace elements, such as arsenic, boron, molybdenum, chromium, and selenium, have been measured in shallow groundwater in several regions in California (e.g., Schmitt et al., 2006; 2009). High levels of uranium, frequently exceeding U.S. EPA MCLs, have also been noted in the Central Valley, and are correlated with high bicarbonate concentrations in the groundwater (Jurgens et al., 2010). Similarly, several counties in California, including Santa Barbara, Ventura, and Kern counties, are considered to be in the U.S. EPA's radon zones 1 and 2, which indicates that they have a high to moderate potential of having radon in soils and groundwater (http://www.epa.gov/radon/zonemap.html).

In studies mostly conducted outside of California, methane concentrations in groundwater have been used as an indicator of unconventional oil and gas development impacts on household sources of drinking water, and as evidence of leakage around active and abandoned wells (Osborn et al., 2011a; Jackson et al., 2013a; Lleweyln et al., 2015). A survey of methane concentrations in Southern California identified eight high-risk areas where methane could pose a safety problem (Geoscience Analytical, 1986). These include the Salt Lake Oil field in Los Angeles; the Newport Oil field; the Santa Fe Springs Oil field; the Rideout Heights area of the Whittier Oil Field; the Los Angeles City Oil field; the Brea-Olinda Oil field; the Summerland Oil field; and the Huntington Beach Oil field. Similar surveys for methane have not been conducted in other parts of California.

Salt content, measured as TDS, is a critical limiting factor for the quality of groundwater. Uses of groundwater typically have a threshold over which higher TDS is aesthetically undesirable or will result in impairment. For instance, the taste of water may become unpleasant and plant growth reduced if TDS levels are above certain thresholds. For these reasons, there are various regulatory limits regarding water quality based on the total dissolved solids content, some of which are listed in Table 2.7-2.

Table 2.7-2. Some regulatory limits regarding total dissolved solids in water.

Maximum TDS ($mg\ L^{-1}$)	Applicability	Enforceability	Overseeing Agency
500	Water supplied by a community water system	Not enforceable, but recommended	Federal EPA and CDPH
1,000		Upper limit[1]	CDPH
1,500		Short term limit[2]	
3,000	All surface and groundwater	Limit of suitability[3]	SWRCB
10,000	Groundwater	Protected, unless exempted[4]	Federal EPA, DOGGR, and SWRCB
TDS – Total Dissolved Solids			
EPA – Environmental Protection Agency			
CDPH – California Department of Public Health			
SWRCB – State Water Resources Control Board			
DOGGR – California Division of Oil, Gas and Geothermal Resources			

[1]*Acceptable if it is neither reasonable nor feasible to provide more suitable water (Cal. Cod. Reg. § 64449)*

[2]*Acceptable only for existing systems on a temporary basis pending construction of new treatment facilities that will reduce the TDS to at least the upper limit or development of acceptable new water sources water (Cal. Cod. Reg. § 64449)*

[3]*All groundwater meeting this threshold, along with various other criteria, should be designated by the Regional Boards as considered suitable, or potentially suitable, for municipal or domestic water , with the exception that groundwater designated previously designated as unsuitable may retain that designation under certain conditions (SWRCB Res.No. 88-63 as modified by Res No. 2006-0008)*

[4]*An underground source of drinking water (USDW) is defined as groundwater with TDS less than 10,000 mg L^{-1} in an aquifer with sufficient permeability and of sufficient volume to supply a public water system. Such water must be protected unless otherwise exempted (40 CFR § 144)*

The California State Water Resources Control Board (SWRCB) operates a groundwater quality and water level portal named the GeoTracker GAMA Information System ("GAMA," which stands for Groundwater Ambient Monitoring & Assessment; data portal available at http://www.waterboards.ca.gov/gama/geotracker_gama.shtml) (SWRCB, 2014a). This portal provides access to data extending back several decades.

We conducted an analysis of water quality near oil and gas operations in California, based on the minimum concentrations of TDS reported in the GAMA database. All the TDS data available from GAMA on October 10, 2014, were downloaded. The minimum value was determined in each 5 km by 5 km (3 mi by 3 mi) square area with groundwater wells in sedimentary basins with wells associated with oil and gas production starting operation from 2002 through late 2013. Figure 2.7-1 shows the results for southern California binned by the TDS thresholds shown in Table 2.7-1. None of the areas with a TDS value has a minimum greater than 10,000 mg L^{-1}, and few have a minimum greater than 3,000 mg L^{-1}. This is likely because groundwater of this quality is of limited use, and so groundwater wells would not tend to exist in these areas.

In general, the minimum TDS is below 500 mg L^{-1} in any area where a result is available (Figure 2.7-1). This is true even in many areas along the west side of the San Joaquin Valley, where Bertoldi et al. (1991) mapped the TDS as greater than 1,500 mg L^{-1}. Groundwater with less than 500 mg L^{-1} TDS occurred in many of the oil fields in this portion of the basin (Figure 2.7-1).

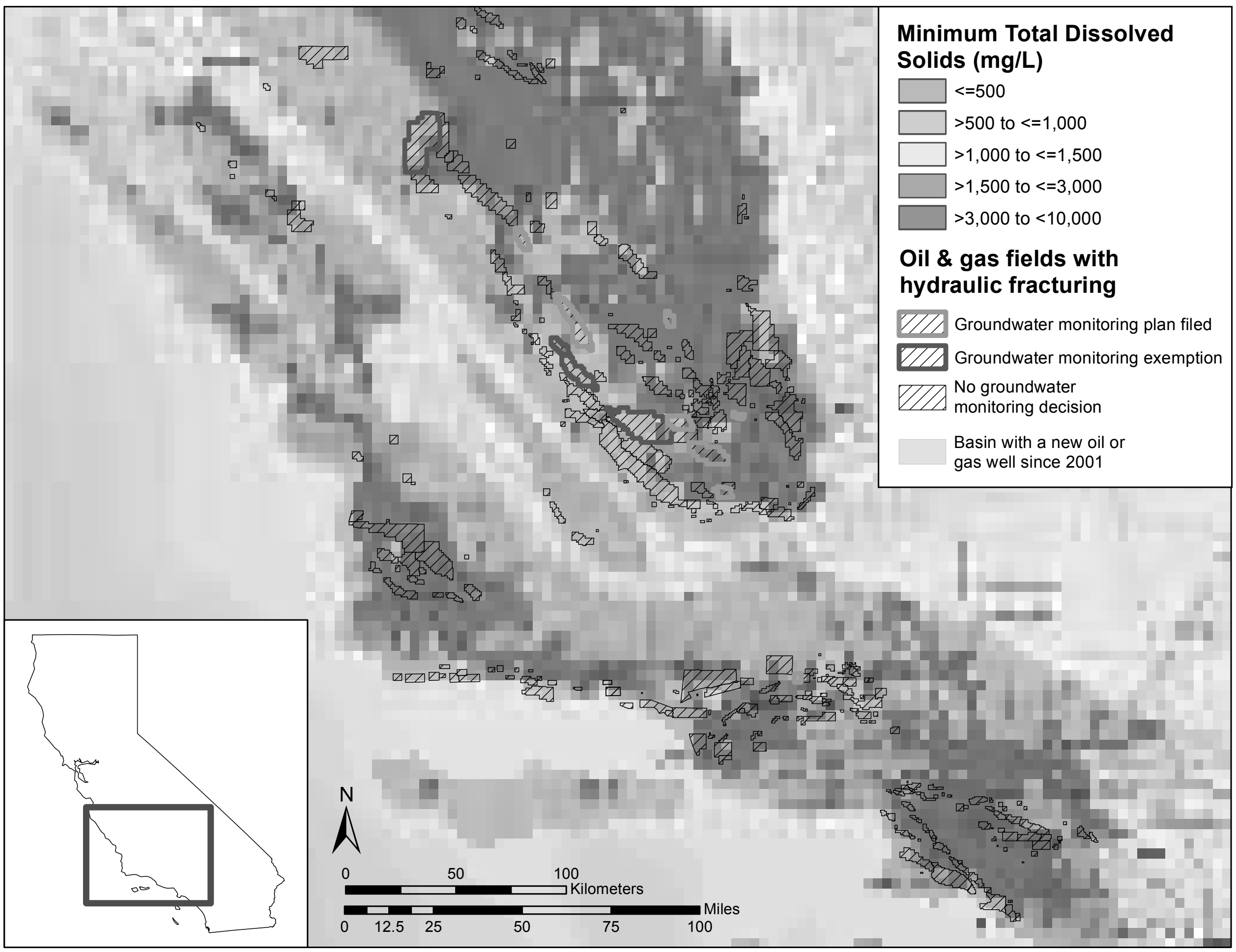


Figure 2.7-1. Minimum total dissolved solids concentration from GAMA in 5 by 5 kilometer (3 by 3 mile) square areas in central and southern California geologic basins with oil production. The status of groundwater monitoring for well stimulation projects is indicated for each field in which they have been filed.

SB 4 exempts groundwater with greater than 10,000 mg L^{-1} TDS from the monitoring requirement, as well as groundwater exempted pursuant to Section 146.4 of Title 40 of the Code of Federal Regulations. The alternative criteria described there include groundwater that occurs with hydrocarbon resources that can be economically produced, as well as groundwater that can be demonstrated to be uneconomical for use. As of October 10, 2014, operators had in some cases applied for and been granted groundwater monitoring exemptions under the TDS and hydrocarbon resource exemption provisions.

The fields for which the SWRCB has approved a groundwater monitoring plan or a groundwater monitoring exemption, according to files posted by DOGGR as of October 10, 2014, are shown in Figure 2.7-1. For the projects that were granted exclusions for groundwater monitoring from the SWRCB, the TDS data available from GAMA were either limited or indicated that the minimum TDS was greater than 1,500 mg L^{-1} (Figure 2.7-1). A possible exception is the North Belridge field.

Figure 2.7-2 shows the locations of unlined percolation pits in the Central Valley and along the Central Coast. According to this figure, percolation pits are active in areas overlying protected groundwater aquifers, especially along the eastern side of the San Joaquin Valley. In some cases, TDS levels are less than 500 mg L^{-1}. It is important to note that groundwater quality beneath the majority of active disposal pits, especially along the West San Joaquin Valley, is not known.

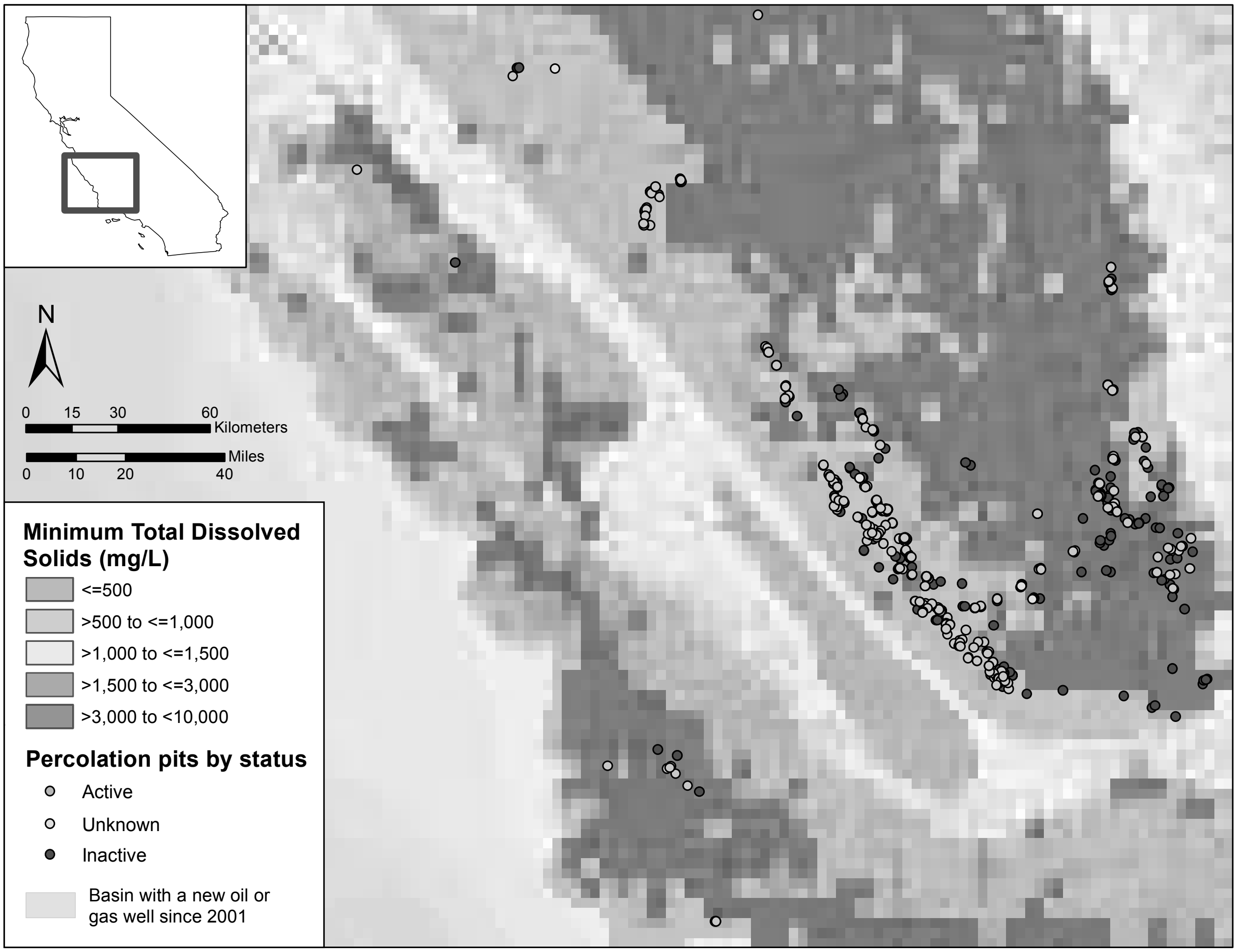


Figure 2.7-2. Minimum total dissolved solids concentration from GAMA in 5 by 5 kilometer (3 by 3 mile) square areas in central and southern California geologic basins with oil production. The location and status of unlined percolation pits in the Central Valley and Central Coast used for produced water disposal is shown. Many unlined pits are located in regions that have potentially protected groundwater.

Figure 2.7-3 provides information about the depth of hydraulic fracturing in each field. Comparison of Figures 2.7-1 and 2.7-2 indicates at least one field, Lost Hills, with hydraulic fracturing of shallow wells (<300 m [1,000 ft] deep) and groundwater of sufficient quality to require monitoring. The minimum depth of fracturing from completion reports discussed in Section 2.6 further supports this. The distribution of minimum fracturing depths indicates most are shallow, and the dataset includes reports of shallow fracturing from fields where groundwater monitoring has been required, indicating protected groundwater is present. The existence of shallow fracturing operations in areas with protected groundwater elevates concern for the hazard of subsurface migration of fluids into groundwater as a result of hydraulic fracturing.

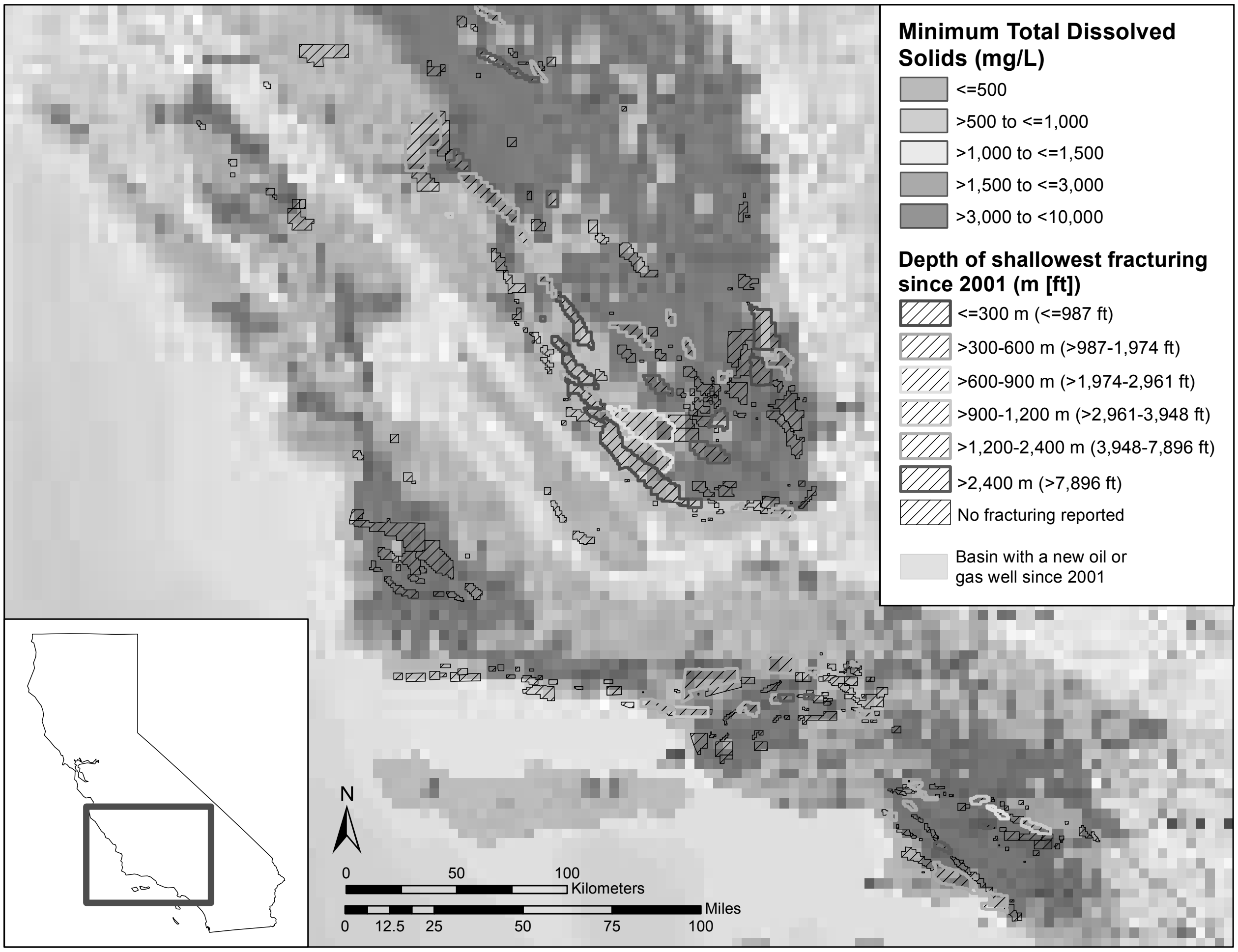


Figure 2.7-3. Minimum total dissolved solids concentration from GAMA in 5 by 5 kilometer (3 by 3 mile) square areas in central and southern California geologic basins with oil production. The available minimum depth of hydraulic fracturing in each field available in Appendix M to Volume I is shown. For most fields, this is the depth of a well in which hydraulic fracturing occurred, so the upper limit of the hydraulic fracture may be in a shallower category. Figure 3-15 of Volume I indicates the type of depth information plotted for each field.

2.8. Alternative Practices and Best Practices

In previous sections, we have examined (1) water use and sources for well stimulation; (2) the known and unknown environmental properties of various chemicals and substances used for well stimulation; (3) the quantities and characteristics of wastewater generated from stimulated wells; (4) the potential surface and subsurface release mechanisms and transport pathways associated with well stimulation; and (5) evidence of possible surface and groundwater contamination from sampling studies conducted near stimulation operations in California and elsewhere. In this section, we describe alternative and best practices that could minimize use of freshwater resources and reduce the risk of water contamination.

2.8.1. Best Practices for Well Drilling, Construction, Stimulation, and Monitoring Methods

Application of good practices while conducting well stimulation can reduce impacts from injected or mobilized fluids. Environmental impacts can be related to surface activities as well as the subsurface aspects of well stimulation. One important concern is the potential loss of containment of subsurface fluids that could result in the contamination of groundwater. Loss of containment is a significant concern for hydraulic fracturing since it is performed at high pressures. Lower-pressure injections (below fracture pressure) of acid for matrix acidizing are less likely to result in loss of containment.

Fracturing in shallower reservoirs has greater potential to result in fractures that have sufficient length to cause loss of containment and possibly impact usable groundwater. The principal way to avoid loss of containment is careful, site-specific characterization of the geologic environment, including determination of the hydrological and geomechanical properties of all stratigraphic layers. This information is then used to develop fracturing models to predict the extent of hydraulic fracturing. The model can then be used to design the injection fluid types, volumes, and rate of injection that should result in fracturing that remains contained within the target reservoir. It should be noted that current industry-standard fracture modeling typically assumes simple bi-wing fracture geometry that is most realistic for gelled fracture treatments (Cipolla et al., 2010; Weng et al., 2011). Tools to model complex fracture geometries (typical of slickwater hydraulic fracturing treatments in very low permeability systems) are relatively less mature (Weng et al., 2011). Traditional bi-wing fracture geometry models tend to overestimate the fracture penetration distance into the reservoir if complex fracture patterns are generated (Smart et al., 2014).

Analysis discussed above has shown that induced fractures that connect with high-permeability structures, such as adjacent wells, are a potential pathway for the contamination of groundwater or the ground surface. Clearly, to avoid problems with leakage along these types of structures, careful characterization of the system is necessary to identify any wells or geologic features within the area expected to be affected by the well stimulation treatment (Shultz et al., 2014). Bachu and Valencia (2014) recommend conducting hydraulic fracturing from offset wells at a safe distance, which is not specified, but would need to be evaluated using fracture modeling and field experience.

Leakage along the well receiving the well stimulation treatment could cause a loss of containment. This is an issue of proper well construction and testing, discussed in detail in Appendix 2.D and reviewed here. The key issue is the isolation of fluid movement up (or down) the well inside the casing, or tubing internal to the casing. Fluid movement along the outside of the casing or fluid exchange between inside and outside the casing, except in zones where such exchange is intended, should be prevented by the casing and cement that bonds the casing to the formation. This aspect of well construction is termed zonal isolation. Factors to be considered as part of well drilling and well construction that are important for achieving zonal isolation are discussed in Appendix 2.D and are available in technical documents describing accepted industry practices (e.g., API, 2010; ISO 10426 standards) and other technical literature (e.g., Aldred et al., 1999; Cook et al., 2012; Khodja et al., 2010; Lal, 1999; McLellan, 1996). Both internal and external well integrity tests can be performed to check on the integrity of the well and the quality of the zonal isolation.

Hydraulic fracturing treatments are routinely monitored through the pressure and flow rates of injected fluids. These monitoring tools can be used to help prevent loss of containment or identify if treatments remain within the targeted formation. Monitoring the pressure and flow rates into the well are fundamental response parameters that can be used to determine if the hydraulic fracturing treatment is proceeding properly. Both the pressure of the injection fluids and the casing pressure between the production casing and intermediate casing should be monitored. The fluid-injection pressure profile should be compared with the expected pressure profile basing on modeling. If significant deviations from the expected pressure profile are found, the hydraulic fracturing operation should be halted, to gather more information about the system and revisit the fracturing model. For instance, an unexpected drop in pressure could indicate a leak of the fracturing fluids through the casing outside the target formation. Similarly, if pressure builds in the casing annulus, treatment should be halted. This indicates flow behind the production casing, either from the targeted formation, from casing leaks above this zone, or directly from overlying formations into the annulus.

Monitoring can also be performed using geophysical measurements of microseismic (acoustic) signals from the fracturing process and from volumetric responses (dilation or compaction) that occur in response to the fracturing treatment. Such monitoring activities are typically used when new techniques or production areas are being evaluated for development, or if models of hydraulic fracturing require more detailed input (API, 2009), but they are not routine measurements. This type of monitoring provides the most detailed map of the locations where fractures generated of any monitoring method. It is performed using microseismic receiver arrays to detect the very small microseisms (or earthquakes) generated by the fracturing process (Warpinski et al., 2009). Such arrays can be placed in a monitoring hole nearby, in the well being fractured, on the ground surface, or buried in the shallow surface (Gilleland, 2011). The measurement is improved when conducted downhole, closer to where fracturing is taking place. This can be used as an after-the-fact assessment of where fractures were generated, but can also be used interactively, where real-time fracture mapping provides information to adjust the hydraulic fracturing treatment as it proceeds (Burch et al., 2009).

Another geophysical measurement device that can assess the extent of fracture growth is called a tiltmeter. This measurement detects the deformation of the earth associated with fracturing, which can then be interpreted in terms of the fracture orientation and geometry (Cipolla and Wright, 2000). Tiltmeters can be deployed in shallow boreholes or in deeper boreholes and, as for microseismic monitoring, better measurements can be obtained when the device is closer to where fracturing is taking place. Tiltmeters and microseismic monitoring have some different sensitivities in terms of the types of geometry that can be deduced from the measurements (Cipolla and Wright, 2000). Tiltmeters have also been used in a real-time mode to help guide fracture treatments as they proceed (Lecampion et al., 2004).

Monitoring of wells continues in the post-treatment period to ensure that well integrity is not compromised during production. A principal method is the monitoring of casing pressure (API, 2009). A common indication of a problem is excess pressure in casing annular spaces, which can be accompanied by a buildup of gas. The gas composition can be analyzed to help identify the source of the leak. Casing pressure limits should be established. Guidelines are provided in API RP 90, *Recommended Practice 90, Annular Casing Pressure Management for Offshore Wells*, which can also be used for onshore wells. Other methods to monitor well integrity include conducting a casing inspection log and inspection of tubulars for corrosion.

2.8.2. Best and Alternative Practices for Well Stimulation Fluids

2.8.2.1. Reuse Produced Water for Well Stimulation

Produced water from oil fields is often pumped back into the oil-bearing formation to enhance oil recovery, maintain reservoir pressure, and mitigate subsidence. In California, produced water that is not reused for enhanced oil recovery is sometimes used for other purposes, such as for cooling or agricultural purposes, typically after treatment. However, reuse of produced water for well stimulation treatments is not routine. Well completion reports filed through mid-December, 2014, indicate that there were only 43 documented instances of oil and gas operators using produced water for well stimulation in California, accounting for about 13% of the water used for well stimulation in 2014. Produced water reuse for well stimulation has been shown to be feasible (e. g. Huang et al., 2005) and is becoming more common across the United States. For example, recycling of wastewater for well stimulation has increased in the Marcellus Shale region: prior to 2011, 13% of wastewater was recycled, and by 2011, 56% of wastewater was recycled (Lutz et al., 2013). Reuse for well stimulation is occurring in Texas, New Mexico, and elsewhere. Given constraints on water supplies and concerns about the adequacy of produced water disposal methods, reuse of oil and gas wastewater for subsequent well stimulation may be an attractive option for operators in California.

Reusing oil and gas wastewater for well stimulation has benefits but also some limitations. Reuse as stimulation base fluid reduces reliance on freshwater supplies and provides a disposal option. Additionally, reuse of wastewater for well stimulation can reduce transportation costs, which can be high if freshwater and/or wastewater must be trucked to and from the site, respectively. An advantage of reusing wastewater for well stimulation is that it does not need to be treated as stringently as if it were to be released into the environment (King, 2012). One of the main challenges with reusing produced water is that there are high concentrations of salts, measured as TDS. Base fluids with elevated levels of TDS can be problematic, because the salts may precipitate in the formation, blocking fractures and reducing formation permeability (Guerra et al., 2011). Removal of TDS typically requires desalination, which often entails extensive pre-treatment to remove organic chemicals that interfere with desalination (e.g., causing biofouling of membrane surfaces). A bench-scale test in New Mexico, however, demonstrated that high-TDS water can be used as a base fluid for cross-linked gel-based hydraulic fracturing fluids (Lebas et al., 2013), eliminating the costly use of RO.

2.8.2.2. Use Alternative Water Supplies for Stimulation Fluids

While most oil and gas operators use freshwater as a base fluid for well stimulation, operators can employ other water sources, such as brackish water or treated municipal wastewater. These alternative water supplies can reduce the use of limited freshwater resources for oil and gas production. For example, Nicot et al. (2012) reports that brackish water accounts for about 20% of water use in the Eagle Ford Shale and 30% of water use in the Anadarko Basin.[12] There are a few documented cases where recycled water from other municipal or industrial users was used as the base fluid for hydraulic fracturing. Operators in the Haynesville Shale gas play in Louisiana, for example, have used treated wastewater from a nearby paper mill (Nicot et al., 2011). A 2012 analysis found that about 30 municipal and industrial facilities provide water to the oil and gas industry in Texas (Nicot et al., 2012).

Use of alternative water supplies can pose a unique set of risks. First, in water-scarce regions with limited freshwater supplies, use of brackish water may compete with more conventional users who may tap this resource and treat it or blend it for municipal or industrial use (Nicot et al., 2012). Second, in areas where the brackish groundwater aquifer is connected to freshwater aquifers, withdrawing brackish groundwater could compromise the quality and availability of water in the freshwater aquifer (Freyman, 2014). An additional risk associated with the use of brackish water is during its transportation and storage, where a spill of this water could have an adverse impact on the local environment. Challenges with using non-oilfield wastewater include guaranteeing a consistent quality of water and the cost of transporting these waters to the well site. Additional research and analysis is needed to determine whether alternative supplies are available for use in stimulation fluids, and whether the use of these supplies poses any concerns for nearby users, including municipalities, industry, and farmers.

12. Brackish water is generally defined as having a salinity greater than freshwater (TDS <1,000 mg L^{-1}) but less than saline or seawater (~35,000 mg L^{-1}) (USGS, 2014a; NGWA, 2010).

2.8.2.3. Apply Principals of Green Chemistry to Chemical Additives used in Stimulation Fluids

Currently, a large number of chemicals are used in well stimulation that have poor or unknown environmental profiles (Section 2.4). There are few controls on what chemicals are being used in hydraulic fracturing, and some chemicals currently being used are toxic, potentially persistent in the environment, or may degrade to toxic or otherwise environmentally harmful products. Properties such as endocrine effects and carcinogenesis, which complete an environmental profile, are unknown for many chemicals listed in Table 2.A-1.

There are many opportunities to apply green chemistry principles to well stimulation formulations and thereby mitigate many of the potential direct impacts of hydraulic fracturing. The principals of green chemistry include developing industrial processes that use chemicals with the best environmental and health profiles, in other words, industrial processes that use chemicals that are non-toxic, do not have other negative or harmful hazardous properties, do not persist in the environment, and do not degrade to undesirable products (U.S. EPA 2011). Some toxic chemical additives that are used in well stimulation could potentially be replaced by non-toxic alternatives. Ideally, the most toxic and/or persistent chemicals could be replaced first. Determination of alternatives for toxic stimulation chemicals would be beneficial, but there currently is very little incentive for oil and gas producers to employ less toxic additive or to invest in research and development of alternatives.

The sheer number of chemicals used makes a full hazard and risk analysis difficult, if not impossible, due in part to the complexity of understanding interactions between chemicals in combination. Reducing the number of chemicals applied would make it easier to evaluate hydraulic fracturing mixtures, insure public safety, and resolve public concerns. Limiting the number of chemicals that can be used in hydraulic fracturing and acid treatments will also assist and simplify regulation. For example, we identified over 60 different surfactants listed in Table 2.A-1, and it may be possible to limit the number of different surfactants being used without compromising effectiveness. Currently, there is no regulatory incentive for oil and gas producers to minimize the number of chemicals used in well stimulation. However, the American Chemical Society (ACS), in partnership with industry and government representatives, has implemented a Green Chemistry Institute, which aims to address issues of pollution prevention and sustainability in chemical use. More sustainable stimulation chemicals could be pursued within this framework.

Characterization of chemicals—including information on toxicity and environmental persistence—is not required prior to use of these chemicals for well stimulation in California. In some cases, data are missing that are needed in the event of an emergency. Recent events associated with the energy industry have underscored some of the risks of a lack of readily (and publicly available) information on chemicals. For example, emergency response to the release of 4-methylcyclohexanemethanol into the Elk River in

West Virginia was hampered by the absence of basic physical, chemical, and toxicological information on that chemical. In the absence of a complete environmental and health profile on a chemical, implementation of a timely and appropriate response by regulatory agencies following releases of these chemicals into the environment is impeded.

The North Sea compact/OSPAR Convention is a good model for how oil and gas production can be done with an eye towards environmental sustainability. In the compact, it is agreed that chemicals will be tested before they are used in the North Sea. The chemicals must pass certain criteria before they are used, and standards for environmental persistence and acute toxicity must be met (OSPAR Commission, 2013). Similar criteria concerning testing for toxicity and environmental persistence are suggested, but not required, in the United States (U.S. EPA, 2011). In another example, Proctor & Gamble established the Environmental Water Quality Laboratory (EWQL), with the mission to measure the toxicity, environmental fate, and physical-chemical properties of chemical ingredients before they were used in their products (http://www.scienceinthebox.com / leadership-in-sustainability-at-pg). These approaches may represent a good model for insuring the safety of unconventional oil and gas development in California.

2.8.2.4. Investigate Application of Waterless Technologies

Companies are developing technologies to reduce or eliminate the amount of water used for well stimulation. Some low-water or waterless stimulation methods have been in use for decades. Alternatives include the use of foams; pressurized gas, such as carbon dioxide or nitrogen; or fluids other than water, such as liquid propane (see e.g., Friehauf and Sharma, 2009; Gupta, 2010; van Hoorebeke et al., 2010). A recent magazine article cites the case of the Marathon Oil Company, which has begun using propane for fracturing in the Eagle Ford Basin in Texas. The company's president stated during testimony to a Congressional committee that the move to waterless fracturing has reduced water consumption by 40 percent in the first 90 days of operations, and as an additional benefit, "The companies are able to resell the propane when it comes up back from the hole" (Wythe, 2013). One industry analyst cautioned, however, that waterless technologies are not poised to have a large effect on water use in the oil and gas industry, barring a major technological breakthrough (Freyman, 2014).

2.8.3. Best and Alternative Practices for Wastewater Characterization and Management

2.8.3.1. Treat and Reuse Oil and Gas Wastewater for Other Beneficial Uses

With proper treatment and monitoring, wastewater generated from oil and gas production—including wastewater generated from stimulated wells—could be used for various beneficial uses. Guerra et al. (2011) identified several beneficial uses currently being practiced in the western United States, including industrial cooling, dust control, irrigation, and water supply to constructed wetlands and wildlife habitats. The advantages of reusing oil and gas wastewater are that the demand for freshwater resources is reduced, and since water is typically treated to remove contaminants prior to reuse, the risk of water contamination from improper disposal is reduced, and the total volume of wastewater produced is reduced. However, the reuse of produced water that is commingled with returned stimulation fluids raises new concerns, since it is not known how stimulation fluid additives may impact the safety of beneficial reuse. The types and amounts of well stimulation additives found in these waters is unknown, so it is not certain what treatment methods are adequate to allow reuse. Additionally, potentially hazardous chemicals resulting from degradation of the added chemicals and the interaction of the stimulation fluid with the formation need to be carefully evaluated.

Proper treatment is required to ensure that well stimulation chemicals are removed from wastewater prior to reuse. In Section 2.5, we evaluated whether various chemical, physical, and biological treatment technologies commonly used on produced water in California and elsewhere will be effective in removing well stimulation chemicals. Results of this analysis indicate that there is no single treatment technology that can independently treat all categories of well stimulation fluid additives (also see Appendix 2.C). Adequate treatment would require the use of multiple technologies in treatment trains to satisfy effluent requirements. Treatment trains that provide only the most basic treatment, e.g., air stripping/gas flotation followed by filtration, will be ineffective at removing most well stimulation chemicals. Treatment trains utilizing RO are expected to provide the highest level of treatment, due to the effectiveness of RO at removing small (0.001-0.0001 μm) constituents and the need for multiple pretreatment steps to prevent membrane fouling. However, the high cost and energy requirements of RO systems may reduce the economic viability of treating well stimulation chemicals.

2.8.3.2. Characterize and Monitor Produced Water and Other Wastewaters

More extensive characterization of the compositions of wastewater generated by stimulated wells in California is needed. Additional testing needs to be done for wastewater that is not being disposed into injection wells, especially to see if wastewater that is being reused for irrigation, disposed into sewers or unlined pits have been effectively treated. Wastewater compositions should be analyzed at several time points to be able to identify the patterns for how they evolve over time, and to identify when returned stimulation fluids are present in the wastewater. Analytes should include surfactants, solvents, biocides, and other compounds used in hydraulic fracturing fluids. Other analytes to be measured should include general water quality parameters (such as pH, temperature, chemical oxygen demand, organic carbon etc.), major and minor cations and anions, metals and trace elements, BTEX, gases (methane and H_2S) and NORM. The list of analytes needs to be periodically updated to reflect current scientific research, as well as understanding of the wastewater composition patterns in California oil and gas fields where stimulation is occurring.

2.8.3.3. Improve Management Practices for Oil and Gas Wastewater

Disposal of wastewater from oil and gas production occurs by Class II disposal wells, discharge into sanitary sewers, percolation in unlined pits, and treatment for reuse. Evaporation-percolation in unlined surface impoundments (percolation pits) is a practice that intentionally introduces wastewater and its constituents into near-surface groundwater aquifers. The U.S. Department of Energy recommends that “all evaporation pits should be lined … to prevent downward migration of fluids” (U.S. DOE et al., 2009). Texas and Ohio have restricted or stopped the use of unlined pits and percolation basins as a disposal practice for produced water, due to documented groundwater contamination incidents (Kell, 2011). Given the concerns regarding disposal in percolation pits, injection into properly located, constructed, and permitted Class II wells for EOR or disposal would be a better practice (Kell, 2011; U.S. DOE et al., 2009). The reuse of wastewater should be encouraged, but reuse of water from stimulated wells will require adequate safeguards, including monitoring for appropriate chemical contaminants and applying multi-stage treatment systems before reuse (e.g., Liske and Leong, 2006; Appendix C).

When oil and gas wastewater is discharged into sanitary sewers, the wastewater is conveyed to domestic wastewater treatment plants that were not necessarily designed to remove all of the constituents found in oil and gas wastewater from stimulated wells. Although the discharges into the sanitary sewer must be compliant with local pre-treatment ordinances, it is not clear that these requirements are sufficient to address well stimulation chemicals.

The environmental impacts of discharging oil and gas wastewater into Class II wells in California are not entirely understood. There are federal and state requirements for construction and placement of Class II injection wells (Veil et al., 2004), but there are concerns that Class II wells in California may be contaminating protected groundwater. Site characterization requirements include a confining zone free of known open faults or fractures that separates the injection zone from underground sources of drinking water, and construction requirements to ensure mechanical integrity of the well (40 CFR 146.22). There are also operating requirements that limit injection pressure and monitoring and reporting requirements (40 CFR 146.23). A recent detailed review of California requirements for Class II injection wells suggested that current rules may not be adequate for protection of all beneficial uses of groundwater (Walker, 2011). In addition, EPA is expected to release (in 2015) recommendations for best practices for limiting induced seismicity associated with wastewater injection by the oil and gas industry (Folger and Tiemann, 2014). An alternative practice would be to determine the location of protected groundwater in the state, to investigate and review current practices to resolve outstanding issues concerning the use of Class II wells for disposal in California, and to conduct site-specific studies to ensure the safety of proposed disposal methods.

2.8.4. Best and Alternative Practices for Monitoring for Groundwater Contamination

Groundwater contamination can be difficult to detect. Comprehensive baseline and monitoring measurements collected before and after drilling, including regional characterization of background concentrations of groundwater constituents, are necessary to determine impacts on groundwater quality from well stimulation or any other oil and gas development activity.

Baseline data on groundwater quality have not been collected at appropriate locations and in a systematic manner to allow the impacts of oil and gas development on groundwater resources in California to be determined. Improved collection and organization of groundwater data would be a better practice. Some information on background levels of many inorganic and organic constituents, including TDS, trace metals, and VOCs in California, is available from the USGS Groundwater Ambient Monitoring and Assessment (GAMA) program (USGS, 2013). These data should be fully analyzed in future investigations of the impact of well stimulation on groundwater quality in California. However, the GAMA program has objectives related to monitoring drinking water and does not currently collect data in many regions of the state with active oil and gas development (Figure 2.7-1). Investigations of regional and site-specific groundwater impacts from unconventional oil and gas development should be directed at determining the importance of specific contamination pathways, and the extent of groundwater contamination. Developing specific programs examining groundwater impacts of oil and gas development would be a better practice.

In other parts of the country, studies have shown that measurements of methane in groundwater and elsewhere can be an important indicator of leakage from well bores and other sources, such as fractures. Methane levels over 45 mg L^{-1} (ppm) have been observed in New York, West Virginia, and Pennsylvania groundwater (Vidic et al., 2013). Best practice for the development of a comprehensive groundwater monitoring program includes coordinated examination of the concentrations and isotope characteristics of methane.

The State Water Resources Control Board (SWRCB) is issuing groundwater monitoring regulations, due to take effect on July, 2015. The groundwater monitoring regulations being developed by the SWRCB will include both a monitoring plan for areas where oil and gas well stimulation are being conducted, as well as a regional monitoring plan. The SWRCB released its draft model criteria for area-specific groundwater monitoring on April 29, 2015 (SWRCB, 2015), which outlines the design for groundwater monitoring, including collection of baseline data, as well as sampling and testing requirements. These monitoring requirements are expected to develop baseline water quality information and improve the current understanding of water quality impacts of both conventional and unconventional oil and gas development.

2.9. Data Gaps

Numerous data gaps were identified during the course of this investigation that can and should be addressed in order to provide a better understanding of unconventional oil and gas development in California, and associated impacts on water and the environment. Overall uncertainty in our analysis was increased by reliance on voluntary reporting, poor data quality, and missing or inaccurate information in state agency datasets. New regulations, put in place under SB 4, are mandating reporting of more information, but an evaluation of the completeness and accuracy of reporting, as well as the relevance and appropriateness of information being reported, needs to occur in the future as part of the ongoing efforts to fully understand the actual and potential environmental impacts of unconventional oil and gas development. Data that are complete and accurate also need to be submitted and published in a timely manner. Scientists and regulators need to be engaged in an ongoing effort of data analysis and interpretation of information, to arrive at a better understanding of the environmental impacts of well stimulation in California. Below, we identify some of the most critical data gaps identified in our investigation of water impacts of well stimulation.

2.9.1. Reports and Data Submissions Have Errors, Missing Entries, and Inconsistencies

Mandatory and voluntary reporting requires data entry by operators and other responsible parties. It was apparent during our investigations that information submitted to the state was not subject to systematic quality checks or verified, and, as a result, datasets resulting from these submissions contained errors and inconsistencies. Due to data entry errors and inconsistencies, data sets required extensive editing and organization before they could be analyzed. Analysis of uncorrected data can and will result in significant errors in interpretation (e.g., chemical function is routinely reported incorrectly, counts on the number of chemicals may be exaggerated, etc.). Maintaining standardized and verified data, ideally in electronic format, would allow rapid and accurate analysis of oil field activities on a near real-time basis.

In many cases, the data collected by DOGGR and other government agencies contained simple typos and other obvious mistakes. In other cases, information is missing or meaningless. For example, DOGGR's Production and Injection database contained records for active production wells where the number of production days was zero and the information on the type of produced water generated was missing or identified as "other" or "unknown."

Reporting units and other formats differ between important databases (e.g., FracFocus, SCAQMD, DOGGR), complicating comparative analysis and making data integration more difficult and prone to error. In the SCQAMD reports, units for reporting mass compositions of fluids were non-standard and resulted in predictable data entry errors. The SCQAMD data entry requirements are different from both FracFocus and DOGGR records, and basic information such as CASRN and API well number are entered in different formats or not at all. FracFocus is not linked or standardized to other information, such as well production information, collected by DOGGR and other agencies.

Implementation of a quality assurance program and standardization would improve the quality of the data and allow ongoing analysis by agencies compiling the data. For example, in the completion reports submitted to DOGGR, it could be required that the percentage of various chemicals reported as added to each operation must always add up to 100% (± 5%). In other cases, simple controls, such as checking that entries match an appropriate range of possible values, would results in marked improvements in data quality. The use of entries such as "other" or "unknown" should not be acceptable for critical parameters or values.

The DOGGR GIS wells file has missing data for many data entry fields, which are needed for assessment of impacts. For example, as of November 2014, only 20% of records have values filled in for well depth. There are also incorrect data for some values; for example, there are some wells that have a latitude or longitude value of zero.

There are also some files where the data is poorly organized, making analysis cumbersome. For example, in the new completion reports, the "Location of Treatment" sheet does not have the actual location of where the stimulation was conducted (such as fields for latitude, longitude, field, area, or county). Instead, this information is located in a different sheet in the file that is intended to list all the chemicals used in each treatment. DOGGR and other agencies should consider normalizing data spreadsheets, and preferably storing the data in an accessible database.

2.9.2. Information is Not Easily Accessible to the Public

Agencies responsible for collecting information do not always make the information easily accessible to the public, limiting the use of these records to inform citizens and policymakers. The use of the industry website FracFocus is a reasonable model for inputting chemical data, but extracting data is difficult, and accessibility to electronic datasets or databases is limited and not freely available to the public. Information on water quality and the location of groundwater extraction wells in GAMA is not reported with appropriate or accurate location information (latitude and longitude or Universal Transverse Mercator [UTM] coordinates) to allow open and public risk analysis. Additionally, lack of publication of well locations hinders the development and public evaluation of monitoring plans that must be submitted under new regulations.

2.9.4. Information is Submitted in Inadequate Data Formats

In many cases, data needed for analysis are only available as PDF documents or displayed on web pages, rather than available in well-organized electronic data structures. The nontransferable nature of the datasets makes data entry and analysis burdensome and time-consuming, as records need to be retyped or extracted from PDF documents. The use of non-standard data formats and the lack of a well-designed database system may have also resulted in a decreased ability to detect errors in data submission, resulting in incorrect entries, typos, and duplicate records. The use of PDF formats for data reporting is an important problem for reporting all types of data.

2.9.4. Poor Collaboration Between State and Federal Data Collection Efforts

In collecting information for this project, we found that datasets collected by different agencies were frequently contradictory, lacked standardization between datasets (e.g., reporting units differed, etc.) and difficult to harmonize. There are currently separate initiatives by the Central Valley Regional Water Quality Control Board, the South Coast Air Quality Management District, DOGGR, and other agencies to collect information, with each agency having its own purposes. The lack of collaboration and standardization between agencies resulted in duplicated efforts.

In many cases, stimulation events were described differently in different databases. For example, we found data for the same well stimulation operation that was reported in FracFocus, in DOGGR completion reports, and in data submitted to the SCAQMD, but these sources sometimes reported different dates, water volumes, and other information that made comparison or integration of information from different sources difficult. Coordinated integration of data collection, standardization of reporting units, and consistent unique identifiers for authorized treatments would require a new level of interdepartmental coordination and cooperation, but would allow improved regulatory oversight. The unique API well numbers should be included with all reports, data, and other documents concerning activities associated with wells or groups of wells (e.g., wastewater management activities).

2.9.5. Chemical Information Submitted by Operators is Incomplete or Erroneous

Chemical data submitted by operators includes errors and omissions. The product CASRN and chemical name are not always included for each chemical reported. Frequently, the chemical purpose is incorrect or missing. Chemicals that are classified as trade secrets, confidential business information, or used in proprietary blends are listed without CASRNs. Products listed without CASRNs cannot be definitively identified by chemical name alone, and thus cannot be adequately evaluated for hazards, fate, and treatment. Even when CASRNs are provided, they are not always correct. For example, chemical CASRNs are sometimes reversed or missing digits altogether (see comments about quality control above). Frequently, the reported chemical purpose includes all possible uses for the chemicals, to the point that the information provided is meaningless. Furthermore, impurities are typically not identified as such, and are instead given the same purpose description as the active ingredient in the chemical product. Hazard and environmental analysis of chemicals used in stimulation fluids is hindered by the lack of quality control and standardization in reported data.

2.9.6. Chemicals Lack Data on Characteristic Properties Needed for Environmental Risk Analysis

Most of the chemicals being used for well stimulation lack publicly available physical, chemical, or toxicological measurements needed for the development of an environmental profile. An environmental profile is needed to provide a complete hazard and risk assessment on a chemical (OECD, 2013; OSPAR, 2013; Stringfellow et al., 2014; U.S. EPA, 2011). At a minimum, the physical, chemical, and biological information needed to develop an environmental profile includes log octanol-water partition coefficients (log K_{ow}), Henry's constants (K_H), soil organic carbon-water partition coefficients (K_{OC}), biodegradability, and acute toxicology. Other information on chronic effects, potential for bioaccumulation, and other properties are also needed. The technical information in an environmental profile is needed for developing environmental fate and transport models, reviewing waste management plans, preparing for spills and accidents, selecting treatment technologies, evaluating reuse projects, and conducting hazard assessments.

Chemical data generated by industrial groups are sometimes contained in material safety data sheets (MSDS); however, these data are not always publicly available and cannot always be confirmed or reviewed. Material safety data sheets cannot be considered reliable sources for chemical, physical, and toxicological data without a public review and validation of the published information.

Publicly available experimental data on the toxicity of many stimulation chemicals to aquatic species, including algae and aquatic animals, and mammalian species are sparse. In particular, aquatic toxicity data are missing testing of native or resident species that are important to California. Measurement or publication of aquatic and mammalian toxicity data is currently not required prior to using chemicals in well stimulation. This lack of available data increases risk to human and environmental health, since the lack of information prevents the ability to make informed decisions and apply an appropriate response during failures and accidents.

In addition to a basic analysis of acute toxicity, data is needed on the potential impacts of chronic exposure to well stimulation fluids in ecological receptors. Measurements of sublethal impacts on plants and animals, such as survival potential and population viability, are not available for most chemicals used in well stimulation. More data is needed on potential sublethal impacts on ecological receptors due to exposures to fluid additives.

The fate and transport of chemical mixtures in the environment is not well understood. Hydraulic fracturing fluids contain complex mixtures, and the interactions of these chemicals in the environment is unknown. For example, easily degradable but toxic components such as methanol are in admixture with biocides, added to prevent biodegradation from occurring. How biocides would influence the persistence of methanol in the environment is unknown, but the methanol might transport further in groundwater in the presence of the biocide, presenting greater risk than methanol alone. Scientific investigation of the environmental fate of chemical mixtures is needed.

2.9.7. Data on Chemical Use from Conventional Oil and Gas Operations are Not Available

Chemical use information for all oil and gas development operations is not available and would be useful for providing context to chemical use during well stimulation. SCAQMD is now collecting data on chemical use during well drilling, installation, and rework in parts of southern California, but similar data are not available for the San Joaquin Valley where the majority of oil and gas extraction takes place. To our knowledge, no data is being collected on chemical use during other oil and gas development activities, such as EOR. Data collected by SCAQMD do not carefully differentiate between well stimulation treatments and other activities, such as well maintenance, making it difficult to interpret and evaluate well stimulation chemical use in the context of overall chemical use. Many of the same chemicals (e.g., biocides, corrosion inhibitors, surfactants, etc.) are used for other oil and gas development activities as are used in production aided by well stimulation. More complete and consistent reporting and tracking of chemical use for all oil and gas development activities will allow a better understanding of the impacts of well stimulation in the context of overall oil and gas development.

2.9.8. Lack of Data Regarding the Chemical Composition of Produced Water from Stimulated Wells

There is a lack of information regarding the characteristics of produced water and other wastewater generated from well stimulation in California. Produced water from stimulated wells will contain chemicals used in hydraulic fracturing, but the amounts of chemicals returning during production and the time period over which they return has not been measured. Data are needed regarding how wastewater constituent concentrations and composition change over time.

Produced waters will contain reaction products from the complex mixtures of chemicals used in hydraulic fracturing. Lack of knowledge concerning the fate of the injected stimulation fluids in the subsurface, and the potential for them to be transformed, or to mobilize formation constituents over the lifetime of production from the well, needs to be determined. The nature of the reaction byproducts, the amounts and types of materials returning to the surface during the lifetime of the well, and hazards associated with these reaction byproducts are entirely unknown and need to be investigated.

Poor understanding of wastewater composition is a major impediment to the safe and beneficial reuse of produced water from stimulated wells. It is unknown how (or if) well stimulation chemicals or their byproducts have been introduced into the environment via disposal or reuse practices, such as percolation or water flooding. California specific investigations of water reuse and disposal practices are needed to fill this data gap.

There are limited data concerning the composition of produced waters from conventional wells, which prevents a comparison between the conventional and unconventional oil and gas development. Current practice in California mingles the produced waters from stimulated and non-stimulated wells before treatment. If there are differences between wastewater from conventional and stimulated oil and gas operations, the differences would have implications for how each wastewater should be handled, treated, and disposed. Previous studies on the chemical quality of produced waste in California were conducted decades ago, and new studies need to be conducted characterizing produced water and other oil and gas industry wastewaters in California.

Water quality analyses required under new regulation and submitted to DOGGR with well completion reports do not typically measure specific stimulation chemicals, with the exception of a total carbohydrate test for guar. Analysis is not conducted for major well-stimulation-fluid components of concern, such as biocides or surfactants, or potentially harmful reaction products that may form within the formation following introduction of the stimulation fluids. The operators also do not report the exact time at which the recovered fluid sample was collected relative to the stimulation event, so it is difficult to interpret what the samples truly represent.

2.9.9. Incomplete Information Regarding Wastewater Management, Disposal, and Treatment Practices

Data on wastewater disposal and management are incomplete. There is conflicting or inadequate information on current disposal and reuse practices, especially concerning percolation pits and Class II wells. Cradle-to-grave documentation on wastewater management would allow individual sources of wastewater, such as individual wells, to be related to a specific disposal or reuse site, such as a percolation pit.

Systems for documentation of wastewater management practice need modernization, and ambiguous or uninformative entries should not be allowed. For example, the third most common disposal method reported by operators was “other.” DOGGR staff confirmed that some operators are using the “other” category to describe disposal that is, in fact, included in some of the other categories—for example, subsurface injection, discharge to a surface water body, disposal to a sanitary sewer system, etc. (Fields, 2014). Some disposal methods—such as reuse for irrigation or groundwater recharge—are not included as separate categories in the DOGGR production/injection database. During meetings held as part of this study, some operators have suggested that their current practices are not consistent with the data they have reported to DOGGR. Insufficient quality control for operator-submitted data, and inadequate categories for wastewater disposal methods, result in an incomplete picture of current wastewater disposal practices.

There is no central resource for data concerning wastewater treatment practices. In collecting information for this project, data sources for confirmation of common treatment practices varied from NPDES permits and government agency reports, to personal communications, brochures, and factsheets. Due to the lack of a centralized data resource, the frequency of specific wastewater treatment practices and overall trends are unknown.

2.9.10. Incomplete Information on the Impacts of Contamination from Subsurface Pathways

Subsurface pathways and mechanisms are difficult to characterize, and information concerning potential groundwater contamination from hydraulic fracturing is very limited. Peer-reviewed studies investigating the possibility of contaminant transport due to fracturing operations have not been conducted in California. Studies conducted in other areas have suggested contamination is possible or has occurred, but the applicability of those results to California cannot be determined without more investigation, due to the unique conditions existing in California.

2.9.11. Lack of Accurate Information Regarding Old and Abandoned Wells

The extent to which abandoned and deteriorating wells may present a hazard in California needs to be assessed. Documentation of the location, construction, and the method of abandonment for currently unused wells are required before assessment of hazards (or methods for remediation) can be performed. DOGGR has a program that requires operators to conduct regular testing of idle wells to ensure that they are not impacting surface and groundwater, but similar testing is not required for abandoned or buried wells. The datasets regarding idle wells are inconsistent. For example, the DOGGR GIS wells file lists 13,450 wells as idle, but another “Idle Wells” file on the DOGGR website lists a total of 21,347 wells as idle.

2.9.12. Lack of Knowledge about Fracture Properties in California

The process of fracture creation and propagation is currently an area of active research, with the bulk of the work focusing on the properties of gas shales in states other than California. This research applies to deep formations and thus evaluates pathway formation scenarios over large vertical distances. Fracturing has been practiced in California for decades (Walker et al., 2002), but fundamental studies of fracturing behavior, fracture propagation, and the orientation of fractures relative to reservoir depth for California geology are lacking. Fully understanding this behavior is particularly important in California due to the possibility of relatively shallow fracturing depths (200–300 m [650–1,000 ft] from surface) compared to other regions using hydraulic fracturing technology.

Although the reporting of the extent of stimulation geometry has been required for operations occurring after January 1, 2014, the resulting data assessed for this report indicates it generally does not regard the extent of fracturing from single stages, limiting what can be discerned about fracture geometry from these data. Some of the reported data are obviously inaccurate (for example, some of the wellbore end depths are shallower than the corresponding wellbore start depths) or inconsistent with reporting requirements (for example, wellbore start depths are sometimes reported as zero instead of the start of the stimulated interval within the wellbore). Further, if data regarding fracture geometry were reported, the accuracy of this data would be unknown unless the data supporting the estimates of fracture geometry, and the methods used to analyze the supporting data, were reported by operators.

2.9.13. Incomplete Baseline Data and Monitoring Studies for Surface and Groundwater

Long-term monitoring and studies of surface and groundwater in oil and gas producing regions of California are needed to determine if groundwater resources have been impacted. There is a lack of information on the quality of surface or groundwater near stimulated oil fields, and baseline (or up-gradient) data collection is needed. Significant data gaps exist regarding current knowledge of groundwater quality in California, including the location and extent of protected groundwater that contains less than 10,000 mg L^{-1} TDS. Concentrations of methane, trace metals, NORM, and organic chemicals in groundwater in oil and gas producing regions are unknown, and are needed to assess impacts of unconventional oil and gas development. New regulations implemented under SB 4 and other programs are beginning to address this data gap. The effectiveness of these regulations needs to be evaluated in the future.

2.9.14. Lack of Information on Spills

As discussed above for other types of data, there are numerous inconsistencies between agencies concerning the information collected on spills and accidental releases in California. Databases maintained by OES and DOGGR on surface spills and leaks associated with oil and gas production often do not agree, increasing uncertainty in our understanding of environmental impacts from accidents. Inconsistencies exist concerning the number of spills that have occurred and details regarding those spills. This discrepancy is likely due in part to the fact that OES sends spill reports electronically to DOGGR, and then a subset of the information is entered into DOGGR's database. Although OES is responsible for collecting spill information and submitting it to the appropriate agencies, there are spills in DOGGR's database that are not in OES's. Similarly, there are oil and produced water spills in the OES database that are not in the DOGGR database. DOGGR often coordinates with operators after spills—especially for large spills or when spills impact waterways—but there is no mechanism for conveying this information back to OES. Operators often submit corrections to OES after a spill takes place, and these corrections are not always entered into either DOGGR's database or the OES database that is available online. Another major concern is that DOGGR only captures information on oil and produced water spills, and therefore does not have record of spills associated with chemicals used for oil and gas production.

2.10. Main Findings

2.10.1. Water Use for Well Stimulation in California

1. We estimate that well stimulation in California uses 850,000 to 1,200,000 m^3 per year (690–980 acre-feet) of water. Our estimate is based on a combination of data sources to provide a best estimate that reflects the uncertainty in both (a) the number of operations that are occurring, and (b) how much water each operation uses on average.

2. Operators obtained the majority of water needed for well stimulation from nearby irrigation districts (68%), produced water (13%), operators' own wells (13%), a nearby municipal water supplier (4%), or a private landowner (1%).

3. Hydraulic fracturing has allowed oil and gas production from some new pools where it was not previously feasible or economical. We estimate that freshwater use for enhanced oil recovery in fields where production is *enabled* by stimulation was 2 million to 14 million m^3 (1,600 to 13,000 acre-feet) in 2013. By comparison, freshwater use for enhanced oil recovery in *all* oil and gas fields was 13 million to 44 million m^3 (11,000 to 36,000 acre-feet) in 2013.

4. Local impacts on water usage appear thus far to be minimal, with well stimulation and hydraulic-fracturing-enabled enhanced oil recovery accounting for less than 0.2% percent of total annual freshwater use within each of the state's planning areas, which range in size from 830 to 19,400 km^2 (320 to 7,500 mi^2). However, well stimulation is concentrated in water-scarce areas of the state, and an increase in water use or drawdown of local aquifers could cause competition with agricultural, municipal, or domestic water users.

2.10.2. Characterization of Well Stimulation Fluids

1. Records describing the chemical composition of hydraulic fracturing fluids between 2011 and 2014 were voluntary, and represent one-third to one-fifth of the total hydraulic fracture treatments thought to have occurred in California during that period.

2. Over 300 different chemicals or chemical mixtures were identified as having been used for hydraulic fracturing in California. Of the disclosed chemicals, approximately one third of the chemical additives lacked a CASRN, and therefore any enumeration of the number of chemicals used in hydraulic fracturing should be considered approximate.

3. Information on chemical use during acid stimulation treatments is very limited. Analysis of regional data and data collected as part of new mandatory reporting requirements in effect since January 2014, identified over 70 individual chemicals or chemical mixtures used during acid treatments, approximately one-third of which were different from chemicals used in hydraulic fracturing.

4. Over 60 chemical additives with a median usage of 200 kg (440 lbs) or more per treatment were found. At least nine of these compounds are proppants, and many are solvents, crosslinkers, gels, and surfactants. Since these compounds were used in significant amounts, they are considered priority compounds for characterization of their hazards and risks.

5. Almost two-thirds of the chemicals reported to be used in hydraulic fracturing or acid treatments did not have publicly available information allowing an assessment of environmental toxicity. Environmental profiles need to be developed for these chemicals.

6. Thirty-three chemicals have a GHS ranking of 1 or 2 for at least one aquatic species, suggesting they could present an environmental hazard if released to surface waters.

7. Significant data gaps exist concerning the hazard, toxicity, and environmental persistence of chemicals used in well stimulation. Additionally, over 100 of the reported materials used for well stimulation are identified by non-specific name and reported as trade secrets, confidential business information, or proprietary information. These materials cannot be evaluated for hazard, risk, and environmental impact without more specific identification.

8. A full understanding of the environmental risk associated with unconventional oil and gas development will require a full disclosure of the chemicals used and better understanding of the environmental profile of each chemical. Environmental profiles include an understanding of a chemical's toxicity, transport properties, and persistence in the environment. A formal environmental review process for all chemicals and chemical mixtures, such as the EPA Design for the Environment program, is recommended.

9. Methods for the detection of chemical additives, their byproducts, and degradation products in environmental samples need to be developed. Many of the chemicals being used do not have standard methods of analysis.

2.10.3. Wastewater Quantification, Characterization, and Management

1. Produced water, recovered fluids, and other wastewaters from stimulated wells will contain chemicals from hydraulic fracturing fluids and their reaction byproducts, but the concentrations of these chemicals in wastewaters will change over time and have not been fully characterized.

2. Produced water, recovered fluids, and other wastewaters from stimulated wells will also contain various other contaminants in dissolved substances from waters naturally present in the target geological formation, substances extracted or mobilized from the target geological formation, and residual oil and gas.

3. During hydraulic fracturing, recovered fluids that are captured before production represent a small fraction of the injected fracturing fluids (~ 5%). In contrast, recovered fluid volumes for acid treatments tend to be a higher percentage of the injected fluid (50–70%), but data on acid fluid recovery is limited and may not be representative.

4. Recovered fluid volumes are a small fraction of wastewater generated within the first month of production. These results indicate that studies from other regions of the country showing significant recovery of “flow-back” fluids have limited application to California.

5. Recovered fluid samples from stimulated wells have been shown to contained high concentrations of salts, trace elements (arsenic, selenium, and barium), naturally occurring radioactive materials, and hydrocarbons. Carbohydrates (gels) were detected in some recovered fluid samples, and this suggests that other stimulation chemicals may also be present. In contrast, produced waters from stimulated wells have not been characterized.

6. Recovered fluids are typically stored in tanks at the well site prior to disposal. According to well completion reports filed and posted through December 2014, more than 99% of recovered fluids are injected into Class II disposal wells. A small amount (less than 0.3%) of the recovered fluids are recycled.

7. The net produced water volumes generated in the first five months of production from stimulated and non-stimulated wells were not substantially different, although their distributions were different. There results suggest there are few differences in the volume of water produced from conventional and unconventional wells, but that some further investigation of these issues could be warranted.

8. There is a lack of information regarding the mass of stimulation fluids recovered after treatment. The concentration of returned stimulation fluids and their reaction byproducts in produced water over time needs to be investigated. The fate of the injected stimulation fluids in the subsurface, and the potential for them to be transformed, or to mobilize formation constituents over the lifetime of the production of the well, needs to be determined.

9. From January 2011 through June 2014, it has been reported that nearly 60% of the produced water from stimulated wells was disposed of by evaporation-percolation in unlined pits. An estimated 36% of the active unlined pits in California are operating without the necessary permits from the Central Valley Regional Board.

10. Subsurface injection in Class II wells, for disposal or enhanced oil recovery, was the second most commonly reported disposition method for stimulated wells in California, accounting for approximately 25% of the produced water from stimulated wells.

11. The impacts on the environment of common disposal practices for produced water that may contain stimulation fluids, including percolation pits and well injection, are poorly understood.

12. Information on current treatment and reuse practices for all wastewater from oil and gas operations in California is limited. Available data suggest that simple treatment technologies (e.g., oil-water separation, water softening, gravity separation, and filtration) are predominantly being used for produced water in California. More complex treatment trains—capable of removing an extensive array of chemicals—are used sporadically.

2.10.4. Contaminant Release Mechanisms, Transport Pathways, and Impacts to Surface and Groundwater Quality

1. Several plausible release mechanisms and transport pathways exist for surface and groundwater contamination associated with onshore well stimulation in California. They are depicted in Figures 2.6-1 and 2.6-2, and summarized in Table 2.6-2.

2. Release mechanisms and transport pathways of high priority for the state are percolation of wastewater from disposal pits; injection of produced water if conducted into protected aquifers; reuse of produced water for irrigation; disposal of produced water into sewer systems; potential leakage through abandoned wells; and potential leakage through fractures.

3. Some of the release mechanisms that were identified are primarily relevant to California, and are uncommon elsewhere, including use of percolation as a disposal method and reuse of produced water for irrigation.

4. Percolation pits provide a direct pathway for the transport of produced water constituents, including returned stimulation fluids, into groundwater.

5. With proper siting, construction, and maintenance, subsurface injection using properly sited Class II wells is less likely to result in groundwater contamination than disposal in unlined surface impoundments.

6. There is growing interest in expanding the beneficial reuse of produced water for agriculture, particularly for irrigation. The use of produced water from unconventional production raises specific or unique concerns. Treatment and reuse of produced water from fields with stimulated wells should include appropriate monitoring and treatment before reuse for irrigated agriculture.

7. According to completion reports, fracturing occurs at shallower depths in California than is typical for other regions of the country. In approximately one-half of the operations, fracturing may extend to depths less than 300 m (1,000 ft) from the surface. The shallow depths of fracturing, combined with the deep groundwater aquifer in the Central Valley, raise concern that fractures may intercept protected groundwater resources. Additional research is needed to determine how often this occurs, if at all, and the consequences if it does occur.

8. Determining where fractures occur is an important component of determining exposure pathways. The reliability of models used by industry to estimate a fracture zone (axial dimensional stimulation area) should be determined.

9. In studies conducted elsewhere, water contamination associated with well stimulation has been documented in some places, but several studies have not found any contamination due to stimulation. No incidents of groundwater contamination due to stimulation have been noted in California to date, although there has been very limited monitoring conducted to detect any water quality impacts.

10. There is a lack of information on the quality of surface or groundwater near stimulated oil fields. Baseline data collection prior to stimulation has not been required in the past. No cases of contamination have yet been reported, but this may be primarily because there has been little to no systematic monitoring of aquifers in the vicinity of oil production sites.

11. Significant data gaps exist regarding current knowledge of groundwater quality in California, including the location and extent of protected groundwater that contains less than 10,000 mg L^{-1} TDS. Concentrations of methane, trace metals, NORM, and organic chemicals in groundwater in oil and gas producing regions are unknown. New regulations implemented under SB 4 and other programs are beginning to address this data gap. The effectiveness of these regulations needs to be evaluated in the future.

2.11. Conclusions

This chapter represents a review and analysis of what is currently known about well stimulation technologies in relation to water resources and the water environment. The quantity of water being used for well stimulation is relatively small and local impacts of water usage appear thus far to be minimal. Well stimulation accounts for less than 0.2% percent of total annual freshwater use within each of the state’s planning areas. Water use for well stimulation, however, is occurring in water-scarce regions and, given the critical availability of water in these areas, could reduce the water available for other uses.

A significant analysis included in this chapter is the identification of the chemicals being used in well stimulation in California. An investigation of the properties of these chemicals shows that many of them are poorly characterized for properties important to determining their hazard and potential impact to the environment. A list of priority stimulation chemicals, requiring further review, was developed based on prevalence of use and toxicity. Additionally, it is apparent that many chemicals are being used that cannot be evaluated for their hazards or potential environmental impact.

The chemical characteristics of produced water generated from stimulated wells in California are largely unknown, however it is apparent that produced water from stimulated wells will contain well stimulation chemicals or their reaction by-products. Under SB 4, chemical data are being collected for “recovered fluids,” but recovered fluids are not representative of returned injection fluids and other wastewater produced over the life of a well. Time-dependent chemical characterization of produced water from stimulated wells are needed to improve management, treatment, and disposal practices. Additionally, mass balance analyses at individual well sites are warranted to clarify the fate of stimulation chemicals remaining in the formation and the quantities of stimulation chemicals in produced water. Geochemical modeling would complement these efforts to characterize chemical fate and transport for stimulated wells.

In California priority potential environmental release mechanisms include disposal of produced water in unlined pits, injection of produced water into potentially protected groundwater, reuse of produced water for irrigation, and disposal of produced water in sewer systems. Unlike in other parts of the country, contamination of water resources due to spills of well stimulation chemicals have not been documented in California, however spills of produced water have occurred. The transport of contaminants through induced fractures to groundwater has not been established, but should be evaluated in California, where fracturing depths are much shallower than in other parts of the country. Other potential subsurface release mechanisms include leakage through compromised wells and leakage through natural subsurface fractures, however the importance of these pathways is also unknown.

In California, no incidents of groundwater contamination due to well stimulation have been documented. Historically, baseline data were not collected on groundwater quality prior to initiating well stimulation activities, making it difficult, and in some cases impossible, to attribute possible contamination to nearby stimulation operations. There has not been a coordinated monitoring program for water resources located in the vicinity of oil and gas fields where stimulation is occurring that could detect or identify sources of contamination.

Application of good practices while conducting well stimulation can reduce impacts from injected or mobilized fluids. Practices such as collection of baseline measurements before drilling, proper well construction, and application of green chemistry principles are advisable. Many significant data gaps were identified. Data collection in many cases is not systematic, of high quality, or well organized. Many of the chemicals used in well stimulation have not been properly identified. Wastewater constituents and concentrations are not well understood. Data on the treatment technologies being used at individual well sites are not available. Although it is possible to identify potential chemical release mechanisms and the associated potential contamination pathways, insufficient data exist to confirm or refute concerns that surface and groundwater resources have been or may be contaminated by unconventional oil and gas development.

It is expected that many of data gaps will be addressed under new regulations being promulgated as part of implementation of SB 4 legislation, but there is a clear need for directed scientific studies related to the water environment. These studies are needed to answer important questions concerning the safety and sustainability of unconventional oil and gas development. How green chemistry principals might be applied to hydraulic fracturing requires scientific study. A better understanding of overall wastewater management practices in the industry are needed, including understanding the fate of injected chemicals, the chemical composition of wastewaters over varying time and spatial scales, and a complete understanding of methods and practices of water reuse and disposal. Mass-balance analyses at individual well sites are warranted to clarify the fate of stimulation chemicals remaining in the formation and the quantities of stimulation chemicals in the wastewater. The effects of legacy and current practices on local and regional groundwater quality need priority investigation, and should be complemented with geochemical modeling to characterize the fate and transport of well stimulation chemicals. Coordinated investigations need to be conducted to determine which, if any, of the identified potential pathways pose a significant risk for releasing well stimulation chemicals or other contaminants into the environment.

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Chapter Three

Air Quality Impacts from Well Stimulation

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3.1. Abstract

Well stimulation has the potential to emit greenhouse gases (GHGs), volatile organic compounds (VOCs), nitrous oxides (NO_x), toxic air contaminants (TACs), and particulate matter (PM). These pollutants can have impacts across various temporal and spatial scales ranging from long-term, global impacts (e.g., from GHGs) to local, short-term impacts (e.g., from TACs). Because oil and gas development in general can have these impacts, the purpose of this chapter is to evaluate what is known about the contribution of well stimulation to general impacts from oil and gas development. This chapter performs analysis at the statewide scale (GHGs) and at regional air district levels (criteria pollutants and air toxics). For an analysis of air impacts at small spatial scales, see Volume II, Chapter 6, which covers public health aspects of oil and gas development.

Detailed air pollution inventories are performed by the California Air Resources Board (CARB) for all major industrial sectors, including oil and gas production. Current inventory methods provide estimates of the air quality impacts related to oil and gas activities (see discussion of inventory data gaps below).

Statewide, oil and gas operations are small contributors to GHG emissions (4%), and most of these GHG emissions are associated with heavy oil production in oilfields developed without well stimulation.

In the San Joaquin Valley air district, oil and gas sources are responsible for significant contributions to sulfur oxides (SO_x) emissions (31%) and smaller contributions to reactive organic gases (ROGs) and NO_x (8% and 4%, respectively). Oil and gas activities in the San Joaquin Valley are estimated to contribute to non-negligible (>1%) fractions of some TAC species (benzene, formaldehyde, hexane, zylene) and the majority (70%) of hydrogen sulfide emissions. The fractional importance of upstream oil and gas sources to air quality concerns is higher in some sub-regions within air districts, such as western Kern County. In the South Coast air district, the oil and gas sector is a small source (<1%) of all studied pollutants.

Well stimulation is estimated to facilitate about 20% of California production, and direct well stimulation emissions represent only one source among many in the oil and gas production process. Applying these weighting factors, well stimulation emissions (direct and indirect) can be estimated at approximately one-fifth of emissions reported above.

Experimental studies of air quality in California suggest that current inventory methods underestimate methane and VOC emissions from California oil and gas sources. This suggests that the above inventory results should be considered lower-bound estimates, and the degree of inventory underestimation varies by study type and location.

Oil and gas activities occur in California air basins that already face severe air quality challenges. The two largest oil and gas-producing regions in California are in the San Joaquin and South Coast air basins, which are non-compliant with federal air quality (ozone and PM) regulations. In some cases, this non-compliance is rated as "severe" or "extreme."

While well stimulation emissions are a small portion of overall emissions sources in California, they can still be improved. A significant reduction in emissions related to well stimulation is possible using currently available technology. Some mitigation technologies are currently mandated by federal or state regulatory requirements, such as "green completions" technologies that capture gas produced during the flowback process (which would otherwise be flared or vented). Current regulatory requirements do not cover or require application of all available control technologies, and the regulatory environment is in flux federally and in California. For example, the California Air Resources Board is currently examining oil and gas sector emissions in order to develop standards to supplement recent federal regulations.

Significant data gaps exist with respect to air emissions from well stimulation. It is not clear how completely the current inventory methods cover air quality impacts from well stimulation, although it appears that at least some well stimulation air impacts will be covered by current inventory methods. Current inventory methods are not designed to separately analyze well stimulation emissions. As noted above, inventories are only infrequently verified experimentally. A small number of studies have directly measured emissions from well stimulation or in regions where well stimulation occurs. A larger body of studies exists on indirect (remote) estimates of oil and gas-related emissions in oil and gas-producing regions. There is no current consensus on where well-stimulation-related emissions specifically are largest, and significant uncertainty exists regarding emissions sources from oil and gas activities in general, although as noted above the experimental estimates of emissions have generally been found higher than inventory levels of emissions from oil and gas sources.

Preliminary quantitative assessment of the impacts due to well stimulation is made in the Volume III case studies for the San Joaquin Valley and South Coast regions.

3.2. Introduction

Well stimulation can impact air quality via emission of a large variety of chemical species. These species can have local, regional, or global impacts, mediated by the regional atmospheric transport mechanisms and the natural removal mechanisms relevant for that species. For clarity, this report groups species into four categories of interest, each with unique potential impacts.

1. Greenhouse gases (GHGs).

2. Volatile organic compounds (VOCs), and nitrogen oxides (NO_x) that cause photochemical smog generation.

3. Toxic air contaminants (TACs), a California-specific designation similar to federal designation of hazardous air pollutants (HAPs).

4. Particulate matter (PM), including dust.

GHGs have global impacts over long time scales through their effects on the radiation balance of the atmosphere. GHGs can also have significant local ecosystem effects, such as ocean acidification from rising atmospheric carbon dioxide (CO_2) concentrations. VOCs have regional impacts over the short- to medium-term through their effects on formation of photochemical smog and exacerbation of chronic health problems. In portions of this report dealing with California inventories of criteria pollutants, the term *reactive organic gases* (ROGs) will be used instead of VOC. ROGs are a defined class of species in California regulation, and have similar membership as other designations such as volatile organic compounds, nonmethane volatile organic compounds or speciated nonmethane organic compounds (ROGs, NMVOCs or SNMOCs). TACs and PM have local and regional health impacts mediated by transport and inhalation processes.

Some chemical species have impacts across multiple categories. For example, in addition to smog-formation potential, VOCs often also function over short and long time scales as GHGs through their eventual decomposition into CO_2. In these cases, species will be discussed primarily in terms of their most notable impact pathway. For example, though the degradation products of benzene can act as GHGs, benzene will be discussed as a TAC due to its larger importance in that domain. Similarly, PM has health as well as climate and aesthetic (visibility) impacts.

3.2.1. Chapter Structure

This introductory section first describes methods of classifying well-stimulation-related air impacts, and the major sources and types of emissions from oil and gas activities (remainder of Section 3.2). This is followed by an outline of current treatment of well-stimulation-related emissions in current California emissions inventories (Section 3.3).

Then, the report discusses the California regions likely to be affected by the use of well stimulation technology (Section 3.3.17) and the hazards associated with possible air impacts (Section 3.4). Next, the report outlines current best practices for managing air quality impacts of well stimulation (Section 3.5). This is followed by a discussion of gaps in data and scientific understanding surrounding well-stimulation-related air impacts (Section 3.6). Finally, a summary of findings and conclusions is presented (Sections 3.7 and 3.8).

3.2.2. Classification of Sources of Well Stimulation Air Hazards

Emissions from well stimulation can be classified as direct or indirect emissions. Direct impacts are uniquely associated with well stimulation and do not occur when oil and gas are produced without the aid of well stimulation. Examples of direct impacts of well stimulation include greenhouse gas emissions from equipment used to stimulate the well, and off-gassing of VOCs from stimulation fluids held in retention ponds and tanks. Indirect impacts stem from the other aspects of the oil and gas production process apart from well stimulation. Examples of indirect impacts include emissions from equipment used for well-pad construction, well drilling, and production of oil and gas; and off-gassing from produced water. This chapter will focus primarily on direct impacts, although important indirect impacts will also be discussed. This is because indirect impacts play an important role in air quality impacts in regions of significant well stimulation activities, and may be important determinants of long-run air quality impacts of well stimulation.

3.2.3. Greenhouse Gas Emissions Related to Well Stimulation

GHG and climate-forcing emissions to the atmosphere associated with well stimulation include the following: carbon dioxide (CO_2), methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O), VOCs, and black carbon (BC) (IPCC, 2013, pp. 738-740). For the purposes of GHG accounting, IPCC practice recommends binning all VOC species by mass of carbon (IPCC, 2013, pp. 738-740). Well stimulation practice can also result in the emission of species with negative climate forcing (i.e., cooling impacts) such as NO_x and organic carbon (OC) (IPCC, 2013). Nevertheless, the net effect of emissions from well stimulation is expected to be primarily warming. The climate impacts, listed using current 20-year and 100-year global warming potentials (GWPs) for well-stimulation-relevant gases, are listed in Table 3.2-1.

Table 3.2-1. Global warming potential of well-stimulation-relevant air emissions. (IPCC, 2013)

Gas species	GWP 20-yr	GWP 100-yr	Notes
Carbon dioxide (CO_2)	1	1	a
Methane, fossil (CH_4)	85	30	a
Nitrous oxide (N_2O)	264	265	a
Carbon monoxide (CO)	5.6 (+/-1.8)	1.8 (+/- 0.6)	b
Volatile organic compound (VOC)	16.2 (+/- 9.2)	5.0 (+/- 3.0)	c
Black carbon (BC)	1200 (+/- 720)	345 (+/- 207)	d
Organic carbon (OC)	-160 (+/- 68)	-46 (+/- 20)	d
Nitrogen oxides (NO_x)	-2.4 (+/ 30.3)	-8.2 (+/- 10.3)	e

a – From (IPCC, 2013) Table 8.A.1
b – From (IPCC, 2013) Table 8.A.4, for CO emissions in North America. CO GWP varies by the region of emissions due to regional differences in atmospheric processes.
c – From (IPCC, 2013) Table 8.A.5. Measured on per-kg of carbon basis. Estimate for North America VOC GWP varies by the region of emissions due to regional differences in atmospheric processes.
d – From (IPCC, 2013) Table 8.A.6. BC and OC GWPs taken from "four regions" study result, which encompasses East Asia, European Union (EU) + North Africa, North America, and South Asia.
e – From (IPCC, 2013) Table 8.A.3. Values for NO_x from North America.

3.2.4. Volatile Organic Compounds and Nitrous Oxides Emissions Related to Well Stimulation

VOCs are a large class of organic compounds that are variously defined. Thousands of chemical species are included in VOC definitions, with many of them present in hydrocarbon gases and liquids. VOCs include benign compounds as well compounds that are directly hazardous to humans. Hazardous VOCs will be discussed in the TACs section below. In certain conditions, VOCs react in the atmosphere to increase ozone formation. Some VOCs are transformed by atmospheric processes to particulate matter (PM).

Definitions of VOCs vary between regulatory regimes. U.S. Environmental Protection Agency (U.S. EPA) definitions list VOCs as organic species with vapor pressure greater than 10^{-1} Torr at 25°C and 760 mmHg (U.S. EPA, 1999). This regulatory definition exempts non-photochemically active species such as CH_4 and ethane (C_2H_6). This definition is designed to include organic species that are likely to exist in gaseous phase at ambient conditions. VOC emissions associated with well stimulation are numerous, with oil-and-gas-focused air studies measuring concentrations of many dozens of species (U.S. EPA, 1999; ERG/SAGE, 2011).

NO_x emissions associated with well stimulation activities derive primarily from use of engines powered by diesel or natural gas, which are used directly in well stimulation applications. Examples include drilling and workover rigs, fracturing trucks with large pumps for generating high fluid injection pressure, and other trucks of various kinds (e.g., proppant delivery trucks). Flaring can be another source of NO_x from oil and gas operations.

3.2.5. Toxic Air Contaminant Emissions Related to Well Stimulation

There are numerous TACs associated with well stimulation, which most commonly fall into the category of toxic organic compounds (TOCs) (U.S. EPA, 1999). These well-stimulation-associated TACs include many of the species defined as VOCs in the Clean Air Act (CAA) Amendments of 1990. TACs can be an acute or chronic concern for workers in the oil and gas industry, due to possibly frequent exposure to elevated concentrations of TACs, as well as long-term work in environments with TACs. TACs may also present a health concern for more remote persons that are less heavily exposed, such as those who live near oil and gas operations.

3.2.6. Particulate Matter Emissions Related to Well Stimulation

PM emissions in oil and gas development occur most commonly due to stationary combustion sources (CARB, 2013b). Other PM sources include heavy equipment in on-road and off-road operations, and land disturbance. Common sources of PM include diesel-powered equipment such as trucks, drilling rigs, generators, and other off-road equipment (e.g., preparatory land-moving equipment) (CARB, 2013b). PM may also be emitted through combustion (flaring) of wet gas (i.e., gas containing high molecular weight hydrocarbons). PM emissions are associated with respiratory health impacts and increased rates of mortality (see Chapter 6 on health impacts).

3.3. Potentially Impacted Resource—Air

3.3.1. California Air Quality Concerns

California has faced air quality concerns for many decades. Historical attention has focused primarily on smog-forming pollutants (e.g., VOCs and NO_x) and toxic air contaminants (TACs). A number of factors result in California air quality being among the most impacted in the nation. First, a large population of 40 million residents results in significant air emissions. Second, some California regions have unfavorable topography for air quality management, including large urban areas surrounded by mountains that prevent mixing and transport of emitted species. Third, the generally warm and sunny conditions in the state promote photochemical reactions and formation of smog. In some regions (noted below), agricultural activities can result in fine particulate pollution of concern.

More recently, regulatory efforts at the California Air Resources Board (CARB) have focused on GHG emissions. This has resulted in the development of broad industry-spanning GHG cap and trade regulations (CARB, 2014a), as well as oil and gas-specific regulatory efforts and ancillary transport-fuel regulations that affect oil and gas operators (CARB, 2014b).

CARB defines 35 Air Pollution Control Districts (APCDs) and Air Quality Management Districts (AQMDs), which are collectively called "air districts" (CARB, 2014c). These air districts are shown in Figure 3.3-1.

The two largest California oil and gas-producing regions are contained within the San Joaquin Valley Unified air district (henceforth SJV) and South Coast air district (henceforth SC). Significant oil production also occurs in the Santa Barbara and Ventura air districts. Non-associated (dry) natural gas production occurs in a number of Northern SJV air districts.

Large quantities of GHGs, VOCs, TACs, and PM are emitted by non-oil and gas sources in California, including primary industry, homes and businesses, and the transport sector.

Figure 3.3-1. California Air Pollution Control Districts (APCDs) and Air Quality Management Districts (AQMDs), collectively called "air districts." Image reproduced from CARB (2014c).

3.3.2. Estimating Current Impacts of Oil and Gas Operations on California Air Quality

Estimates of emissions for species of interest in California are tabulated, estimated, or inventoried for a variety of sources in the oil and gas sector. These estimates include:

1. Field-level estimates of GHG emissions produced for transport GHG intensity regulations (i.e., Low Carbon Fuel Standard).

2. State-level inventories of GHGs, ROGs, TACs, and PM compiled by the California Air Resources Board (CARB)

3. State-level surveys of emissions from oil and gas operators

4. Federal databases of GHG emissions and toxics releases (U.S. Environmental Protection Agency)

5. Detailed (spatially and temporally) inventories of air emissions for photochemical grid-based modeling of ozone formation.

This chapter covers the first four of these sources of information, with a strong focus on California-specific methods (first three sources in above list). These methods are described in order below, starting with field-level GHG intensity estimates. Each section describes the estimation methods and estimates derived for each species of interest, in the order of GHGs, VOCs, TACs, and PM. Table 3.3-1 shows a summary of where data were obtained for each type of assessment.

Table 3.3-1. Coverage of different assessment methods and key sources for each method.

Estimate type	**Resulting data**	**Data source**	**Source**
Field-level GHG estimates	t CO_2eq[1]. GHGs per year	DOGGR[2] production data	(DOGGR, 2014)
		CARB and OPGEE[3] model results of GHG intensities	(Duffy, 2013) (El-Houjeiri et al., 2013, 2014)
State-level emissions inventory	t CO_2eq. GHGs per year	CARB yearly GHG inventory	(CARB, 2014d,e) (CARB, 2013a)
	t ROG per year	CARB criteria pollutants inventory, incl. stationary and mobile sources	(CARB, 2013b)
	kg TACs per year	CARB overall toxics inventory (California Toxics Inventory)	(CARB, 2013c)
	kg TACs per year	CARB facility-level toxics reporting	(CARB, 2014j)
	t PM per year	CARB criteria pollutants inventory, incl. stationary and mobile sources	(CARB, 2013b)
Surveys of oil and gas operators	t CO_2eq. GHGs per year	CARB special survey of oil and gas operators	(Detweiler, 2013)
Federal GHG and toxics databases	Various	Not studied extensively in this report	(U.S. EPA, 2012)

[1]*CO_2-equivalent*

[2]*Department of Oil, Gas, and Geothermal Resources*

[3]*Oil Production Greenhouse Gas Emissions Estimator*

The scale at which emissions are assessed, and how emissions and their impacts are quantified, can influence study results. With regard to spatial scale, this chapter covers emissions at the statewide scale (in the case of GHG emissions) and at regional air district scales (in the case of criteria pollutants and air toxics). GHG emissions are assessed for the state as a whole, because GHGs are a global problem largely independent of location of emissions. In contrast, regional air districts are assessed for other pollutants, because these regions are designated by CARB as regions where atmospheric mixing and transport require the pollutants in a given region to be co-regulated. With regard to how emissions are quantified in this chapter, we examine mass-emissions rates and the fractional responsibility of oil and gas industry sources to the air quality problems studied.

Other spatial scales can matter for some pollutants. For example, emissions responsibility for oil and gas operations over smaller spatial scales can be higher than for an air district-wide measure. For example, when emissions are assessed for Kern County alone, there is larger responsibility of oil and gas sources than those found in this chapter for the San Joaquin Valley air district. At an even finer spatial scale, the specific location of an air toxics source can be very important. These smaller-scale assessments can be found in the following locations:

- County-scale assessment of impacts: Volume III, San Joaquin Basin Case Study; Volume III, Los Angeles Basin Case Study.

- Local-scale assessment of emissions near sensitive populations: Volume II, Chapter 6 and Volume III, Los Angeles Basin Case Study and San Joaquin Basin Case Study.

Also, there are other ways to measure the importance of emissions than mass-emissions rates and the fraction of responsibility for a given industry. For example, in public health studies generally, the concentration of pollutant and the mass of pollutant being inhaled by the studied population is of concern, not necessarily the overall mass emissions rate in an air basin. Some health-damaging pollutants may therefore be of great concern at a local scale, even with small mass-emissions rates (e.g., oil and gas associated TACs such as benzene or toluene). See Volume II, Chapter 6, and Volume III, Los Angeles Basin Case Study for more information.

3.3.2.1. California Air Resources Board Field-Level Estimates of Greenhouse Gas Emissions from Oil Production

CARB produces an estimate of the greenhouse gas intensity of different producing oilfields in California, as part of the Low Carbon Fuel Standard (LCFS) effort (Duffy, 2013). The LCFS seeks to incentivize the production and consumption of transportation fuels with lower life cycle greenhouse gas intensity compared to conventional oil resources. Because the structure of the regulation assesses alternative fuels in comparison to oil-derived fuels, an accurate baseline emissions intensity for oil consumed in California is required.

As part of this effort, 154 California oil fields are assessed using the Oil Production Greenhouse Gas Emissions Estimator (OPGEE), an open-source tool produced by researchers at Stanford University (El-Houjeiri et al., 2014, El-Houjeiri et al., 2013). OPGEE takes the properties of an oilfield and uses them to estimate the greenhouse gas emissions associated with producing, processing, and transporting the crude oil to the refinery inlet gate. While OPGEE cannot be used to assess emissions individually from pools that are facilitated or enabled with well stimulation technologies, it can be used to assess the emissions from oilfields within which well-stimulation-enabled pools exist.

Using information from Volume I, Appendix N, a total of 45 pools across California were determined to be facilitated by or enabled by well stimulation technologies. These pools are located in 28 California oilfields. While the pools themselves were found to account for ~20% of California oil production, the fields within which these pools exist were responsible for nearly 40% of California's oil production in 2012. The fields in which these pools exist, in general, contain lighter crude oil and result in lower greenhouse gas intensity than the average California oilfield (see Figure 3.3-2). The production-weighted-average GHG intensity for well-stimulation-enabled pools is approximately 74% that of non-stimulated pools and 64% of California fields in general.

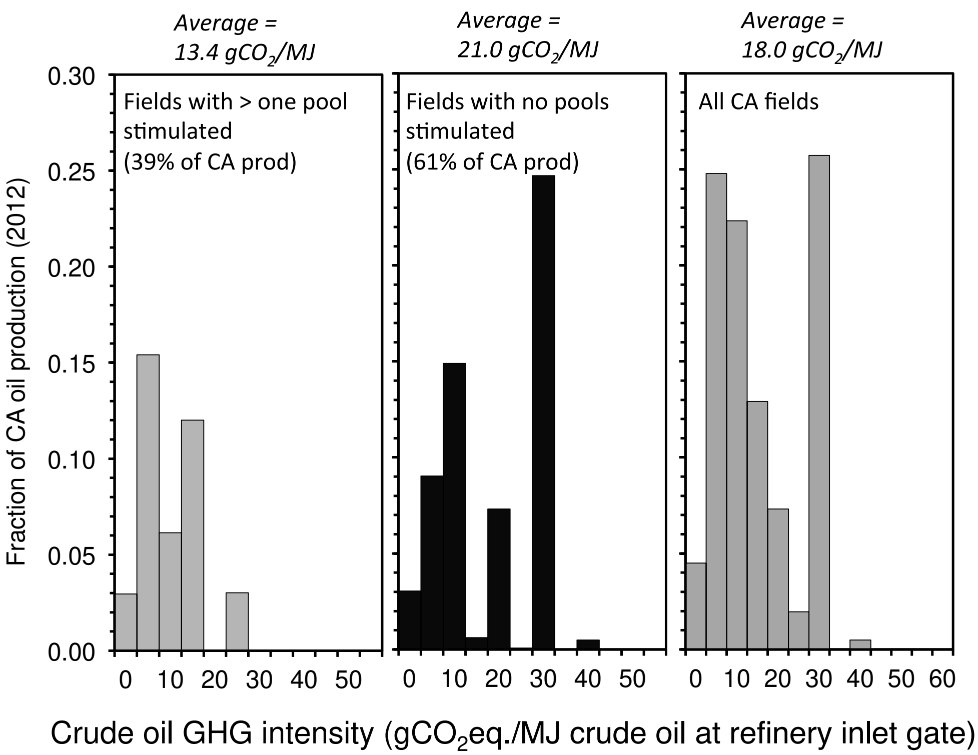

Figure 3.3-2. Distribution of crude oil greenhouse gas intensity for fields containing well-stimulation-enabled pools (left), those that are not stimulated (middle) and all California oilfields (right).

An important question regarding GHG emissions from stimulated wells is: "What would happen to GHG impacts if well stimulation were not practiced in the state?" If well stimulation were disallowed and consumption of oil and gas in California did not drop in response, the required oil would come from some other oilfields. That is, more oil and gas would be required from non-stimulated California fields or regions outside of California. This substitution would be the result of oil market shifts that would occur in response to the shift in California production.

Depending on the source of substituted oil and gas, overall greenhouse gas emissions due to oil production could increase if well stimulation were stopped. Computing the net GHG change associated with well stimulation therefore requires understanding of both in-state and out-of-state production, as well as the likely sources of "new oil." Thus, estimating the scale of impact requires a market-informed life cycle analysis (LCA). (This type of analysis is sometimes called "consequential" LCA.)

3.3.2.2. State-Level Emissions Inventories Produced by California Air Resources Board (CARB)

The California Air Resources Board (CARB) produces annual inventories of emissions of GHGs, VOCs, TACs and PM. These inventory methods and results are described in order below. In all cases, numerical results for 2012 will be presented, due to incomplete reporting for the year 2013 at the time of analysis.

The methods used to generate emissions inventories vary by the gas of interest. In general, emissions inventories collect data at the district level, and aggregate results to generate broader statewide estimates (CARB, 2014d). Direct measurements do not generally underlie emissions estimates included in inventories. For example, stationary source emissions are generally estimated using established emissions factors that are applied to the number of facilities of a given type in an analyzed region for a particular year. Similarly, rather than directly measuring vehicle emissions, databases of vehicle activities are used along with mobile source emissions factors (CARB 2014d). A full description of inventory methods is beyond the scope of this report, but where possible, methods and their impacts on emissions estimates are discussed.

3.3.2.2.1. CARB GHG Inventory for Oil and Gas Operations

CARB GHG inventories are produced on a yearly basis for the “six Kyoto gases”: carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), sulfur hexafluoride (SF_6), hydrofluorocarbons (HFCs), and perfluorocarbons (PFCs) as well as nitrogen triflouride (NF_3) (CARB, 2014d). CARB GHG inventories report mass emissions of each gas, as well as CO_2-equivalent (CO_2eq.) emissions using IPCC Assessment Report (AR4) GWP factors.

Results from CARB GHG inventories can be queried by economic sector, as well as subsectors of various levels (CARB, 2014e). Direct well stimulation (WS) GHG emissions would be included in the subsector “Industrial > Oil & gas extraction.” Additional indirect well-stimulation-related emissions, such as those resulting from induced hydrocarbon production, may occur more broadly (e.g., oil refining, refined product transport).

CARB GHG inventory methods

For each CARB-defined subsector, an “Activity” is defined. Activities with relevance for WS and for oil and gas activities include “Fuel Combustion” and “Fugitive Emissions.” Within the “Fuel Combustion” activity, activity subsets exist to record the type of fuel consumed (e.g., natural gas, associated gas, distillate fuel). Each activity subset can result in emission of numerous GHGs. Combustion processes typically result in CO_2, CH_4, and N_2O, while fugitive emissions are a large concern due to their CH_4 content. The classification scheme under which direct well stimulation emissions would be classified in CARB GHG inventories is shown in Table 3.3-2. Many indirect emissions induced by WS activities would also be inventoried in these categories.

Table 3.3-2. CARB GHG inventory emissions of interest for WS (CARB, 2014e).

Main sector	Sub-sector Level 1	Sub-sector Level 2	Sub-sector Level 3	Main activity	Activity subset	GHG emitted
Industrial	Oil & gas extraction	Not specified	None	Fuel combustion	Associated gas	CH_4, CO_2, N_2O
		Not specified	None	Fuel combustion	Distillate	CH_4, CO_2, N_2O
		Not specified	None	Fuel combustion	Natural gas	CH_4, CO_2, N_2O
		Not specified	None	Fuel combustion	Residual fuel oil	CH_4, CO_2, N_2O
		Petroleum gas seeps	Fugitives	Fugitive emissions	NA	CH_4
		Process losses	Fugitives	Fugitive emissions	NA	CH_4, CO_2, N_2O
		Storage tanks	Fugitives	Fugitive emissions	NA	CH_4
		Wastewater treatment	Fugitives	Fugitive emissions	NA	CH_4

The CARB oil and gas GHG emissions inventory methodology is based on two key data sources and methodologies. First, CARB uses IPCC Guidelines with state and federal data sources (IPCC, 2006). More recently, CARB has augmented IPCC-based methods with more detailed reporting under the Mandatory Reporting Regulation (MRR), a state-level regulation requiring detailed reporting of GHG emissions by large emitters.

The IPCC methodology primarily tracks energy use. Briefly, energy use is gathered for a given sector, and this use is multiplied by a fuel-specific emissions factor for each fuel type (CARB, 2014f, pp. 56-58). To complete its oil and gas GHG inventory, CARB obtains fuel use data for oil and gas activities from the following state and U.S. federal sources: U.S. Energy Information Administration (EIA), California Energy Commission, and the California Department of Oil, Gas, and Geothermal Resources (DOGGR) (CARB, 2014f, p. 58). Fugitive emissions are estimated in this methodology using information generated from the California Emission Inventory Development and Reporting System (CEIDARS) database (CARB, 2014f, p. 59), which is developed for tracking criteria pollutants such as VOCs. See significant additional discussion of CEIDARS below.

More recently, the California GHG inventory leverages California MRR datasets relevant to well stimulation and oil and gas activities. MRR data are gathered from the category of processes entitled “Petroleum and Natural Gas Systems” (CARB, 2014g, sect. 95101). MRR data reporting is required from all oil and gas operators whose stationary and process emissions of CO_2, CH_4 and N_2O exceed 10,000 tonnes (t) of CO_2eq. per year, or whose stationary combustion, process, fugitive, and vented emissions of the above gases equal or exceed 25,000 tCO_2eq. per year (CARB, 2014g, sect. 95101). Detailed methods are given for estimation of emissions from various oilfield operations (CARB, 2014g, sect. 95150), with different oil and gas subsegments required to report information using a separate set of individual methodologies (CARB, 2014h). These methods rely on emissions-factor-like approaches for some categories, as well as engineering-based equations for other categories.

Coverage of well stimulation activities in GHG inventory

The CARB GHG inventory covers oil and gas emissions using a variety of mechanisms. With regard to combustion emissions analyzed under IPCC methods, the most important quantities are fuel consumption during well stimulation activities (e.g., diesel fuel to operate hydraulic fracturing operations). Distillate fuel consumption in California for the CARB GHG inventory is taken from U.S. EIA dataset "Adjusted Sales of Distillate Fuel Oil by End Use" (CARB, 2014f, p. 58). This dataset reports distillate fuel consumption partitioned by sector at the state level (U.S. EIA, 2014a). The end use sector of interest is the U.S. EIA-defined "Oil Company" sector, which is defined as per U.S. EIA definitions:

> *"An energy-consuming sector that consists of drilling companies, pipelines or other related oil companies not engaged in the selling of petroleum products. Includes fuel oil that was purchased or produced and used by company facilities for operation of drilling equipment, other field or refinery operations, and space heating at petroleum refineries, pipeline companies, and oil-drilling companies. Sales to other oil companies for field use are included, but sales for use as refinery charging stocks are excluded." (U.S. EIA, 2014b)*

This U.S. EIA definition is sufficiently general such that it should include diesel fuel use for WS activities. Because of the aggregated nature of the U.S. EIA diesel fuel consumption dataset, strictly maintained to provide operator confidentially, no greater specificity can be provided about how accurately this portion of the CARB GHG inventory accounts for combustion GHG emissions directly related to WS. If some California WS-related operators did not report fuel use to the U.S. EIA under these requirements, their use would not be counted.

Non-combustion emissions estimates that are not modeled using mandatory reporting regulation (MRR) methods are derived from the CEIDARS database of criteria air pollutants (CARB, 2014f, p. 59). Fugitive emissions of CH_4, CO_2 and other gases (VOCs) that arise during oil and gas operations are estimated using CEIDARS data. The CEIDARS total organic gases (TOG) emissions inventory (CARB, 2014f) is used for this purpose. This inventory is discussed further below, because this TOG inventory includes VOCs as well as methane emissions. A speciation model (CARB, 2000) is used to estimate emissions of GHGs from TOG emissions sources (CARB, 2014f, p. 59). As discussed below, the coverage of well-stimulation-related activities in the criteria pollutants inventories is uncertain.

Starting a few years ago, the above methods are being supplemented by data reported directly by operators through CARB's MRR program (CARB, 2014f, p. 59). MRR reporting requires reporting of fuel consumed by operators above a size threshold. MRR sources are also required to estimate "fugitive emissions from pipes, storage tanks, and process losses in the oil & gas extraction…sectors" (CARB, 2014f, p. 59), using a series of methods that have been harmonized with federal emissions reporting requirements. It is unknown what fraction of wells drilled in California are drilled by companies reporting to MRR databases, although most data from inventories are still derived from non-MRR sources.

Most relevant to well stimulation activities, flowback emissions from natural-gas well completion, post-well-stimulation activities are to be computed and reported in methods equivalent to U.S. EPA federal reporting requirements using the U.S. EPA GHGRP (GHG reporting program) (GHGRP Subpart W, see below). These methods are a mix of empirical and engineering-based methods for estimating emissions given technology characteristics and operating conditions (e.g., operating pressure).

Given the above level of detail required as part of MRR reporting, it is likely that many well-stimulation-related emissions sources will be included in MRR data. Some well-stimulation-related emissions may not be covered if subcontractor emissions occurring during well stimulation do not meet reporting thresholds related to operator size. It is not possible to discern the exact coverage (or lack thereof) of well stimulation activities within the MRR dataset, due to the aggregated nature of public data reporting.

Results of CARB GHG inventory

Statewide GHG emissions in California totaled 466 Mt CO_2eq. in 2012. The "Industrial > Oil & gas extraction" sector was responsible for ~17 MtCO_2eq., or somewhat less than 4% of statewide emissions (CARB, 2014d).

The dominant contributor to the oil and gas GHG inventory was CO_2 emissions resulting from fuel use. Fuel use in oil and gas development in California is heavily influenced by combustion of fuels for thermal enhanced oil recovery. Fugitive emissions from oil and gas totaled <1.5 Mt CO_2eq., or 0.3% of statewide emissions (CARB, 2014d). As shown in Figure 3.3-3, the overall trend in California oil and gas GHG emissions is downward over time, likely due to decreasing California oil and gas production.

If more recent IPCC AR5 GWPs (see Table 3.2-1) are used instead of CARB-applied IPCC AR4 GWPs, CO_2-equivalent GHG emissions from the California oil and gas industry increase by only a small amount between 2000 to 2012—specifically, the yearly increase ranges from 0.5% to 1.2%. Note that emissions sources classified as "combustion" sources can result in CH_4 emissions due to incomplete combustion or direct loss from combustion equipment.

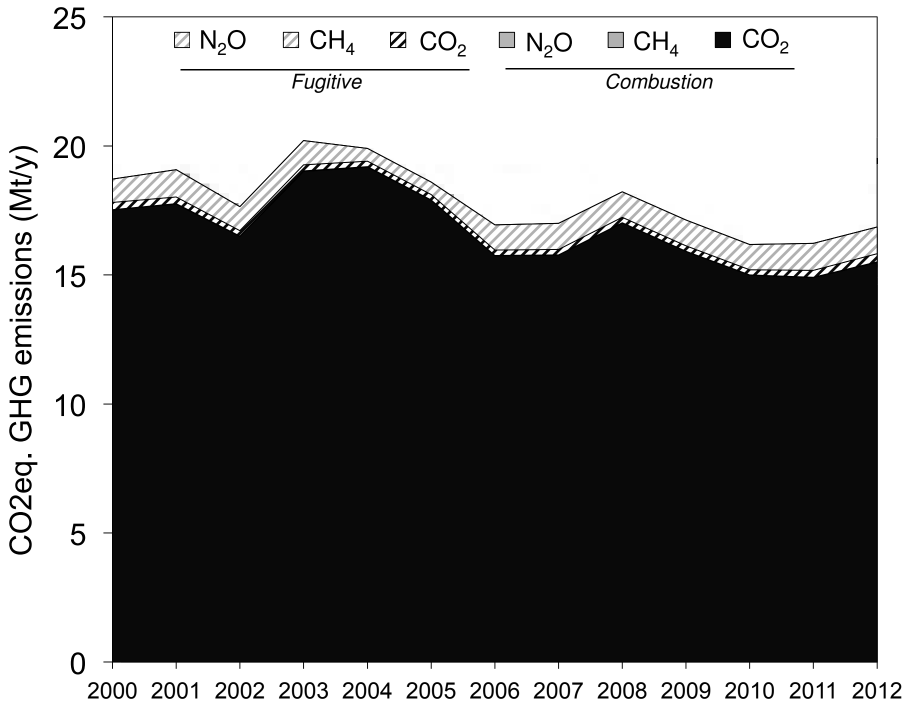

Figure 3.3-3. Emissions from "Oil and Gas Extraction" sources as reported in California GHG inventory. Source: plotted from (CARB, 2014d). Emissions in million metric tonnes per year (109 kg per year). Oil and gas extraction activities account for <4% of total statewide GHG emissions."

Summary of CARB GHG inventory coverage

The CARB GHG inventory is likely to include emissions from many well stimulation activities. To summarize the discussion above:

- Baseline data for the GHG inventory data appear to derive from a combustion emissions inventory that uses (among other sources) federally reported fuel consumption data for a broadly defined oil and gas sector.

Oil and gas emissions are also subject to (for large producers) MRR requirements, which specify detailed reporting methodologies and broad coverage of combustion and noncombustion sources. Even given this broad reporting requirement and comprehensive coverage, it is not clear that GHG emissions from all well stimulation activities are reported as part of the CARB GHG inventory. For example:

- Smaller producers are exempt from MRR requirements. Given that the criteria pollutants inventory does not definitively include well stimulation activities, these operators could be a source of missing well-stimulation-related GHG emissions.

- MRR data reporting has some known coverage gaps. For example, MRR data reporting includes flowback emissions during completion of well stimulation applied to natural gas wells. However, MRR does not appear to require reporting of flowback emissions from well stimulation applied to oil wells.

3.3.2.2.2. CARB Inventories for VOC and NOx (Smog-Forming) Emissions

CARB inventories of criteria air pollutant emissions are performed on a yearly basis for each air district. Detailed estimates of emissions by sectors, sources, and subsources are presented for a variety of species, including total organic gases (TOG), reactive organic gases (ROG), NO_x, SO_x, CO, and PM. CARB documentation suggests that CARB ROG emissions are very similar to (though not exactly equal to) U.S. EPA-defined VOC emissions (CARB, 2000). TOG emissions include ROGs/VOCs, as well as non-photochemically active organic gases such as CH_4 and C_2H_6 (CARB, 2000). For the remainder of this section, we will use the CARB terminology of ROG.

CARB criteria pollutant inventory methods

The CARB criteria air pollutant inventory is divided broadly into three categories: stationary sources, area-wide sources, and mobile sources. These categories are then broken down into sectors, subsectors, and sources. Inventory methods vary for each broad source category, as well as within each source category. The most relevant categories for smog-forming emissions from well stimulation and oil and gas operations are given in Table 3.3-3. It does not appear that area-wide sources are relevant for well stimulation or oil and gas operations. Detailed lists of contributing equipment or technologies for each subsector are presented below.

Table 3.3-3. CARB criteria pollutant inventory sector/subsector pairings of interest for oil and gas and well stimulation emissions.

Broad category	Sector	Subsector
Stationary sources	Fuel combustion	Oil and gas production (Combustion)
	Petroleum production and marketing	Oil and gas production
Mobile sources	On-road motor vehicles	Various
	Other mobile sources	Off-road equipment

Compared to the GHG inventory described above, the criteria pollutants inventory reports emissions sources in considerable detail. For example, in the stationary source criteria pollutants inventory, emissions are tracked for multiple types of combustion technologies (i.e., reciprocating engines, boilers, turbines, steam generators) rather than a broad “combustion emissions” category. Also, more fuels are represented, with fuel subspecification available for types of distillate fuel or types of gaseous fuel. Lastly, different emissions mechanisms within a given equipment category are represented. For example, ROG emissions from tanks are classified into breathing and working losses for both fixed and floating roof tanks.

To determine the coverage of stationary-source oil and gas emissions, all sources classified in the “Stationary sources > Fuel combustion > Oil and gas production (Combustion)” and “Stationary sources > Petroleum production and marketing > Oil and gas production” subsectors are summed for the SJV and SC regions. The resulting sources and materials (e.g., fuel, working fluid, or chemical) responsible for emissions in these subsectors are listed in Table 3.3-4. While other possible sources might exist in other air basins, these two air basins are indicative of California oil and gas operations and are responsible for the majority of state oil production. The list of sources in Table 3.3-4 is therefore likely to be representative of statewide oil and gas sources (CARB, 2013b).

Table 3.3-4. CARB ROG/CO stationary source inventory emissions sources and material drivers of emissions within the broad categories “Oil and Gas Production” and “Oil and Gas Production (Combustion). Sources and materials taken from SJV and SC air district data.

Sources	**Materials**
Reciprocating engines	Diesel/Distillate oil (unspecified) Gasoline (unspecified) Natural gas Gaseous fuel (unspecified) Propane
Turbine engines	Natural gas Diesel/Distillate oil (unspecified)
Boilers	Natural gas Propane Process gas Residual oil #6 (Bunker C)
Process heaters	Natural gas Residual oil (unspecified)
Steam generators	Natural gas Process gas
Fugitives – Oil/water separator	Crude oil (unspecified)
Fugitives – Wet gas stripping/field separator	Gaseous fuel (unspecified)
Fugitives – Pumps	Crude oil (unspecified)
Fugitives – Compressors	Crude oil (unspecified)
Fugitives – Well heads	Crude oil (unspecified)
Fugitives – Well cellars	Crude oil (unspecified)
Fugitives – Valves	Natural gas Crude oil (unspecified)
Fugitives – Fittings	Crude oil (unspecified)
Fugitives – Sumps and pits	Crude oil (unspecified)
Fugitives – Miscellaneous	Crude oil (unspecified)
Floating roof tanks – Working	Organic chemicals (unspecified)

Fixed roof tanks - Working	Diesel #2 Crude oil – RVP 5 Ethylene Glycol Aromatics (unspecified) Acid (unspecified) Jet Naphtha (JP-4) Glycols (unspecified)
Floating roof tanks - Breathing	Organic chemicals (unspecified)
Fixed roof tanks - Breathing	Diesel #2 Crude oil – RVP 5 Organic chemicals (unspecified) Benzene Methanol Jet Naphtha (JP-4)
Tank cars and trucks - Working	Diesel/Distillate oil (unspecified) Gasoline (unspecified) Crude oil (unspecified)
Natural gas prod.	Natural gas
Steam drive wells	Crude oil (unspecified)
Cyclic steam wells	Crude oil (unspecified)
Oil production - Heavy oil test	Crude oil (unspecified)
Vapor recovery/flares	Process gas Liquefied Petroleum Gas (LPG)
Other	Material not specified Crude oil (unspecified) Mineral and metal products (unspecified) Natural gas

Mobile sources of criteria pollutants are estimated by air district for a variety of on-road and "other" mobile sources.

On-road vehicles are classified by duty class (CARB, 2013b), e.g., light duty trucks, medium-duty trucks, or heavy duty trucks. It is probable that transport of light equipment and personnel for well stimulation activities would take place using light duty trucks, while proppant, steel well casing, bulk materials, or chemicals would be hauled in heavy duty trucks. On-road truck emissions are subspecified at various levels of detail. For example, the "Heavy Heavy Duty Diesel Trucks" category has a variety of subcategories, including agriculture, construction, and port use. In contrast, "Light Heavy Duty Diesel Trucks" are not subspecified in results by the industry which employs them.

No on-road categories reported petroleum-related subcategories, so use of on-road trucks for oilfield activities such as well stimulation are not able to be determined from inventory results. At least some of the reported on-road criteria pollutant emissions are likely due to well stimulation or oil and gas activities, but inventory results are not specific enough to differentiate these uses. With access to the underlying models, examining truck use by industry sector may be possible.

The category of "Other mobile sources," however, does present oil and gas-relevant categorization under the heading of off-road equipment. The relevant category is "Other mobile sources > Off-road equipment > Oil drilling and workover." The types of equipment in this category are listed in Table 3.3-5. A variety of oilfield equipment (e.g., pumps, lifts, rigs) are modeled for a variety of equipment sizes. Mobile source emissions are modeled using a methodology that tracks populations of vehicles, vehicle usage and load factors, and vehicle distribution within the state air districts, etc. (CARB, 2010).

The vehicle database DOORS (Diesel Off-road On-line Reporting System) tracks the numbers of off-road vehicles in the state, as well as their rated horsepower for categorization (CARB, 2010). In the model base year (2010), documentation states that oilfield equipment in DOORS included 184 drilling rigs and 638 workover rigs (adjusted from reported values based on CARB estimated non-compliance rates). The load factor (fraction of maximum engine output) assumed is 50% for oilfield equipment (CARB, 2010, p. D-10). Oilfield rigs are assumed to operate for ~1,000 hours per year (CARB, 2010, p. D-14). Oil drilling equipment is allocated to the following air basins: SJV, 61.1%; Sacramento Valley, 14.5%; SC, 13%; and South Central Coast, 8.5% (CARB, 2010, p. D-33). Consulting the underlying DOORS database confirms that only these drilling rigs and workover rigs are included in the newest (2011) version of the DOORS database.

Table 3.3-5. CARB mobile source inventory emissions sources within the category "Off road, oil drilling and workover, diesel (unspecified)" taken from SJV and SC air district data.

Sources	Sizes
Compressor (workover)	D25, D120, D175, D250, D500, D750, D1000
Drill rig	D120, D175, D250, D500, D750, D1000
Drill rig (mobile)	D50, D120, D175, D250, D500, D750, D1000
Workover rig (mobile)	D50, D120, D175, D250, D500, D750, D1000
Generator (drilling)	D50, D120, D175, D250, D500, D750
Generator (workover)	D120, D175, D250, D500, D750, D9999
Lift (drilling)	D120, D175, D250, D500, D750
Other workover equipment	D120, D175, D250, D500, D750, D1000
Pressure washers	D250
Pump (drilling)	D120, D175, D250, D500, D750, D9999
Pump (workover)	D120, D175, D250, D500, D750, D9999
Snubbing	D120
Swivel	D120, D175, D250, D500

Coverage of WS activities in ROG inventory

From the stationary-source specification provided in Table 3.3-4, it appears likely that at least some well-stimulation-related activities are represented in the stationary source criteria pollutant inventory. For example, flowback emissions might be included in the category "Fugitives—Well heads" or "Fugitives—Oil/Water Separator." The fundamental data underlying these categories are summed from facility-level data. CARB methodologies do not describe exactly what is or is not included in each source category, nor how emissions estimates might have been updated in light of development of new technologies such as well stimulation. Users are recommended to contact a particular air district for more information on how a particular source was estimated.

Regarding on-road mobile sources, no information is available about how well-stimulation-related on-road emissions might be counted in the inventory.

Regarding off-road and oilfield equipment, the information presented in Table 3.3-5 suggests that at least partial coverage of well-stimulation-related equipment is provided in the mobile source inventory (e.g., rigs, pumps, generators). The exact coverage of well stimulation equipment in these databases cannot be determined.

CARB ROG inventory results are presented in mass of ROG per year. Calculations of impacts based on species-specific reactivities have been used in California regulation for assessing the actual ozone-formation potential for different species (CARB, 2011). For this report, we use the reported mass emissions of ROGs.

Results of CARB criteria-pollutant inventory: ROG and NO_x

Criteria pollutant inventory results show that oil and gas operations are generally responsible for a minority of stationary ROG and NO_x emissions. In 2012 in the SJV Unified air district, upstream oil and gas emissions totaled 25.1 t ROG/d, representing ~7.7% of ROG emissions from anthropogenic sources and 2.3% of ROG emissions from all sources (natural and anthropogenic). In the SC air district, the equivalent values were 0.23% and 0.16%, respectively (CARB, 2013b). A breakdown of ROG emissions from oil and gas operations in these two air districts is shown in Figure 3.3-4 (SC) and Figure 3.3-5 (SJV) at two levels of specificity. The left-hand side of each figure groups sources listed in Table 3.3-4 into broader categories. Major stationary sources in the SJV air district are mixed evenly between fugitive emissions and production wells. Major stationary sources in the SC air district include fugitive sources, tanks, and engines.

Note that these stationary emissions only include upstream oil and gas production and surface processing emissions; they do not include petroleum refining emissions nor consumption of the refined fuels that are produced from oil (e.g., fugitive VOC emissions from automobile fueling).

Speciation of stationary source fugitive TOG emissions is determined based on emissions source using established standard speciation profiles (CARB, 2014i, see Figure 3.3-6). These speciation profiles have lower CH_4 concentrations than other observations (see below), perhaps due to the dominance of oil and heavy oil production over dry gas production in California.

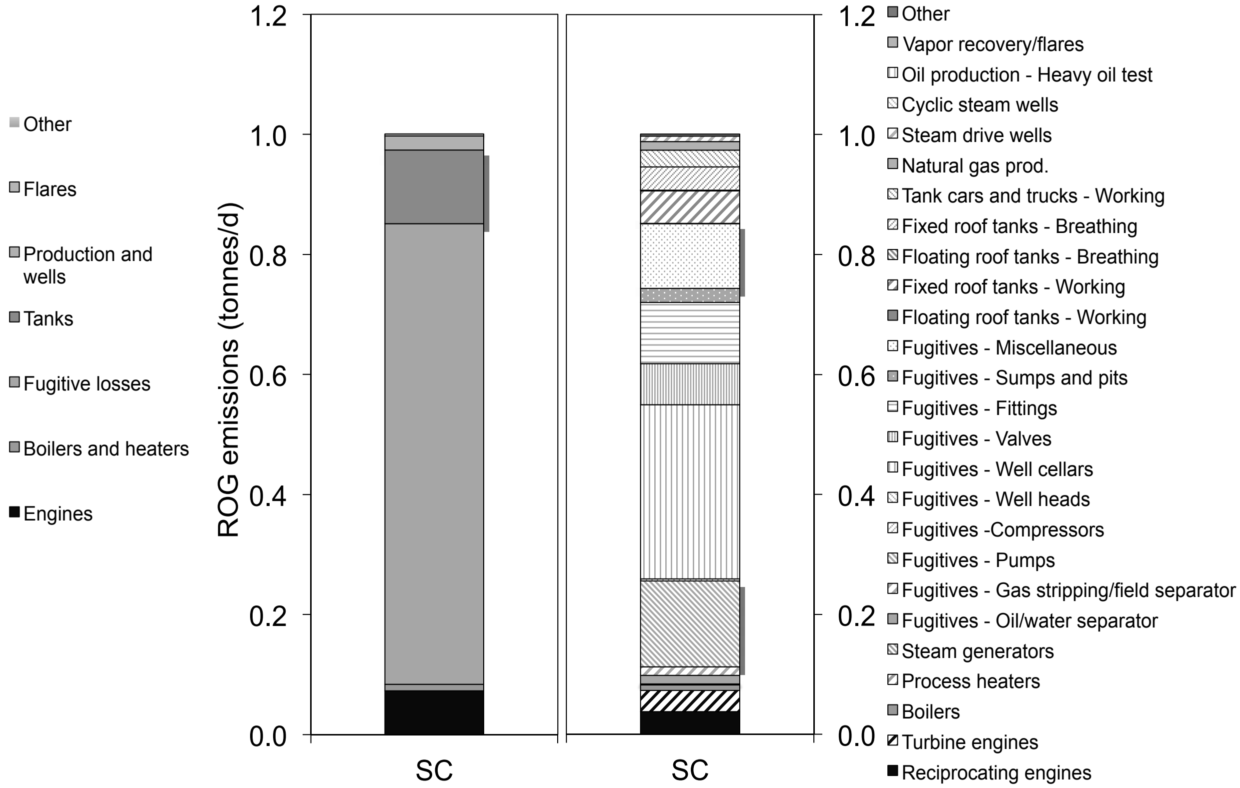

Figure 3.3-4. Stationary source 2012 emissions of reactive organic gas (ROG) from all stationary oil and gas production sources in the South Coast air district. Emissions in tonnes per day (1,000 kg/d).

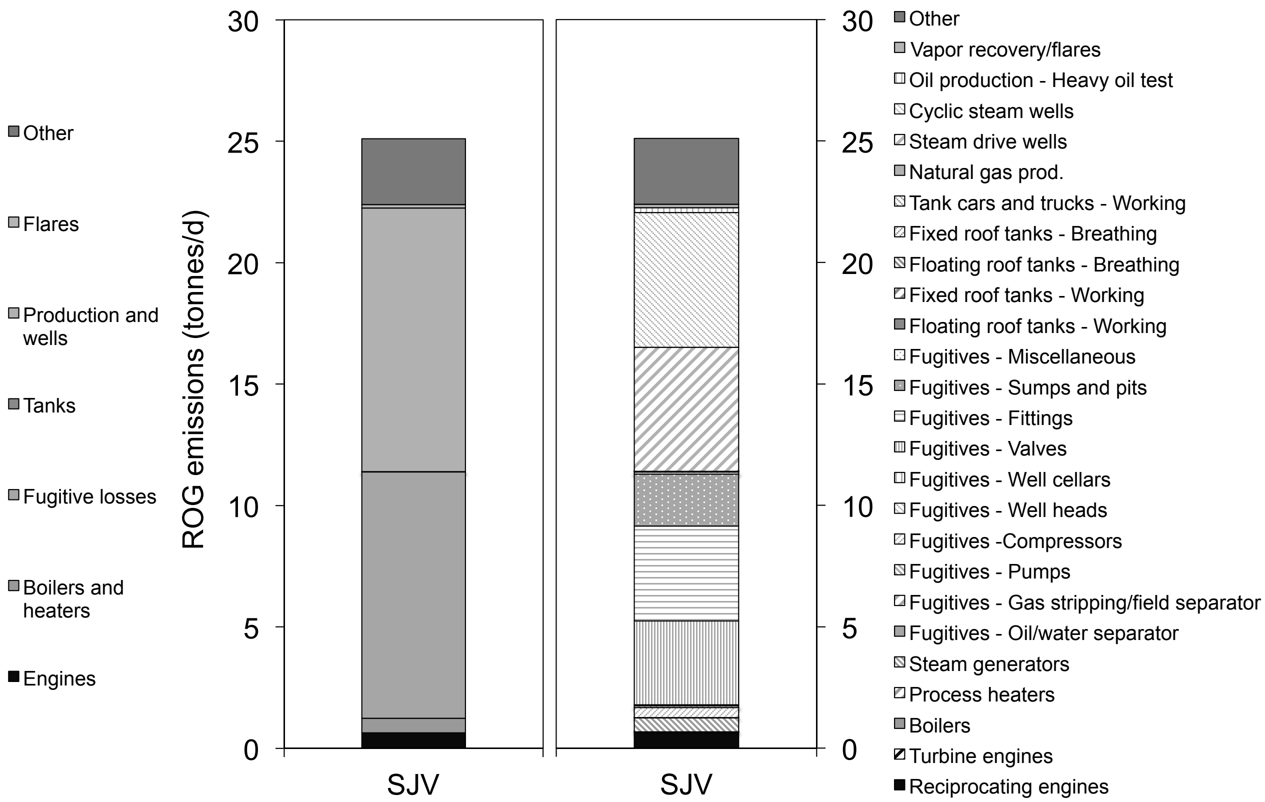

Figure 3.3-5. Stationary source 2012 emissions of reactive organic gas (ROG) from all stationary oil and gas production sources in the San Joaquin Valley Unified air district. Emissions in tonnes per day (1,000 kg/d).

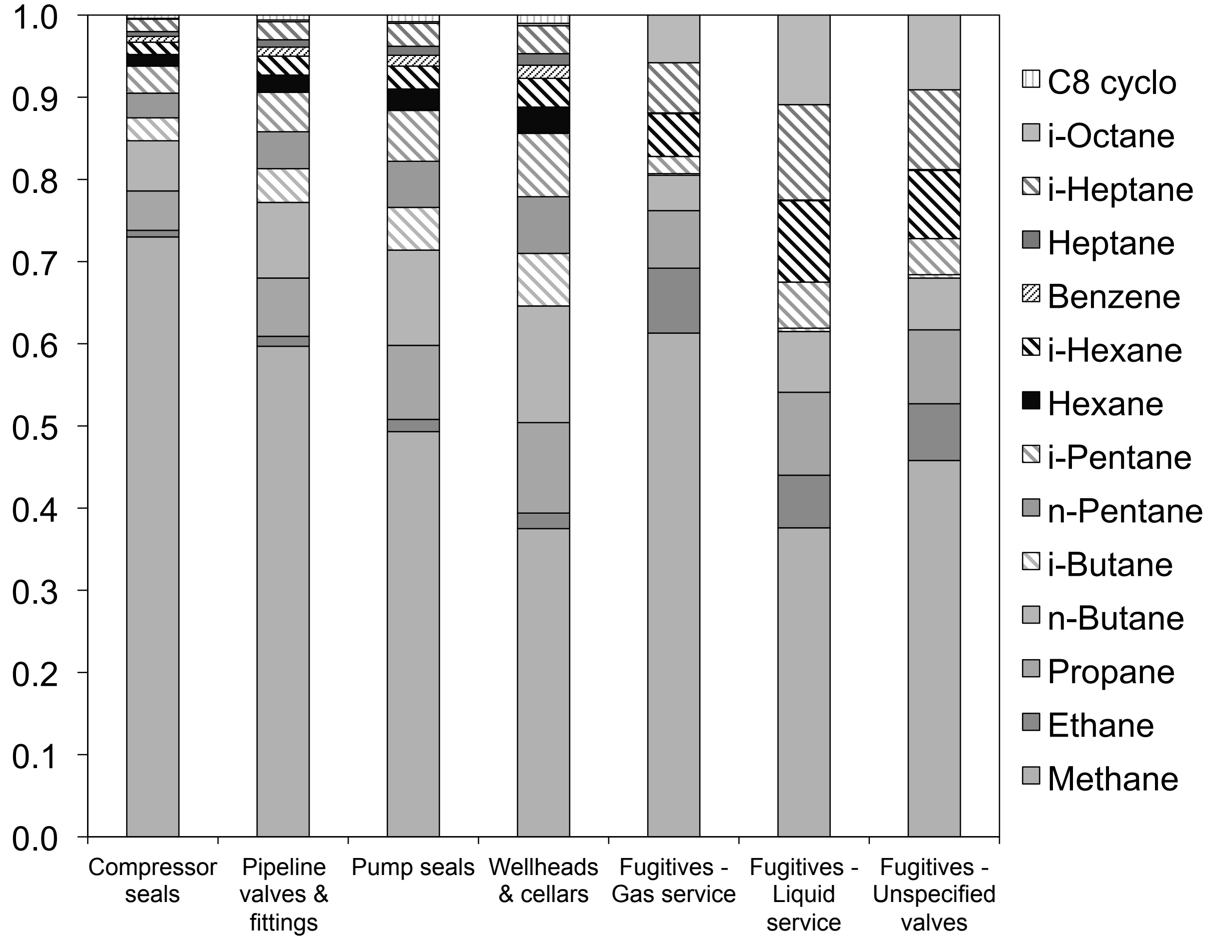

Figure 3.3-6. Stationary source speciation of TOG fugitive oil and gas production sources. Source: (CARB, 2014i)

Stationary source NO_x emissions from oil and gas sources can be computed similarly to the stationary source ROG/VOC emissions methods. Using the above methods, stationary source emissions in the "Oil and Gas Production" and "Oil and Gas Production (Combustion)" categories make up 1.1% and 0.3% of the stationary source NO_x emissions in the SJV and SC regions, respectively (CARB, 2013b).

Emissions from on-road vehicles associated with well stimulation or with the oil and gas industry cannot be partitioned from the inventory, but do represent some fraction of on-road ROG/VOC emissions.

ROG emissions from off-road oil and gas sources for the SJV and SC regions are shown in Figure 3.3-7. Off-road oil and gas ROG emissions in SJV region are 0.59 t per day. In the SJV air district, this is equivalent to 0.75% of mobile source ROG emissions and 0.18% of ROG/VOC emissions from all sources. In the SC region, ROG emissions from off-road oil and gas sources are 0.08 t per day: 0.04% of mobile-source ROG/VOC emissions, and 0.02% of total ROG emissions from all sources.

Using a reasonable assumption of the share of in-state on-road trucks used by the oil and gas industry, these sources will make up a small fraction of mobile source ROGs in California. This conclusion is not surprising, due to the relatively small size of the oil and gas sector in California compared to the many other industries supporting 40 million California residents.

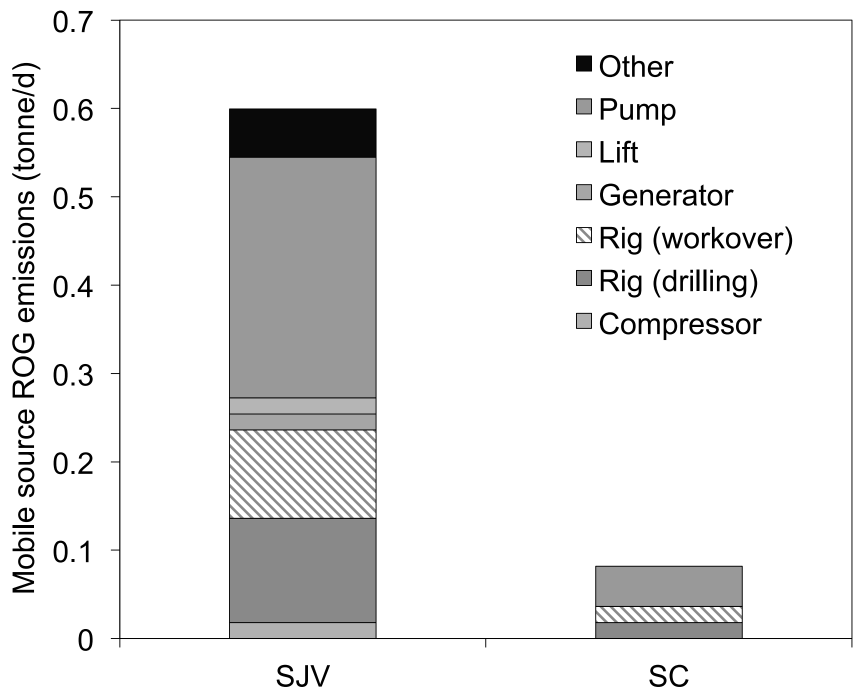

Figure 3.3-7. Mobile source 2012 emissions of reactive organic gas (ROG) from all sources within the categorization "Off road, oil drilling and workover, diesel (unspecified)" Emissions in tonnes per day (1,000 kg/d). Source: CARB (2013b).

The oil and gas industry represents a larger fraction of mobile source NO_x emissions and total NO_x emissions than ROG/VOC emissions. Using similar methods to those used above, NO_x emissions from off-road oil and gas equipment are 7.3 and 1.6 t per day in the SJV and SC regions, respectively. In the SJV region, this represents 2.9% of mobile source NO_x, and 2.5% of total NO_x emissions. In the SC region, oil and gas NO_x emissions of 1.6 per day represent 0.3% of total mobile sources and 0.3% of all NO_x sources (CARB, 2013b).

Summary of CARB criteria pollutant inventory coverage

A summary of all criteria pollutant emissions from oil and gas operations (stationary and mobile) is shown in Table 3.3-6.

Table 3.3-6. CARB criteria pollutant overview in emissions of criteria pollutants in t per day (1,000 kg/d). Includes all anthropogenic as well as all sources, natural and anthropogenic.

		ROG	**NO_x**	**SO_x**	**PM**	**PM10**	**PM2_5**
SJV	Oil and gas - Stationary	25.1	3.2	2.9	1.9	1.8	1.8
SJV	Oil and gas - Mobile	0.6	7.3	0.0	0.2	0.2	0.2
SJV	Total anthropogenic	324.9	295.2	9.4	479.8	255.7	68.7
SJV	Total anthropogenic + natural	1112.8	306.2	12.8	517.5	291.9	99.4
SJV	Percentage of anthropogenic	7.9%	3.6%	31.2%	0.4%	0.8%	3.0%
SJV	Percentage of total	2.3%	3.4%	23.0%	0.4%	0.7%	2.0%
		ROG	NO_x	SO_x	PM	PM10	PM2_5
SC	Oil and gas - Stationary	1.0	1.7	0.0	0.1	0.1	0.1
SC	Oil and gas - Mobile	0.1	1.6	0.0	0.0	0.0	0.0
SC	Total anthropogenic	438.2	517.7	17.7	228.3	157.6	66.7
SC	Total anthropogenic + natural	613.8	521.8	19.8	257.1	185.3	90.1
SC	Percentage of anthropogenic	0.2%	0.6%	0.1%	0.1%	0.1%	0.2%
SC	Percentage of total	0.2%	0.6%	0.1%	0.0%	0.1%	0.1%

The CARB criteria pollutant inventory is likely to include some emissions from well stimulation activities, but the level of coverage is uncertain. To summarize the above:

- Oil and gas criteria pollutant emissions are estimated using detailed categorization in stationary source and off-road mobile source databases. Criteria pollutant emissions from on-road oil and gas sources cannot be determined because the on-road emissions databases do not partition emissions by sector.

- Given the detailed categorization of stationary source emissions (e.g., "Fugitives—Well head") and off-road mobile source emissions (e.g., "Rigs—Workover") , it is possible that well-stimulation-related sources are being tracked. The level of well-stimulation-related coverage cannot be determined from reported data.

3.3.2.2.3. CARB Inventories for TACs

A variety of TACs can be released from well stimulation activities. Key TACs include VOC or fugitive hydrocarbon emissions, particulate matter (discussed separately below), and emission of substances used in hydraulic fracturing fluids.

Because of the large scope and complexity of TACs emissions (both in number of species and number of emissions processes), all results in this section are computed for 10 indicator TACs that can be emitted from oil and gas sources. These indicator TACs include the largest 5 sources from a recent EPA risk assessment for oil and natural gas production (U.S. EPA, 2011). This source lists oil and gas production associated TACs ordered by rate of emissions across 990 facilities in an EPA dataset (U.S. EPA, 2011, Table 4.1.-1). Next, ethyl benzene is included as in indicator TAC due to its importance in the suite of BTEX (benzene, toluene, ethylbenzene, and xylenes) hydrocarbon emissions (ethyl benzene was also ranked 6th in the EPA list by emissions rate). Hydrogen sulfide is included as an important hydrocarbon-related compound with potential health effects. We note that hydrogen sulfide is not technically classified as a TAC (U.S. EPA, 2014), but serious human health impacts are associated with breathing small amounts of hydrogen sulfide, resulting in stringent safety requirements and controls around hydrogen sulfide (H2S) releases. Lastly, four indicator species are added that were found upon inspection of CARB databases to be significantly driven by oil and gas sector sources.

The resulting 10 indicator TACs species are: 1,3-butadiene, acetaldehyde, benzene, carbonyl sulfide, ethyl benzene, formaldehyde, hexane, hydrogen sulfide, toluene, and xylenes (mixed). This list is not meant to be exhaustive of all possible species of interest, but indicative of the possible contributions of oil and gas sources.

CARB TAC inventory methods

TACs emissions by species for a broad variety of sources are reported in the CARB California Toxics Inventory (CTI). The CTI is not computed frequently: data were reported most recently for year 2010 (CARB, 2013c). Unlike the facility-scale “hot spot” dataset (see below), the CTI includes a variety of nonstationary sources, such as area-wide sources and mobile sources (gasoline, diesel, off-road equipment, etc.).

The CTI reports emissions by air district for ~340 toxic species (CARB, 2013c). These data are compiled from facility-level data noted above, as well as mobile sources and dispersed stationary sources such as homes and nonreporting businesses. TACs from mobile sources, which are not otherwise subject to air toxics reporting requirements, are estimated by applying speciation factors to criteria pollutant inventories noted above (CARB, 2013d). For example, ROG emissions from off-road combustion are obtained from the criteria pollutant inventory described above, and speciation factors are applied to these ROG emissions estimates to estimate emissions of a given TAC chemical species (CARB, 2000).

TACs from regulated stationary sources are recorded in CARB datasets at the facility level (CARB, 2014j). Emissions data are reported to CARB from stationary facilities as part of the Air Toxics “Hot Spots” program (AB 2588, enacted 1987). Various criteria are used to determine whether a facility must report data to the CARB (CARB, 2013e; 2014j; 2014l), but a chief criterion is the manufacture, formulation, use, or release of any of 600 substances subject to the regulation. Reporting requirements differ by chemical species or substance. Some species/substances require reporting only with regard to air emissions, while other species/substances are required to be reported if used or manufactured, regardless of estimated air emissions rate.

Facility-level TACs data are searchable by Standard Industrial Classification (SIC) code, air district, county, facility code, and chemical species. These facility-level data are further compiled by air districts, which publish annual reports summarizing emissions of TACs within each district from all sources (e.g., CARB, 2014k).

More recently, the South Coast Air Quality Management District (SCAQMD, or SC air district as above) passed legislation—Rule 1148.2—which requires reporting of use of potential TACs in oil and gas well stimulation (SCAQMD, 2013; PSR et al., 2014). This rule goes beyond existing TACs reporting requirements by specifically requiring reporting of the volume or mass of use of certain chemicals which are TACs, rather than reporting the estimated emissions rate.

In 2010, in the SJV air district, TACs of importance included (in order of mass rate of emissions): acetaldehyde, diesel PM, formaldehyde, and benzene (SJVAPCD, 2014). Mobile sources are responsible for over half of SJV TACs, while stationary sources were responsible for ~15% of emissions (SJVAPCD, 2014). Three of these four species are in the set of 10 indicator TACs species, and diesel PM is discussed further below.

Coverage of WS activities in CARB TAC inventory

Direct well-stimulation-related TAC emissions will occur in the upstream portion of the oil and gas industry. Key possible TACs impacts from well stimulation activities include:

- Release of hydrocarbons during the well completion (“flowback”) process;
- Release or volatilization of components of the fracturing fluid, which could represent toxic hazards;
- Release of combustion byproducts or hydrocarbon (HC) fugitives from consumption of fuels during WS activities (e.g., by pumps, generators, compressors, or other on-site engines);

The above activities could result in TAC emissions from a mixture of point-source and mobile source emissions. To the extent that stationary sources associated with oil and gas report TAC emissions as part of AB 2588, these emissions will be included in the TAC inventory. Given the detailed source categories treated in the off-road mobile source inventory (noted above), it is likely that at least some mobile source TACs from well stimulation activities are counted in the current inventory.

It is not clear how emissions unique to well stimulation (e.g., emissions during fracturing fluid preparation, injection or flowback) are treated in current TAC inventory methods. No data exist in either the "hot spots" dataset or in the CTI to clearly differentiate well stimulation from non-well-stimulation oil and gas emissions.

Results of CARB TAC inventory

Results of the CTI for the most recent year (2010) are presented in Figure 3.3-8 and Figure 3.3-9 for the SJV and SC regions respectively. Tabular data are presented in Table 3.3-7 and Table 3.3-8 for the SJV and SC regions respectively.

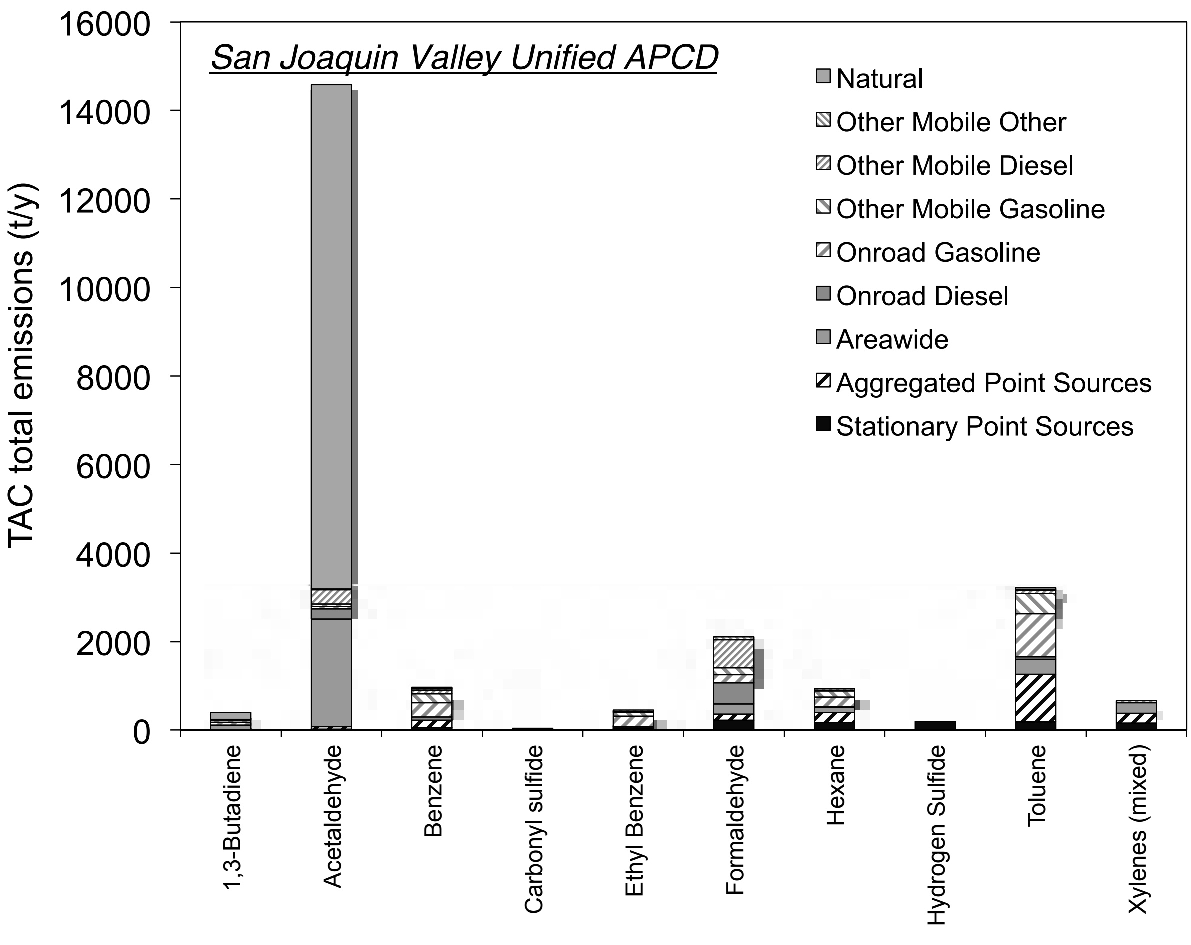

Figure 3.3-8. Total TACs releases for 10 indicator species in SJV region. Results for calendar year 2010, most recent available (CARB, 2013c). Emissions are in tonnes per year (1,000kg/y).

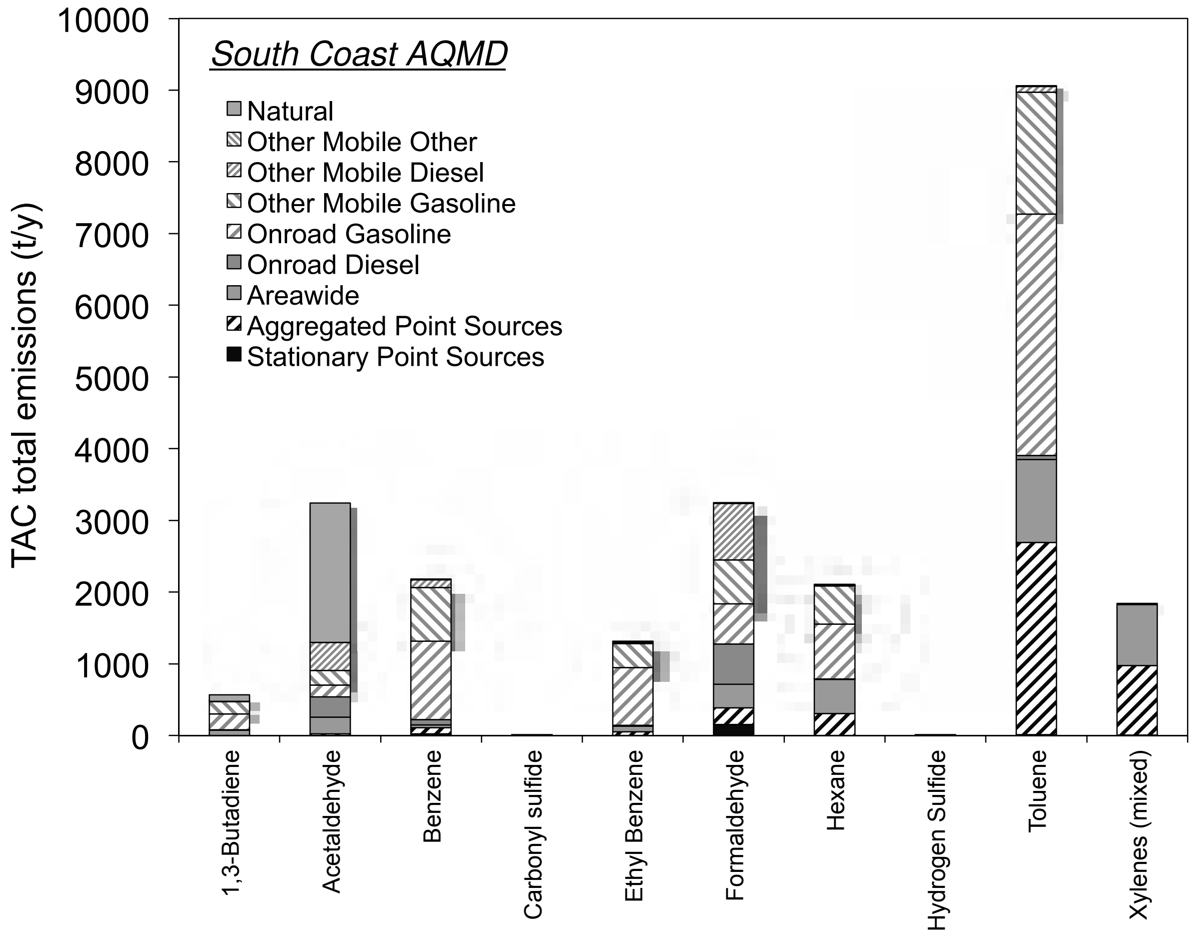

Figure 3.3-9. Total TACs releases for 10 indicator species in SC region. Results for calendar year 2010, most recent available (CARB, 2013c). Emissions are in tonnes per year (1,000kg/y).

Table 3.3-7. Overall toxics inventory results for indicator species in SJV region. Emissions are in tonnes per year (1,000kg/y). Results from 2010 calendar year (CARB, 2013c).

	Stationary Point Sources	Aggregated Point Sources	Area-wide	Onroad Diesel	Onroad Gasoline	Other Mobile Gasoline	Other Mobile Diesel	Other Mobile Other	Natural	Total	Fraction stationary point sources
1,3-Butadiene	0.4	1.4	105.9	6.1	67.3	44.5	8.2	10.3	150.9	395.0	0.1%
Acetaldehyde	20.4	53.3	2432.3	237.7	53.9	52.5	317.0	21.4	11395.1	14583.6	0.1%
Benzene	62.2	155.2	10.7	64.7	328.4	197.8	86.3	21.3	0.7	927.2	6.7%
Carbonyl sulfide	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0%
Ethyl benzene	19.9	24.3	26.3	10.0	232.4	88.1	13.4	6.1	0.0	420.5	4.7%
Formaldehyde	222.5	141.8	231.1	475.7	178.8	160.8	634.4	59.7	0.0	2104.7	10.6%
Hexane	168.5	229.1	120.3	5.2	221.9	143.1	6.9	3.3	0.0	898.3	18.8%
Hydrogen sulfide	193.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	193.8	100.0%
Toluene	189.9	1074.7	340.3	47.5	980.7	451.7	63.4	29.7	0.7	3178.6	6.0%
Xylenes (mixed)	161.7	216.0	244.8	0.0	0.0	0.0	0.0	0.0	0.0	622.5	26.0%

Table 3.3-8. Overall toxics inventory results for indicator species in SC region. Emissions are in tonnes per year (1,000kg/y). Results from 2010 calendar year (CARB, 2013c).

	Stationary Point Sources	**Aggregated Point Sources**	**Area-wide**	**Onroad Diesel**	**Onroad Gasoline**	**Other Mobile Gasoline**	**Other Mobile Diesel**	**Other Mobile Other**	**Natural**	**Total**	**Fraction stationary**
1,3-Butadiene	2.9	0.1	72.6	7.2	217.8	168.7	10.2	0.0	86.5	566.1	0.5%
Acetaldehyde	2.8	21.4	232.8	280.3	168.9	199.2	393.5	0.2	1944.2	3243.4	0.1%
Benzene	23.2	83.2	39.9	76.3	1092.7	747.8	107.1	1.4	0.0	2171.5	1.1%
Carbonyl sulfide	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	100.0%
Ethyl Benzene	3.6	46.3	81.8	11.8	806.0	333.3	16.6	0.1	0.0	1299.5	0.3%
Formaldehyde	152.4	231.6	328.7	561.0	563.3	609.9	787.6	5.5	0.0	3240.0	4.7%
Hexane	1.9	304.1	478.2	6.1	763.2	531.8	8.6	0.6	0.0	2094.6	0.1%
Hydrogen Sulfide	9.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.3	100.0%
Toluene	10.0	2680.9	1160.0	56.1	3363.4	1701.3	78.7	1.0	0.0	9051.4	0.1%
Xylenes (mixed)	8.5	966.0	852.7	0.0	0.0	0.0	0.0	0.2	0.0	1827.5	0.5%

As can be seen from Table 3.3-7 and Table 3.3-8, key sources of the indicator TACs species in both SJV and SC region include vehicular sources (gasoline in particular) and aggregated (i.e., not individually reported) point sources. Some species (carbonyl sulfide and hydrogen sulfide) are emitted primarily or completely by stationary facilities. Facilities that report TACs emissions as part of the point-source reporting program are discussed in more detail below.

To the extent that oil and gas development contributes to overall diesel consumption (both on-road and off-road), some contribution of TACs from oil and gas activities in these categories is to be expected. Note that Figure 3.3-7 and the associated discussion suggest that the importance of ROGs from oil and gas related mobile sources is likely to be small. It is therefore likely that TACs from mobile source oil and gas activities are also small. However, no sector- or activity-level breakdown is available in the CTI TACs database as was available in the ROG database.

In contrast to the overall CTI results which cover all sources (stationary, areawide, mobile, natural), a much more detailed facility-level inventory is generated using reported data for facilities under the "Hot Spots" program, AB 2588. Using this data, it is possible to estimate TACs impacts of oil and gas activities from stationary source reporting facilities. In order to do this, the facility-level TACs databases were searched using Standard Industrial Classification (SIC) codes representing upstream oil and gas activities. Using OSHA (Occupational Safety and Health Administration) databases of SIC codes, 12 codes were determined to be related to oil and gas activities. Five of these codes are included in this report's estimates of upstream oil and gas activities (see Table 3.3-9).

It is not possible to separate these facility-level stationary-source TACs emissions by oilfield or pools. The reporting facilities to the TACs database do not line up with pools or fields as reported in DOGGR databases. Also TACs emissions source facilities in the database are sometimes very generally defined (e.g., "AERA Energy LLC, heavy oil production")

Table 3.3-9. SIC codes used in analysis of facility-level TAC emissions. Source: OSHA SIC database search for “petroleum” and “oil” (OSHA, 2014).

SIC code	Description	Included as upstream O&G?
1311	Crude petroleum and natural gas	Y
1321	Natural gas liquids	Y
1381	Drilling oil and gas wells	Y
1382	Oil and gas field exploration services	Y
1389	Oil and gas field services, not elsewhere classified	Y
2911	Petroleum refining	N
2922	Lubricants and greases	N
3533	Oil and gas machinery	N
4613	Refined petroleum pipelines	N
5172	Petroleum product wholesalers	N
5541	Gasoline stations	N
5983	Fuel oil dealers	N

Unlike the stationary source criteria pollutant inventory, no data source was found that separates TACs emissions by subsource (CARB, 2013c).

The distribution of facility-reported emissions for the 10 indicator species is shown in Figure 3.3-10 and Figure 3.3-11 for the SJV and SC regions, respectively. Tabular results are shown in Table 3.3-10 and Table 3.3-11 for SJV and SC regions. These values differentiate between emissions from the five SIC codes noted in the above table (“Y”) and aggregate emissions from all other SIC codes. The five upstream oil and gas SIC codes noted in Table 3.3-9 are responsible for between 0% and 70% of the emissions of these species from stationary sources in the SJV air district. In the SC air district, these upstream stationary oil and gas sources were responsible for 0% to 10% of the emissions of these species from all stationary sources.

Because treatment of mobile source TACs in the CTI is derived from speciation of the criteria pollutant inventory, coverage of TACs from mobile sources associated with well stimulation or oil and gas activities will be subject to the same issues noted above. To recapitulate, a variety of mobile sources relevant to oil and gas (and presumably to well stimulation) activities are tracked, especially for off-road diesel equipment. However, it is not clear how to apportion these activities between conventional HC production and well stimulation activities without detailed study.

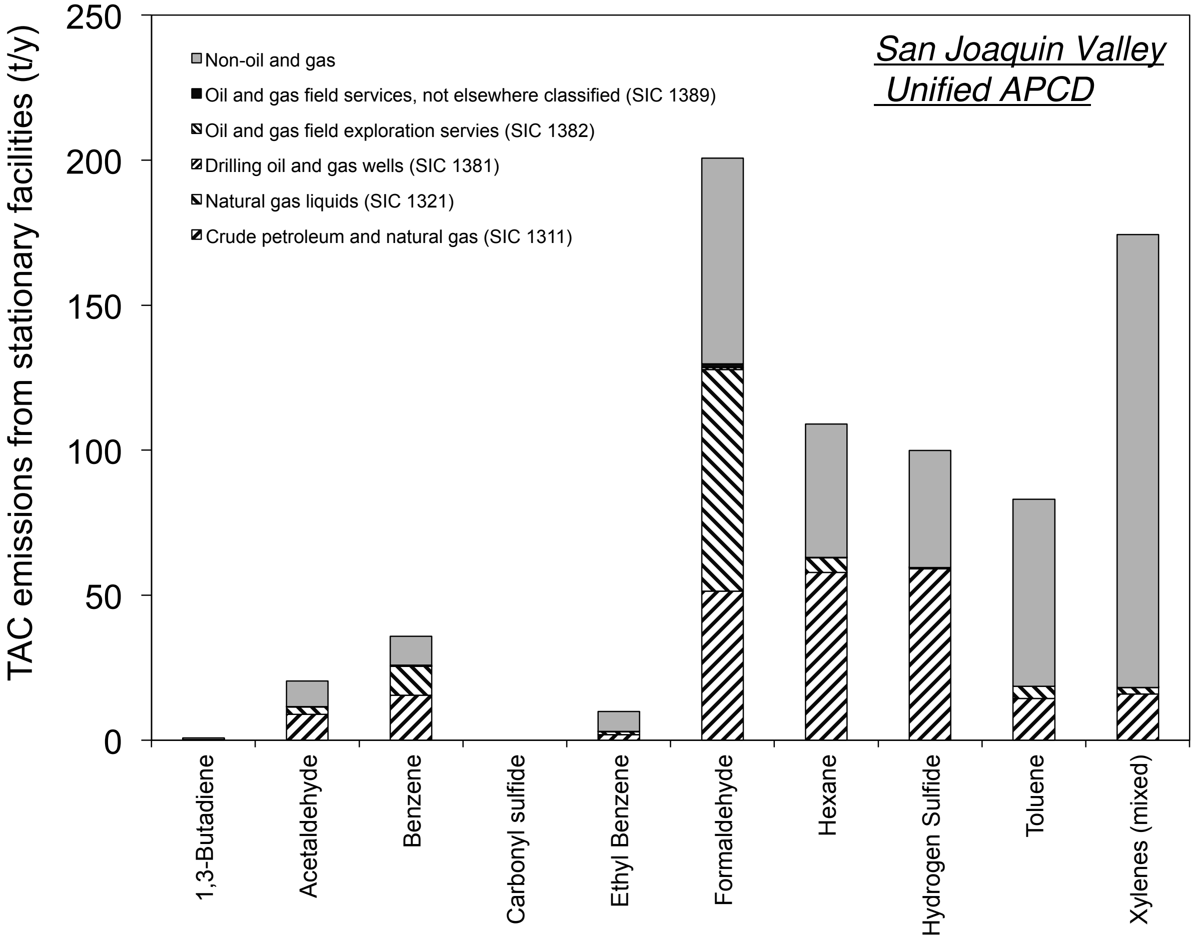

Figure 3.3-10. Summed facility-level TAC emissions in San Joaquin Valley (SJV) air district CARB, 2014j). Emissions plotted for indicator species for SIC codes 1311, 1321, 1381, 1382, and 1389. Facility-level emissions derived from CARB facility emissions tool. Total emissions are emissions from all SIC codes in the air district, including gasoline fueling stations.

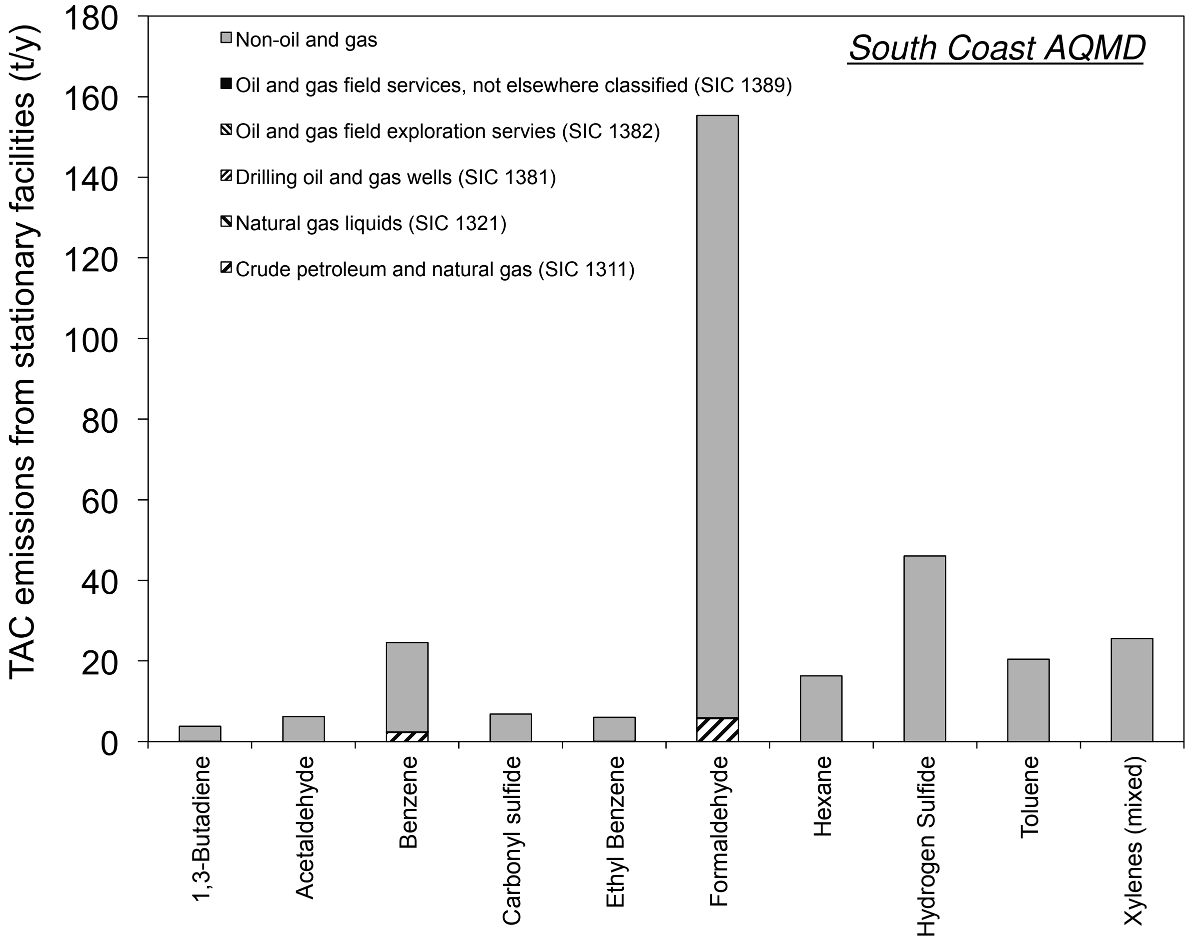

Figure 3.3-11. Summed facility-level TAC emissions in South Coast (SC) air district CARB, 2014j). Emissions plotted for indicator species for SIC codes 1311, 1321, 1381, 1382, and 1389. Facility-level emissions derived from CARB facility emissions tool. Total emissions are emissions from all SIC codes in the air district, including gasoline fueling stations.

Table 3.3-10. Emissions rates from stationary facilities in SJV region, as reported to facility-level reported TACs database (CARB, 2014j). Data from calendar year 2012. All emissions in tonnes per year (1,000kg/y).

	Crude petroleum and natural gas (SIC 1311)	Natural gas liquids (SIC 1321)	Drilling oil and gas wells (SIC 1381)	Oil and gas field exploration services (SIC 1382)	Oil and gas field services, not elsewhere classified (SIC 1389)	Total oil and gas (SIC 1311-1389)	Non-oil and gas	Percentage oil and gas
1,3-Butadiene	0.2	0.0	0.0	0.0	0.0	0.3	0.5	35.2%
Acetaldehyde	8.9	2.5	0.0	0.0	0.2	11.6	8.8	57.0%
Benzene	15.6	9.8	0.0	0.1	0.5	25.9	9.9	72.4%
Carbonyl sulfide	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0%
Ethyl Benzene	1.9	1.0	0.0	0.0	0.2	3.1	6.8	31.2%
Formaldehyde	51.4	76.5	0.0	0.6	1.3	129.8	70.9	64.7%
Hexane	57.9	5.0	0.0	0.0	0.1	63.0	46.0	57.8%
Hydrogen Sulfide	59.2	0.3	0.0	0.0	0.1	59.5	40.3	59.6%
Toluene	14.4	4.2	0.0	0.0	0.1	18.6	64.4	22.4%
Xylenes (mixed)	15.9	2.3	0.0	0.0	0.0	18.2	156.1	10.4%

Table 3.3-11. Emissions rates from stationary facilities in SC region, as reported to facility-level reported TACs database (CARB, 2014j). Data from calendar year 2012. All emissions in tonnes per year (1,000kg/y).

	Crude petroleum and natural gas (SIC 1311)	**Natural gas liquids (SIC 1321)**	**Drilling oil and gas wells (SIC 1381)**	**Oil and gas field exploration services (SIC 1382)**	**Oil and gas field services, not elsewhere classified (SIC 1389)**	**Total oil and gas (SIC 1311-1389)**	**Non-oil and gas**	**Percentage oil and gas**
1,3-Butadiene	0.1	0.0	0.0	0.0	0.0	0.1	3.7	1.5%
Acetaldehyde	0.0	0.0	0.0	0.0	0.0	0.0	6.3	0.0%
Benzene	2.2	0.0	0.1	0.0	0.0	2.4	22.2	9.6%
Carbonyl sulfide	0.0	0.0	0.0	0.0	0.0	0.0	6.9	0.0%
Ethyl Benzene	0.0	0.0	0.0	0.0	0.0	0.0	6.0	0.5%
Formaldehyde	5.7	0.0	0.0	0.0	0.1	5.9	149.5	3.8%
Hexane	0.0	0.0	0.0	0.0	0.0	0.0	16.3	0.0%
Hydrogen Sulfide	0.0	0.0	0.0	0.0	0.0	0.0	46.0	0.0%
Toluene	0.0	0.0	0.0	0.0	0.0	0.0	20.4	0.0%
Xylenes (mixed)	0.0	0.0	0.0	0.0	0.0	0.0	25.6	0.0%

It does not appear possible to directly compare these two datasets. While the CTI (produced less frequently) reports emissions from "stationary point sources" that are in theory derived from the facility-level reporting explored above by SIC code, examination of the data for the 2010 year from both datasets shows discrepancies for all ten indicator species. For example, benzene emissions estimated by CTI datasets for the SJV region (defined as the San Joaquin Valley Unified APCD) in 2010 were 62.1 t/y for the category "stationary point sources." In contrast, querying the facility-level toxics database for 2010 for the same region, and summing resulting benzene emissions from all reporting facilities, results in total emissions of 38.6 t/y.

Given the above caveat, an approximate estimate of the relative importance of oil and gas stationary sources can be generated by comparing the upstream oil and gas facility level emissions (summed by SIC code as above) to the total CTI results for the same year. These results are shown in Table 3.3-12 and Table 3.3-13 below.

In summary, in the SJV region, upstream oil and gas point-source facilities are responsible for the great majority of H2S emissions (>70%) and are small contributors to emissions of benzene, formaldehyde, hexane and xylenes (1-10%). In the SC region, oil and gas sources are negligible contributors to emissions of our ten indicator TACs.

Note again that there will also be oil and gas mobile source TACs that are not accounted for in Table 3.3-12 and Table 3.3-13. Because oil and gas mobile sources are small contributors to both ROG and PM emissions (see Table 3.3-6), this is unlikely to affect the general results of this comparison.

Table 3.3-12. San Joaquin Valley oil and gas facility-reported emissions of ten indicator TACs compared to California Toxics Inventory estimates for all sources, both for year 2010. Emissions in tonnes (1,000 kg) per year.

SJVUAPCD	Total oil and gas (t/y, facility-level 2010)	Total all sources (t/y, CTI 2010)	Fraction
1,3-Butadiene	0.1	435.1	0.0%
Acetaldehyde	10.8	16061.2	0.1%
Benzene	24.2	1021.2	2.4%
Carbonyl sulfide	0.0	0.0	0.0%
Ethyl Benzene	2.3	463.1	0.5%
Formaldehyde	126.9	2318.0	5.5%
Hexane	47.2	989.3	4.8%
Hydrogen Sulfide	151.2	213.4	70.8%
Toluene	17.6	3500.7	0.5%
Xylenes (mixed)	17.6	685.6	2.6%

Table 3.3-13. South Coast oil and gas facility-reported emissions of ten indicator TACs compared to California Toxics Inventory estimates for all sources, both for year 2010. Emissions in tonnes (1,000 kg) per year.

SCAQMD	Total oil and gas (t/y, facility-level 2010)	Total all sources (t/y, CTI 2010)	Fraction
1,3-Butadiene	0.3	623.4	0.0%
Acetaldehyde	0.0	3572.1	0.0%
Benzene	2.3	2391.5	0.1%
Carbonyl sulfide	0.0	0.1	0.0%
Ethyl Benzene	0.0	1431.1	0.0%
Formaldehyde	8.5	3568.3	0.2%
Hexane	0.1	2306.8	0.0%
Hydrogen Sulfide	0.0	10.2	0.0%
Toluene	0.1	9968.5	0.0%
Xylenes (mixed)	0.1	2012.7	0.0%

3.3.2.2.4. SCAQMD Reporting of Hazardous Materials

The South Coast Air Quality Management District (SCAQMD, or SC region as above) recently approved regulation (Rule 1148.2), which requires the reporting of use of potentially hazardous materials in well stimulation, drilling, or workover activities. The chemicals which were reported in this regulation, as well as the average quantities injected or used, are tabulated in Table 3.3-14 and Table 3.3-15.

The SCAQMD database does not directly report masses of chemicals injected. For all operations reported in the SCAQMD database, the mass flow of each injected material (e.g., proppant) was reported, as well as the "maximum concentration" of a number of individual chemical constituents (e.g., proppant might be made of crystalline silica (max 95%) and phenol-formaldehyde resin (max 5%). These data are combined to determine a maximum injection rate for individual chemicals.

Table 3.3-14. TAC species associated with fracturing fluids extracted from SCAQMD dataset.

Toxic air contaminant	**Average maximum kg injected per well**
Crystalline Silica Quartz (SiO2)	67060
Phenol-Formaldehyde Resin	16369
Methanol	1619
Hydrogen Chloride	622
Ethylene Glycol	443
Hydrofluoric Acid	45
2-Butoxy Ethanol	37
Hexamethylene Tetramine	33
Sodium Hydroxide	31
Silica Fumed	2
Cristobalite (SiO2)	1

Table 3.3-15. TAC species associated in matrix acidizing extracted from SCAQMD dataset.

Toxic air contaminant	**Average max. mass injected (kg)**
Crystalline Silica (Quartz)	3546
Hydrocholoric Acid	1058
Phosphonic Acid	406
Aminotriacetic Acid	309
Xylene	207
Hydrofluoric Acid	179
2-Butoxy Ethanol	213
Ethylbenzene	63
Methanol	34
Thiourea Polymer	15
Isopropanol	13
Sulfuric Acid Ammonium Salt (1:2)	7
Acrylic Polymer	7
Toluene	4
1,2,4 Trimethylbenzene	2
Diethylene Glycol	1
Ethylene Glycol	1
Naphthalene	1
Cumene	<1

These reported chemicals are not universally present in Clean Air Act lists of TACs, but their usage is required to be reported by SCAQMD. Also, the list of chemicals reported to SCAQMD may differ from other lists of potential toxics noted elsewhere in this report.

Additives and components of fracturing fluids could potentially be released to the air during mixing and preparation, injection, or flowback of fracturing fluids. The volatility of each of the additives can vary. For example, the largest mass additive is crystalline silica quartz (proppant). This proppant is not generally volatile, and only the proppant fines are of a concern from an air quality perspective.

3.3.2.2.5. Naturally Occurring Radioactive Materials

One possible concern about hydraulic fracturing is the release of naturally occurring radioactive materials (NORM). NORM can result in contamination of water with radioactive species, as well as result in air impacts through liberation of species that can enter gaseous phase (e.g. radon).

Though NORM is a serious concern for some shale formations (in particular the Marcellus formation of Pennsylvania, where it poses serious water quality issues), it is seen as less concerning in California (U.S. EPA, 2015).

California does contain deposits of radioactive elements in Kern County (USGS, 1954; USGS, 1960). However, these deposits are found to the south and east of the Kern County oilfields (USGS,1954; USGS, 1960). EPA studies suggest that well fluids and oilfield equipment in California are not significantly affected by radioactive species (U.S. EPA, 2015).

3.3.2.2.6. CARB Inventories for PM emissions

As with the above-described ROG and NO_x inventories, CARB criteria pollutant inventories track PM emissions of various classifications from both stationary and mobile sources.

CARB PM inventory methods

The CARB criteria pollutant inventory also estimates emissions of total particulate matter (PM) as well as PM10 and PM2.5.

The stationary source PM inventory is performed using the same classification and categorization scheme noted above for the stationary source ROG and NO_x inventories. See discussion above for details of stationary source inventory construction and categorization.

The mobile source PM inventory is performed using the same classification and categorization scheme noted above for the mobile source ROG and NO_x inventories. As noted above, on-road mobile source emissions are not clearly differentiated into oil and gas-associated emissions sources. Off-road mobile source emissions have a detailed classification for oilfield equipment (see above).

Total PM emissions estimated for a given source are partitioned into PM size bins using a set of standard PM speciation factors for ~500 stationary sources, mobile sources, or industrial/agricultural activities (CARB, 2014m). Oil and gas-specific PM size data are not available, but data are available for multiple categories of off-road diesel vehicles, which will comprise the majority of well-stimulation-site emissions of PM. These emissions are differentiated by type of vehicle and vehicle age (CARB, 2014m). Given that oil and gas- or well-stimulation-specific PM will often be emitted from processes similar to those used across industries (e.g., heavy diesel equipment), the use of non-oil and gas-specific PM speciation factors is a reasonable approach.

Coverage of WS activities in CARB PM inventory

Coverage and scope of well-stimulation-associated PM emissions in the CARB PM inventory should be identical to coverage and scope noted above for the ROG and NO_x inventories, because the inventory structure is similar.

Results of CARB PM inventory

Oil and gas stationary sources in the SJV region are responsible for 1.9 t per day of total PM and a nearly identical amount of PM2.5 (see Figure 3.3-12). This amounts to 0.4% of stationary anthropogenic emissions of total PM and 2.7% of stationary anthropogenic emissions of PM2.5. In the SC region, total stationary oil and gas-related sources of PM and PM2.5 equal 0.04% of stationary anthropogenic emissions of total PM and 0.13% of stationary anthropogenic emissions of PM2.5 in the SC region.

In the SJV region, oil and gas off-road sources are responsible for 0.2 and 0.18 t per day of total PM and PM2.5, respectively (see Figure 3.3-13). These equal 1.4% and 1.6% of the mobile-source PM in the SJV. However, due to large area-wide sources of PM in the SJV, off-road oil and gas sources are only responsible for 0.04% of total PM and 0.26% of PM2.5. In the SC region, off-road oil and gas sources emit some 0.03 t per day of PM and PM2.5 (see Figure 3.3-13). This represents ~0.3% of total mobile-source PM and 0.01% of total PM from all sources.

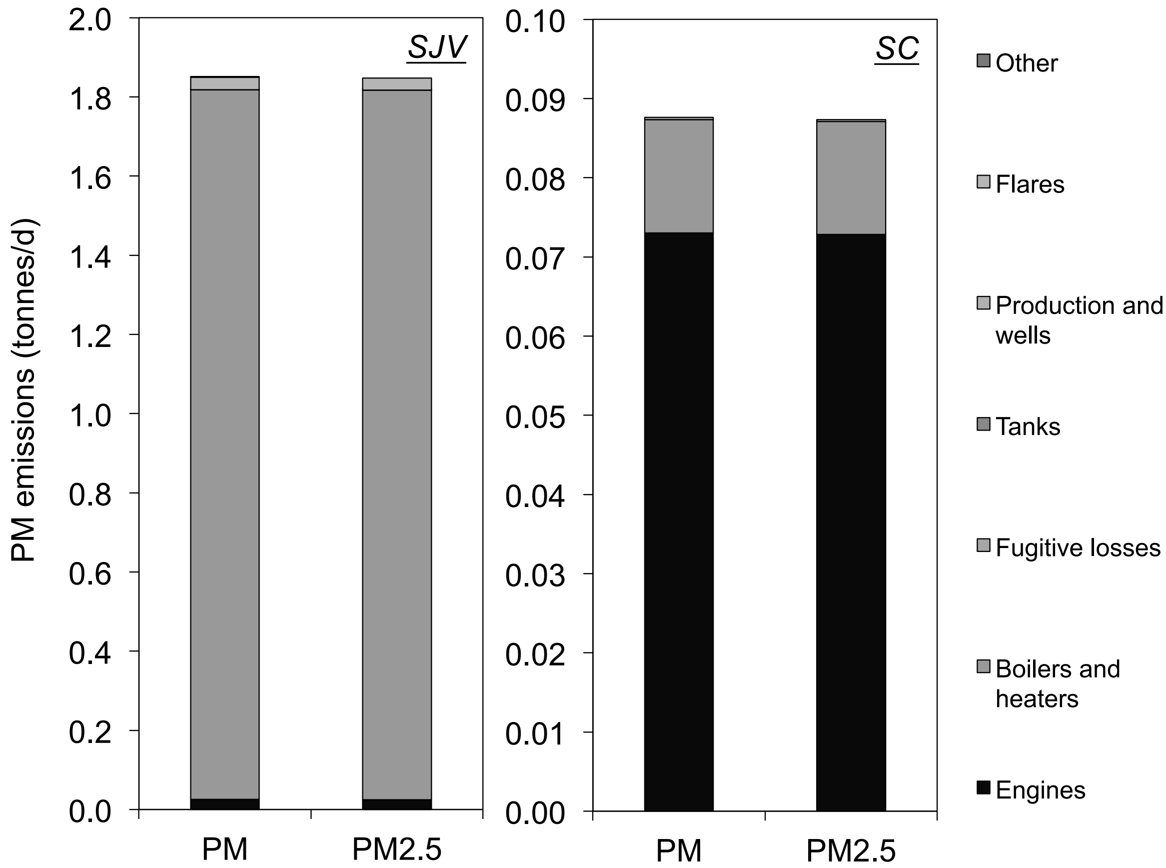

Figure 3.3-12. 2012 stationary source emissions of total particulate matter (PM) and particulate matter of less than 2.5 micrometer (PM2.5) from all oil and gas production sources in San Joaquin Valley and South Coast air districts. Source: CARB (2013b).

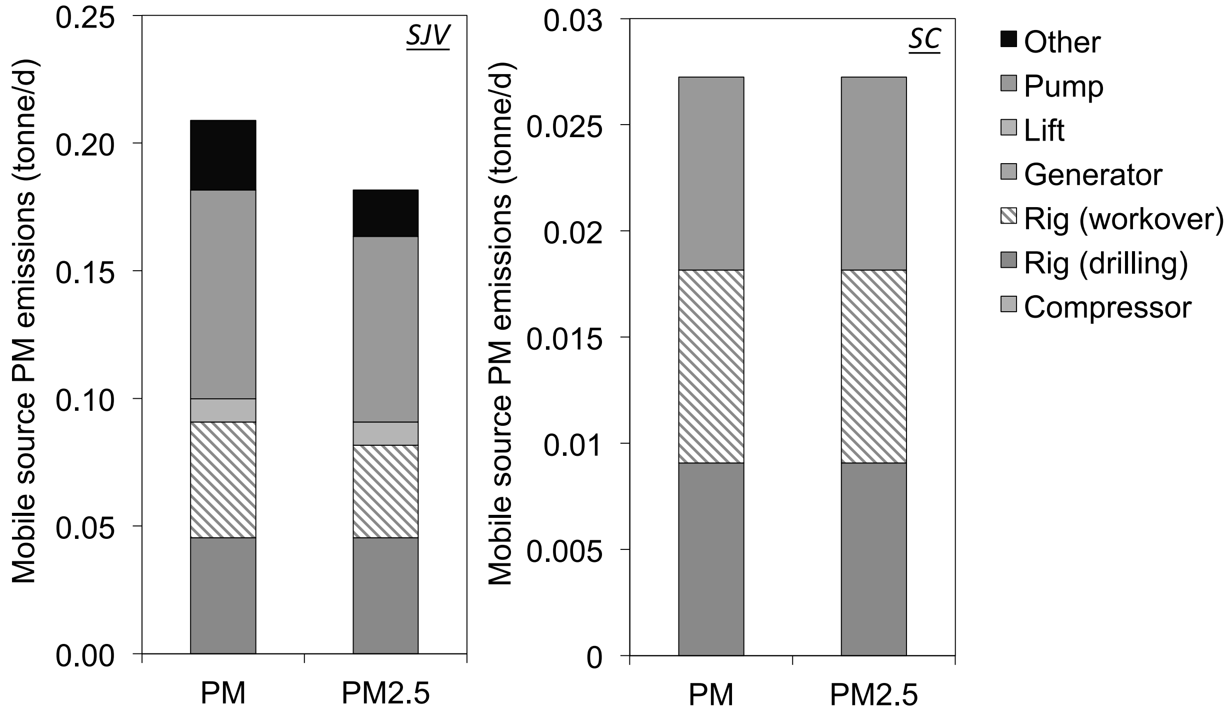

Figure 3.3-13. 2012 off-road mobile source emissions of total particulate matter (PM) and particulate matter of less than 2.5 micrometer (PM2.5) from all oil and gas production sources in San Joaquin Valley and South Coast air districts. Source: CARB (2013b).

3.3.2.2.7. CARB Inventories of Dust Emissions in Rural Regions

A major concern for air quality in rural California is the presence of dust from agricultural activities and other industrial activities occurring on non-paved surfaces. Dust is of particular concern in the SJV, where it contributes to the high levels of PM found in SJV air. If well stimulation technologies significantly affected regional dust levels, this could be an important air quality impact.

CARB creates inventories of dust emissions as part of their criteria pollutant inventory, adding dust emissions from all sources to PM, PM10, and PM2.5 totals. The breakdown of PM from dust sources, as compared to all sources of PM in the SJV region is shown in Table 3.3-16. All dust sources contribute a total of 86% of total PM and 41% of PM2.5 in the SJV (CARB, 2013b). Natural background dust sources are not included in the inventory (CARB, 2013b) and are likely difficult to determine, given the large extent of landscape modification undertaken in the SJV.

Most of the dust sources in the SJV region are farming related. Oil and gas operations could contribute to a variety of categories, including: "Construction and demolition - Building and construction dust: Industrial", "Construction and demolition – Road construction dust", "Fugitive windblown dust - Dust from unpaved roads and associated areas", "Unpaved road travel dust – city and county roads" and "Unpaved traffic area - Private". These sources contribute a total of 24% of total PM and 24% of PM2.5. It is unclear in general how important oil and gas sources are in these inventory categories.

The San Joaquin Valley Unified Air Pollution Control District (SJVUAPCD) has developed methods for assessing dust emissions in more detail than other air districts. For unpaved, non-farm roads, a simple methodology is used that computes emissions based on an assumed number of trips per day on each mile of unpaved county and other non-farm road (CARB, 2004a). No specification is made about how the oil and gas industry might contribute to these trip loads (CARB, 2004a). For unpaved private roads, a simple scaling of the above results is performed (CARB, 2004b), assuming one-tenth the travel seen on county and other public unpaved roads. For the "Unpaved traffic area – Private" source category, a method was developed by SJVUAPCD to include sources such as farms, mines, landfills, and oil and gas operations (CARB, 2003). The emphasis in this dataset is on parking lots, working areas, and other cleared land that is driven on. The results of these methods are shown in Table 3.3-17 for the year in which the methodology was developed. As can be seen, the oil and gas industry contributes to a small fraction (0.6%) of the unpaved road dust emissions in the SJV region, with farms dominating emissions again.

While exact quantification is not possible, these results suggest that farming is the major source of anthropogenic dust in the San Joaquin Valley, and that oil and gas is a minor contributor.

Table 3.3-16. Dust emissions contribution to overall PM emissions in the SJV region (CARB, 2013b). Emissions in tonnes per day (1,000kg/d).

	PM	**PM10**	**PM2_5**
Farming operations	155.6	70.7	10.6
Fugitive windblown dust	87.5	40.3	6.9
Paved road dust	68.6	31.4	4.7
Unpaved road dust	57.2	37.6	3.8
Other dust	17.5	8.6	0.9
Total dust	386.4	188.5	26.8
All PM sources	479.8	255.7	68.7

Table 3.3-17. PM10 from dust emissions from unpaved traffic areas (non-road), tonnes per year (1,000kg/y). (CARB, 2003).

	PM 10	%
Farms	2073.0	86.1%
Cotton processing	31.8	1.3%
Landfills	217.9	9.1%
Mining	37.2	1.5%
Oil drilling	14.3	0.6%
Construction	32.7	1.4%

3.3.2.2.8. Natural Sources of Hydrocarbon-Related Air Emissions (Geologic Seeps)

Hydrocarbon species can be emitted to the air from natural sources by surface expressions of hydrocarbon materials (e.g., tar pits) or geologic conduits, fractures, or fissures connecting hydrocarbon-containing reservoirs or sediments to the surface. This possible source is particularly important in the SJV and SC regions, where large oil and gas deposits exist. Because of the co-location of hydrocarbon seeps and oil and gas activities, this has been noted as a potential confounding factor in attributing atmospheric observations to oil and gas activities (e.g., see Wennberg et al., 2012 or Peischl et al., 2013).

The CARB criteria pollutant inventory indicates that petroleum seeps were responsible for 20.0 t per day of ROG out of a total of 6354 t per day across California from all sources (natural and anthropogenic). This can also be compared to 1579 tons per day of ROG from anthropogenic sources. CARB estimates that the vast majority of these ROG emissions are emitted in the South Central Coast Air Basin from offshore seeps in the Santa Barbara region (CARB, 1993).

3.3.2.2.9. Summary of CARB Inventories Treatment of WS Activities

A number of general conclusions can be drawn from the above discussion:

1. Oil and gas activities are responsible for small (generally <5%) fractions of GHG, VOC and PM emissions to California air basins where oil and gas activities are concentrated.

2. Oil and gas activities are responsible for small fractions of NO_x emissions in regions of significant oil and gas activities (<10%)

3. Oil and gas activities are responsible for significant fractions of SO_x emissions in the SJV region (~30%)

4. Oil and gas activities are responsible for significant fractions (30%-70%+) of some stationary source TAC emissions in the SJV (see Figure 3.3-10). Because of large mobile sources and dispersed sources of our indicator TACs, fractions of overall TACs contributions are smaller (see Table 3.3-12).

5. Oil and gas activities appear responsible for large fractions of total hydrogen sulfide and hexane emissions in the SJV region.

6. While dust is a major air quality concern in the SJV region, all evidence points to oil and gas sources being a minor contributor to overall anthropogenic dust emissions and PM from dust.

7. Current inventory methods do not generally allow for clear differentiation of well stimulation from non-well-stimulation oil and gas activities. Better understanding of sources for emissions (e.g., produced hydrocarbon release, processing, other ancillary processes) would allow for further differentiation between stimulation and non-stimulation sources.

8. In some categories (e.g., off-road mobile source), evidence points to some coverage of well-stimulation-related emissions sources, although the exact coverage of well-stimulation-related emissions cannot be determined from the inventory.

9. For some category/pollutant combinations (e.g., crystalline silica dust), coverage of the well-stimulation-associated emissions is unclear or uncertain.

10. Given the relatively small contributions of overall oil and gas activities to most pollutant inventories, treatment of well stimulation emissions is not likely to result in significant changes to larger-scale inventories (e.g., regional or state level).

11. Given the major contribution of overall oil and gas activities to emissions of some stationary TACs in the SJV, application of well stimulation technologies could significantly affect emissions of these TACs, either directly during well stimulation activities or indirectly due to increased production.

3.3.2.3. State-Level Industry Surveys Produced by the California Air Resources Board (CARB)

In 2009, CARB began a detailed survey of oil and gas producer emissions. This survey covers the year 2007, and was released in 2011. As the survey results were further analyzed, corrections were performed and the current version of the results is presented in the revised version of October 2013 (Detwiler, 2013).

3.3.2.3.1. Survey Methods

In early 2009, a survey was mailed to 325 companies representing 1,600 oil and gas facilities. These facilities represented ~97% of the 2007 oil and gas production in California (Detwiler, 2013). The results of the survey were used to compute emissions of CO_2eq. GHGs.

The survey coverage was designed to include all upstream, transport, and refining processes in California. Production and processing activities included all activities required to lift oil and gas to the surface, process it (e.g., acid gas removal), and prepare it for transport to refineries. Oil and gas extraction facilities were included in the survey (Detwiler, 2013, p. 1-3) as were drilling and workover companies (Detwiler, 2013, p. 1-4). In addition, companies that perform ancillary services such as produced water disposal were also included (Detwiler, 2013, p. 1-4). It is not clear if specialized oilfield services involved in well stimulation (e.g., companies that only perform hydraulic fracturing services as subcontractors) were included in the survey coverage.

Related to well stimulation activities, companies were required to report number of active wells, well cellars, and well-maintenance activities (Detwiler, 2013, p. 4-1). By CARB definitions, hydraulic fracturing is considered a well-maintenance activity (Detwiler, 2013, p. 4-1), so results from fracturing should be included in survey results. No discussion of acid fracturing or matrix acidization is explicitly included in the survey, although companies may have reported such activities under “well maintenance” or “well workover.”

Of particular importance is the fact that this survey is significantly more detailed in coverage, scope, and specificity than the above-described inventory methods. The survey consists of 16 tables, with very specific required reporting (e.g., presence of access hatch or pressure relief valve on tank) (Detwiler, 2013, p. A-20). Other examples include nearly 30 types of pumps listed in survey appendices (Detwiler, 2013, p. D-4) and dozens of types of separators included in survey definitions (Detwiler, 2013, p. E-3). The calculation methods for determining emissions rates from reported data are included in a detailed methodological appendix.

3.3.2.3.2. Survey Results

The survey found CO_2eq. GHG emissions from the oil and gas industry to be 17.7 Mt CO_2eq. in 2007. This figure does not include refinery emissions, so results should be equivalent to the above GHG inventory result. The emissions are partitioned by type and gas, as shown below in Table 3.3-18. The emissions are partitioned by industry segment as shown in Table 3.3-19. Emissions by air district are reproduced below in Table 3.3-20.

The survey also reports emissions broken down by type of equipment. The most important emissions sources in the inventory were found to be as follows (all sources greater than

10% of total emissions presented): steam generators (41%), combined heat and power systems (22%) and turbines (17%) (Table 3-6 of Detwiler, 2013). Vented emissions came primarily from automated control devices (31%), compressor blowdowns (17%), natural gas gathering lines (14%), gas sweetening and acid gas removal (13%), well workovers (11%) and gas dehydrators (11%). Fugitive emissions came primarily from compressor seals (42%) and storage tanks (27%).

Table 3.3-18. Results from CARB 2007 oil and gas industry survey. Reproduced from Detwiler (2013), Table 3-1.

Type of source	Gas	Emission (kt/y)	Emission of CO_2 eq.
Combustion	CO_2	16073.4	16398.3
	CH_4	10.8	
	NO_x	0.3	
Venting	CO_2	56.0	392.6
	CH_4	16.0	
	NO_x	0.0	
Fugitive	CO_2	107.3	895.5
	CH_4	37.5	
	NO_x	0.0	

Table 3.3-19. Results from CARB 2007 oil and gas industry survey, presented by business type. Reproduced from Detwiler (2013), Table 3-2.

Type of source	Gas	Emission (kt/y)
Onshore crude production	CO_2eq.	10343.1
Natural gas processing	CO_2eq.	1043.4
Onshore natural gas production	CO_2eq.	547.6
Crude processing and storage	CO_2eq.	407.2
Natural gas storage	CO_2eq.	334.8
Offshore crude production	CO_2eq.	140.1
Crude pipelines	CO_2eq.	89.8
Other	CO_2eq.	4764.7

Table 3.3-20. Results from CARB 2007 oil and gas industry survey, presented by air district. Reproduced from Detwiler (2013), Table 3-4.

Air district	Gas	Emission (kt/y)
San Joaquin Valley	CO_2eq.	14,006
South Coast	CO_2eq.	1,205
Santa Barbara County	CO_2eq.	1,049
Monterey Bay Unified	CO_2eq.	498
Ventura County	CO_2eq.	265
Other	CO_2eq.	636
Total	CO_2eq.	17,659

3.3.2.3.3. Survey Alignment with other California Inventories

The above reported total emissions figure of 17.7 Mt CO_2eq. is slightly higher than the reported inventory value of 17.0 Mt CO_2eq. reported in the 2014 inventory for the year 2007 (CARB, 2014d). A disparity of <5% between these two estimates is consistent with differences that would occur due to methodological differences between the two fundamentally different studies (survey vs. inventory).

In the 2014 GHG inventory, estimated fuel combustion emissions of CO_2, CH_4, and N_2O totaled 15.767 Mt CO_2eq, or 92.7% of the total inventory. In the CARB survey, combustion emissions are responsible for 93% of the total inventory emissions (Detwiler, 2013, p. 3-1). In terms of emissions per gas, the GHG inventory reports 94% of total emissions (combustion + fugitive/venting) as CO_2, with the remainder from CH_4 and N_2O. In contrast, the survey estimates 91.8% of statewide oil and gas emissions as CO_2. Alignment in these sub-results between two different methodologies (e.g., industry survey vs. inventory) increases the confidence in these results.

The yearly GHG inventory is created at the state level, which does not allow for comparison of the emissions by air district reported in Table 3.3-20. Because the GHG inventory does not report emissions by source technology (e.g., steam generator), no comparison between the survey and the inventory at the technology level can be performed. In summary, there is generally very good agreement between the CARB GHG inventory (2014 version) and the CARB oil and gas emissions survey. Such alignment increases the confidence that GHG emissions from the oil and gas industry in California are well understood.

3.3.2.4. Federal Emissions Inventories

While the chief focus of this report is on California information sources, some U.S. federal data are also available on California oil and gas operations.

The U.S. EPA GHG reporting program (U.S. EPA GHGRP) is an annual facility-level-reported inventory for GHG emissions. The requirement for reporting emissions is currently set at 25,000 tonnes per year CO_2eq., and the facilities are separated by industry and state starting in 2008 (U.S. EPA, 2010). This is unlike the national GHG emissions inventory, which is an inventory of GHG emissions from oil and gas production aggregated by activity. The GHG inventory is constructed using standard emissions factors combined with activity counts across the national oil industry. This is compared to the GHGRP, where producers report emissions directly to U.S. EPA.

Within the U.S. EPA GHGRP, the specific section relating to the oil and gas sector is Subpart W, which covers all emissions from the well to the refinery gate. Each operator reports emissions for a facility, which is defined by Subpart W as all emissions associated with wells owned by a single operator, in a producing basin. Reported emissions are computed through a combination of direct measurement, mass balance, and emission factors. Most venting and fugitive emissions are based upon "engineering estimations," which primarily utilize default emissions factors as a function of activity (U.S. EPA, 2010).

The emissions reported the GHGRP are aggregated over twenty categories (see Table 3.3-21). Emissions are reported by species (typically CO_2, N_2O, or CH_4). The data also report specific information about the equipment utilized on site (e.g., whether a vapor recovery system is used for atmospheric storage tanks). For most operators, emissions are reported for only a few of these categories. There is a designation for emissions that occur during well completions from hydraulic fracturing of gas wells. However, since well stimulation in California is utilized primarily for oil production, no well-stimulation-related emissions are reported in the GHGRP for California producing basins.

Table 3.3-21. Emissions reporting categories for GHGRP subpart W, reproduced from the FLIGHT tool (U.S. EPA, 2015b).

Natural gas pneumatic devices	**Associated gas venting and flaring**
Natural gas driven pneumatic pumps	Flare stacks
Acid gas removal units	Centrifugal compressors
Dehydrators	Reciprocating compressors
Well venting for liquids unloading	Other emissions from equipment leaks estimated using emission factors
Gas well completions and workovers	Local distribution companies
Blowdown vent stacks	Enhanced oil recovery injection pump blowdown
Gas from produced oil sent to Atmospheric tanks	Enhanced oil recovery hydrocarbon liquids dissolved CO_2
Transmission tanks	Onshore petroleum and natural gas production and distribution combustion emissions
Well testing venting and flaring	Offshore sources

Due to the 25,000 tonnes per year requirement, emissions from small oil and gas production will likely be undercounted. This is demonstrated in Table 3.3-22, which shows reported GHG emissions for California oil operations separated by basin and source of emissions. Note that these emissions are significantly below those reported by operators in the California oil and gas survey discussed above. For example, compare the results for the San Joaquin production basin from which GHGRP emissions are reported as ~2,050 kt/y CO_2eq., while California survey emissions are reported as ~14,000 kt/y CO_2eq (see Table 3.3-20).

Table 3.3-22. Results from GHGRP subpart W, presented by production basin for California. Reproduced from FLIGHT tool (U.S. EPA, 2015b).

	Reported GHG emissions (kt CO_2eq/y)						
Basin	**Combustion**	**Pneumatics**	**Venting & Flaring**	**Tanks**	**Compressors**	**Other**	**Total**
San Joaquin	1924	18	56	13	18	19	2048
Los Angeles	220	0	2	2	0	0	224
Offshore	235	0	4	0	0	35	274
Other	125	79	5	24	4	40	277

3.3.2.5. Emissions From Silica Mining

One example of an offsite impact of concern is crystalline silica dust emissions during mining and processing of proppant. Crystalline silica dust is a TAC, and can affect human health. Silica dust can be created during the proppant mining, production, and usage phases. Silica is a topic of concern as a WST occupational hazard; for example, see work by Esswein et al. (2013) for exposure assessment in locations outside of California. No oil-and-gas-associated SIC codes in the SJV or SC air districts reported emissions of silica dust. In the SJV air district in total, facility-level total emissions of crystalline silica (CARB species ID 1175) equaled ~749,000 lbs, with a total of 85 facilities reporting from a variety of SIC codes. Similar values are reported for total crystalline silica emissions in the CTI database for 2010 in the SJV region. However, none of the five upstream (or 12 total) SIC codes associated with oil and gas report emission of crystalline silica. In the SJV air district, key SIC codes reporting silica emissions are as follows: construction sand and aggregate (SIC 1442); nonmetallic minerals (SIC 1499); asphalt paving mixtures (SIC 2951); and minerals, ground or treated (SIC 3295). It is not clear from public documents if any of the above facilities are supplying proppant materials for well stimulation activities, and therefore if they may be considered a well-stimulation-related emissions source.

3.3.3. Potentially Impacted California Air Basins

While GHGs have impacts on global climate, the other three well-stimulation-related emissions (VOC/NO_x, TACs and PM) are relatively short-lived and primarily influence the air basins where oil and gas extractions occur. According to the Department of Oil Gas and Geothermal Resources (Figure 3.3-14), the two largest California oil and gas producing regions are contained within the San Joaquin Valley (SJV) Unified air district and South Coast (SC) air district. Specifically, significant oil production occurs in the Kern, Santa Barbara, and Ventura air districts, while non-associated (dry) natural gas production occurs in a number of Northern SJV air districts.

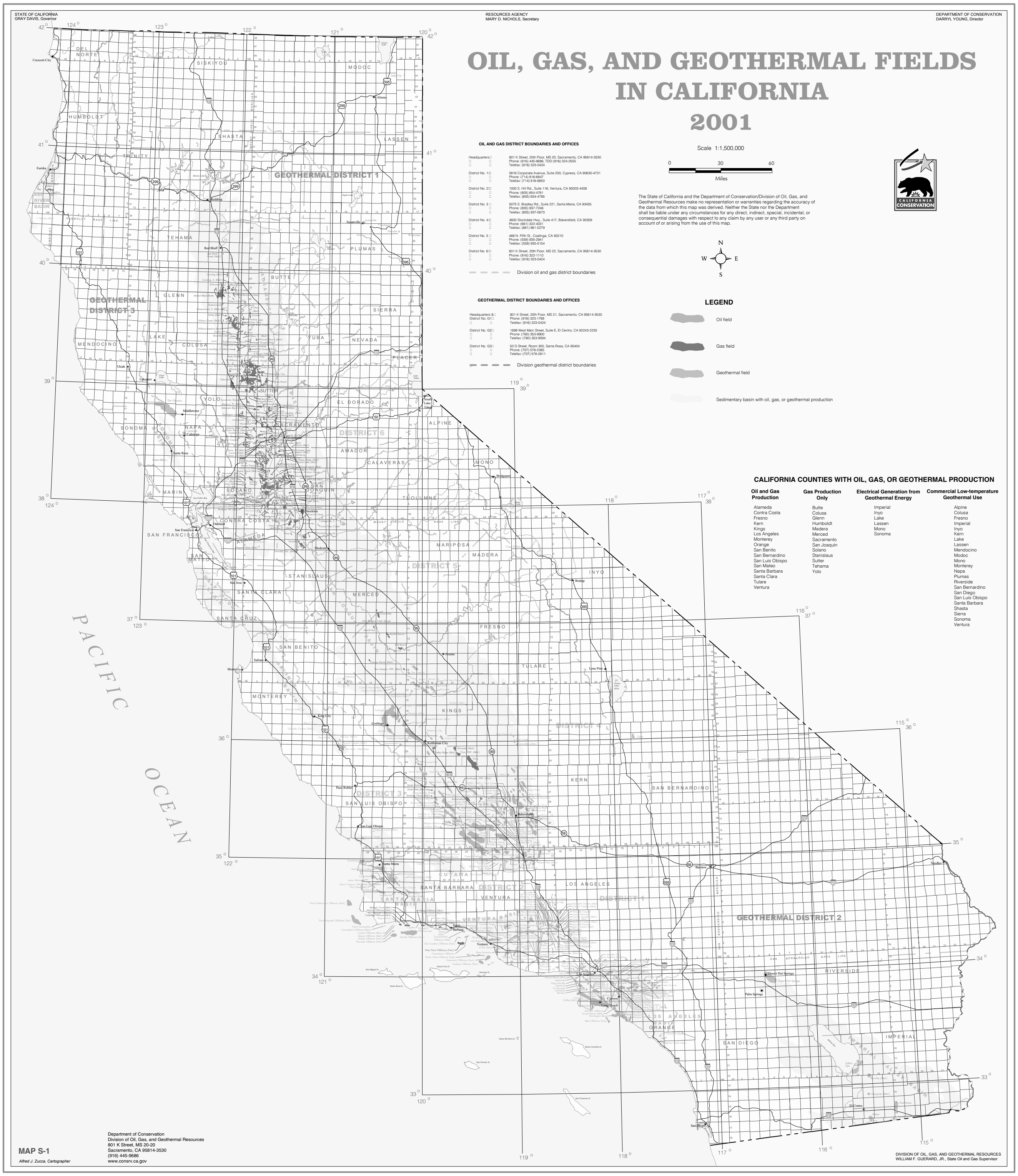

Figure 3.3-14. Map of oil (green) and gas (red) fields in California. Image courtesy of California Department of Oil, Gas and Geothermal Resources [DOGGR] http://www.conservation.ca.gov/dog/Pages/Index.aspx.

3.3.3.1. Status And Compliance in Regions of Concern

The two air basins (San Joaquin Valley and South Coast) most strongly impacted by oil and gas production also coincide with the worst air quality in California. Both air basins are currently out of compliance with both national ozone and PM2.5 standards.

Figure 3.3-15 shows the current designation of ozone attainment status of air districts in California. Ozone pollution level is characterized by a “design value,” a three-year rolling average of the fourth highest 8-hour ozone concentrations measured at the monitoring station. The designation of attainment status is determined by comparing the design value to the National Ambient Air Quality Standard. The nonattainment areas are further divided into six classes from “marginal” to “extreme” depending on the extent to which the design value exceeds the standard. Both San Joaquin Valley and South Coast are classified as an “extreme” nonattainment area, meaning their design values are greater than 175 ppb, which is more than double the current 8-hour ozone standard (75 ppb) and almost three times EPA’s proposed update to the standard (65 ppb). The Sacramento air district, located in the dry gas producing region, is also a “severe” nonattainment area for ozone (Figure 3.3-15).

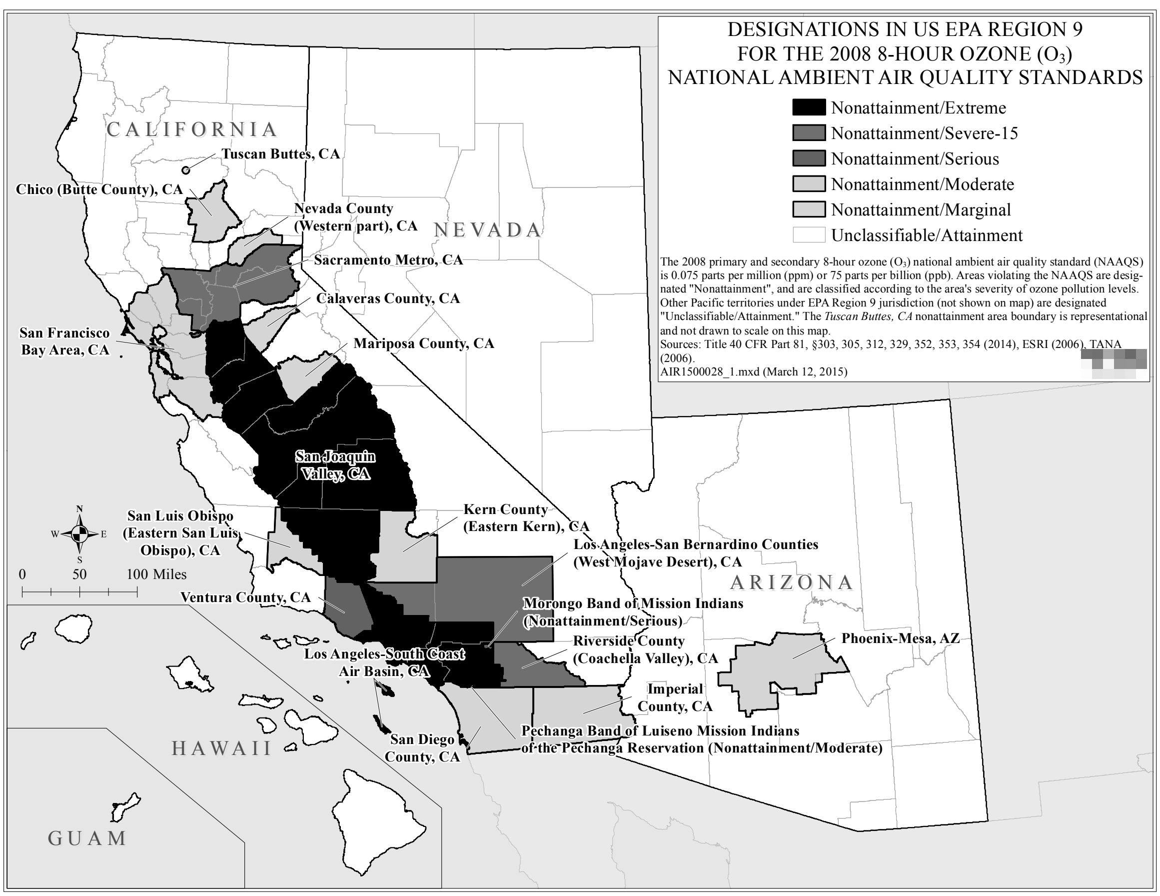

Figure 3.3-15. Air basin designation of 2008 8-hour ozone standard. Source: U.S. EPA, 2012.

The majority of the counties within the SJV and SC air basins are out of compliance with the 24-hour PM2.5 standard (35 mg/m^3), and a few with the newly established annual PM2.5 standard (12 mg/m^3) (Table 3.3-23). Sacramento is a nonattainment area for the 24-hour PM2.5 standard, but not for the annual standard.

A number of factors result in the poor air quality in the SJV and SC regions. First, unfavorable topography creates local circulations trapping emissions in the air basins. Second, besides the oil and gas industry, both regions host diverse emission sources including industry, agriculture, residential homes, businesses, and the transport sector. The SJV is among the nation's largest agriculture areas, with emissions from dairy farms contributing to fine particle formation. Los Angeles County is the most populated county in the U.S. (U.S. Census, 2010) resulting in significant air emissions. Third, the generally warm and sunny conditions in the state promote photochemical reactions and formation of smog in the summer. Cool and humid conditions in winter promote fine particulate formation in the winter.

Table 3.3-23. Attainment status for PM2.5 in the main oil and gas producing regions. (Source: U.S. EPA, 2009 and CARB, 2013f)

Area name	**Counties exceeding 24-h PM2.5 standard (35 ug/m³)**	**Counties exceeding annual PM2.5 standard (12 ug/m³)**
Los Angeles-South Coast Air Basin	Los Angeles Orange Riverside San Bernardino	Mira Loma, Riverside County
San Joaquin Valley	Fresno Kern Kings Madera Merced San Joaquin Stanislaus Tulare	Clovis, Fresno County
Sacramento Air District	El Dorado Placer Sacramento Solano Yolo	

3.3.3.2. Likely Distribution of Impacts Across Air Basins

According to the emission inventory reviewed in previous sections, the well-stimulation-related emissions of VOC/NO_x and PM are likely to be a small fraction of all the emissions occurring in California. More specifically, oil and gas activities are generally responsible for <5% of the emissions of VOC and PM and for <10% of NO_x to the SJV and SC. Oil and gas activities, however, are responsible for significant fractions of some TAC emissions in the SJV with significant oil and gas activities (30–60%). This is particularly the case for VOC-related TACs.

As natural gas produced from a well with natural gas liquids and oil (wet gas) will be richer in VOCs than that from a well producing mostly natural gas (dry gas) (Jackson et al., 2014), the dry-gas producing regions, such as Sacramento Valley, are expected to have smaller contributions to VOCs and TACs.

As noted above, the contribution of oil and emissions in SC is generally much smaller than that in SJV. The population density in the Los Angeles region is also more than 10 times greater than the SJV region, and is largely collocated with oil and gas activities. As a result, the health hazard from oil and gas emissions can still present a significant problem in SC when source proximity and exposed population are taken into consideration (see Volume II, Chapter 6).

3.4. Hazards

3.4.1. Overview of Well-Stimulation-Related Air Hazards

In this chapter, various chemical species related to well stimulation have been grouped into the following four categories.

1. Greenhouse gases (GHGs);

2. Volatile organic compounds (VOCs) and nitrous oxides (NO_x) emissions leading to photochemical smog generation;

3. Toxic air contaminants (TACs);

4. Particulate matter (PM).

In general, GHGs impact global climate and the other three affect air quality.

GHGs: GHGs include carbon dioxide (CO_2), methane (CH_4), carbon monoxide (CO), nitrous oxide (N_2O), VOCs, and black carbon (BC) (IPCC, 2013, pp. 738-740). These species absorb and emit infrared radiation and thus affect the global radiative balance of the atmosphere. GHG emissions considered here generally produce an increase in the average temperature of the Earth.

VOCs and NO_x: VOC emissions are generated during venting of gases from the well and evaporation of chemicals from flow-back or produced liquids. NO_x emissions associated with WS activities will derive primarily from use of engines powered by diesel or natural gas, which are used directly in WS applications. Processing and compression facilities can also contribute to VOCs/NO_x emissions. VOCs/NO_x lead to environmental and health impacts through various pathways.

- NO_x and VOC can enhance formation of ozone, a key constituent of photochemical smog.
    - Ozone is designated as a criteria pollutant by the Federal Clean Air Act (U.S. EPA, 1994) due to its adverse effects on human health (Bell and Dominici, 2008) and on agriculture productivity (e.g., Morgan et al., 2003). Children, elderly, and people with lung diseases such as asthma are particularly sensitive to ozone concentrations.
    - Ozone and its photolysis also affect climate, because ozone is a greenhouse gas and its photolysis products strongly influence the oxidant content of the atmosphere, which, in turn, affects the lifetimes of other important greenhouse gases and TACs.
- The oxidation products of NO_x and VOC can condense into particle phase and lead to an increase in PM burden. PM formed during these processes is called secondary particulate matter, including particulate nitrate and organic carbon. Secondary particulate matter is associated with increased rates of premature mortality through their deep penetration into the lungs.
- In addition to smog-formation potential, VOCs often also function in the short and long-term as GHGs through their eventual decomposition into carbon dioxide (CO_2).
- Some VOCs are carcinogens or endocrine disruptors and are directly hazardous.

TACs: TACs included in a recent EPA risk assessment are listed in Table 3.4-1. These TACs are emitted in similar processes that contribute to VOCs. TACs can be a concern for workers in the oil and gas industry due to their frequent exposure to TACs. TACs may also present a health concern for those who live near oil and gas operations.

PMs: Diesel equipment used to pump the fluid into the well and the diesel trucks used to bring supplies to the well are the major sources of PM emissions directly related to WS activities. Incomplete combustion at flaring units can also produce PM (soot). PM has both environmental and health impacts. Note that PM considered in previous sections are direct emissions, also known as primary PM. Particles formed through complex reactions in the atmosphere (i.e., secondary PM), such as in NO_x and VOCs oxidation products, are not included in the estimation. Formation of secondary organic carbon is likely to be minor (Gentner et al., 2014), because the hydrocarbons emitted from well stimulation are mostly light alkanes whose oxidation products do not tend to condense into particle phase.

- PM is associated with respiratory health impacts and increased rates of premature mortality through its deep penetration into the lungs. Particulate matter with aerodynamic diameter less than 2.5 micron (PM2.5) and 10 micron (PM10) are both regulated by U.S. EPA as criteria pollutants, due to their adverse health effects.

- Fine particles (PM2.5) are the main cause of regional haze. Reduced visibility is a main concern in national parks and wilderness areas.

Table 3.4-1. TAC emissions ranked by mass emissions rate from oil and natural gas production source category, reproduced from U.S. EPA risk assessment. (U.S. EPA, 2011, Table 4.1-1)

TAC	Emissions (tons per year)	Number of facilities reporting (out of 990 facilities)	Included as TACs indicator species in this report?
Carbonyl sulfide	4,151	727	Y
Hexane	1,666	836	Y
Toluene	1,344	940	Y
Benzene	936	963	Y
Xylenes (mixed)	576	924	Y
Ethyl benzene	111	818	Y
Methanol	88	67	
2,2,4-Trimethylpentane	30	733	
Ethylene glycol	27	727	
Naphthalene	17	754	
Chlorobenzene	11	18	
m-Xylene	11	23	
1,1,1-Trichloroethane	9	4	
Glycol ethers			
- Glycol ethers	5	3	
- Ethylene glycol methyl ether	0.1	5	
- Triethylene glycol	0.02	2	
- Ethylene glycol ethyl ether	0.007	4	
p-Dichlorobenzene	4	13	
Formaldehyde	2	255	Y
Biphenyl	2	2	
Cumene	2	23	
Carbon disulfide	1	726	
Diethanolamine	1	4	
o-Xylene	0.8	7	
1,4-Dioxane	0.6	6	
o-Cresol	0.6	1	
Methylene chloride	0.5	1	
p-Xylene	0.3	7	

Phenol	0.2	16	
Acetaldehyde	0.02	245	Y
Polycyclic organic matter			
- PAH, total	0.02	29	
- Benzo[a]Pyrene	0.002	3	
- Chrysene	0.002	3	
- Anthracene	0.000006	1	
- Benz[a]Anthracene	0.000005	2	
- Benzo[b]Fluoranthene	0.000003	2	
- Benzo[k]Fluoranthene	0.000003	2	
- Dibenzo[a,h]Anthracene	0.000003	2	
- Indeno[1,2,3-c,d]Pyrene	0.000003	2	
Propylene oxide	0.009	4	
Cresols (mixed)	0.009	6	
Ethylene dichloride	0.007	9	
Chloroform	0.004	1	
Hydrochloric acid	0.004	4	
Acrolein	0.002	8	
Dibenzofuran	0.001	1	
Ethylene dibromide	0.0006	5	
Styrene	0.0003	4	
Vinylidene chloride	0.0003	1	
Ethylene oxide	0.00002	1	
Acrylamide	0.00002	1	
Methyl bromide	0.0000005	1	

3.4.2. Hazards due to Direct vs. Indirect Well Stimulation Impacts

Due to their different transport, transformation, and removal mechanisms, species emitted from well stimulation have atmospheric lifetimes ranging from hours (e.g., TACs) to more than 100 years (e.g., CO_2). Accordingly, their spatial impacts range from local, regional, to global scales. GHGs have global impacts over long time scales. As a result, for any given amount of GHG emissions, their impact is not tied to the source locations and emitted time. For the other three categories (NO_x/VOC, TACs, and PM), with generally shorter atmospheric lifetimes and local to regional dispersion, the impacts are closely related to their temporal and spatial allocation at different phases of the well stimulation life-cycle. Some TAC species, such as TAC metals species, can be persistent in the environment and may become more widely dispersed than the reactive organic TACs.

3.4.2.1. Direct Well Stimulation Impacts

On-site air hazards of NO_x/VOC, TACs, and PM affect the surroundings of the well site and downwind areas. Emissions from various phases of on-site activities generally affect the concurring time periods plus a few weeks after, as their atmospheric lifetimes are not longer than a few weeks. The phases of on-site activities and their occurring time frame are as follows:

- Well stimulation application and well completion last days to weeks for a single well and up to a couple of months for multiple wells. During these processes, NO_x and PM are emitted from the on-site diesel engines for trucking and pumping. Hydrocarbons, including smog-forming VOCs and TACs, are the major components from fugitive and vented emissions from well stimulation materials, pipe lines, and flowback water.

- On-site handling of proppant materials could pose a health risk to workers due to particulate matter emissions associated with silica proppant sands.

3.4.2.2. Indirect Well Stimulation Impacts

Some indirect WS hazards are described in this report:

- Well-stimulation-induced production impacts are impacts associated with hydrocarbon production that would not have been economically viable without application of well stimulation technology. The production of additional hydrocarbons on-site may last years after the well completion. Production and processing activities such as dehydration and separation can produce VOC and TAC emissions from equipment leaks, intentional venting from produced water storage tanks, and flaring. PM and NO_x can also be produced by incomplete combustion during flaring and use of diesel engines and compressor engines.

- Supply-chain impacts associated with well stimulation activity include air emissions generated during the course of numerous industrial activities associated with the preparation, distribution, and maintenance of well-stimulation-related materials.

In summary, the well-stimulation-induced hydrocarbon production produces continued air hazards affecting the well site and downwind areas. The supply chain impacts and macroeconomic impacts are more distributed in time and space.

3.4.3. Assessment of Air Hazard

3.4.3.1. Greenhouse Gas Hazard Assessment

GHGs affect global radiative balance. Due to their relatively long atmospheric lifetimes, emissions are well mixed globally once they enter atmospheric circulation. As a result, their impact can be well represented by the mass emitted. As described by the IPCC (2013), the hazard is usually assessed by global-warming potential (GWP), a relative measure of how much heat is trapped by a GHG relative to the amount of heat trapped by the same mass of carbon dioxide. The GHG impacts, listed using current 20-year and 100-year global warming potentials for well-stimulation-relevant gases, are presented in Table 3.2-1. When the GWP of a GHG is applied as a multiplier to its emissions, a CO_2 equivalency amount is derived. Essentially, CO_2 equivalency describes the amount of CO_2 that would have the same GWP as the given amount of emissions of another GHG over a specified time scale (e.g., 100 years). Using the GWP, emissions of a mixture of GHGs can be expressed by a single quantity of CO_2 equivalency to represent the time-integrated (100 years) value of radiative forcing of the mixture.

3.4.3.2. Air Quality Hazard Assessment

3.4.3.2.1. Overview of Air Quality Assessment Methods

NO_x/VOCs, TACs and PM emissions have shorter-term effects than GHGs, and the spatial impacts are not homogeneous due to their shorter atmospheric lifetimes. The air quality of a region is characterized by measurements of ambient concentrations of specific pollutants, including PM and ozone, from central monitors in that region. Before the pollutants emitted from well stimulation are measured by the monitoring devices, they are dispersed by wind and may undergo chemical transformation in the atmosphere. The manner in which the same emissions will affect air quality will differ, depending on the meteorological conditions and the other pollution already present in the atmosphere (the chemical transformations depend on total pollution levels). Although oil and gas activities have relatively low contributions to criteria air pollutant emissions (NO_x, VOCs, PM), as summarized in previous sections, they do in some cases produce relatively high contributions to TACs. Atmospheric dispersion of TACs needs to be tracked with models in order to determine their impacts on populations at varying distances downwind.

There are several methods one might employ to evaluate how well stimulation emissions impact air quality. One could try to determine the impact of emissions through analyzing air quality measurements, comparing air quality on days with high well stimulation activity to days with low well stimulation activity. However, the variability in meteorology and atmospheric chemistry between days would likely overwhelm any signal that might exist from well stimulation variability. Instead of depending only on measurements, air quality models are often used to describe how pollutants are dispersed through the atmosphere and chemically transformed. The models connect the pollutant emissions to their air quality impacts. Two different air quality models are discussed below; their suitability for application depends on the nature of the pollutant of interest.

Gaussian plume dispersion modeling is a simple yet powerful tool to calculate the evolution of air pollutant concentrations during the course of wind-driven transport and dispersion of non-reactive or first-order decaying pollutants, with decay rate linearly related to concentration. Gaussian plume models (see review by Holmes and Morawska, 2006) are based on analytical solutions to the advection-diffusion equation in simplified atmospheric conditions. Gaussian dispersion models can handle complex terrain and can be adapted to account for some atmospheric processes such as deposition. Gaussian plume models cannot account for interactions between plumes; they are not able to track nonlinear chemistry that leads to secondary pollutant formation in the atmosphere, such as ozone formation. They require relatively little data and computational resources.

A more useful method for calculating ambient pollution levels is to use chemical transport models (CTMs). CTMs solve the advection-diffusion equation numerically for a reactive flow on a gridded domain. CTMs implement chemical mechanisms containing hundreds of reactions. They can also include time-resolved representations of nonlinear chemistry and particle dynamics with various degrees of complexity. CTMs require very detailed meteorological forcing inputs, such as wind velocity, temperature, humidity, etc., at each grid cell of the domain, are computationally expensive, and require advanced training. The advantages of using CTMs to estimate exposure include the capacity to account for the effects of space- and time-resolved influential parameters (e.g., detailed wind and temperature fields) and the capacity to model nonlinear processes such as second-order chemistry and particle dynamics in a time-resolved manner. This approach is suited for simulating concentrations of secondary pollutants such as ozone and secondary organic carbon.

Air quality hazards discussed here include species of emissions associated with well stimulation (directly emitted species) and the pollutants formed through chemical transformation of these emissions in the atmosphere (chemically formed species). Suitability of air quality models for assessing these two types of pollutants is discussed further in the following sections.

3.4.3.2.2. Well-Stimulation-Induced Air-Quality Hazard Assessment: Directly Emitted Species

Directly emitted species can be tracked with Gaussian plume dispersion models, which link the amount of emissions from the source locations to changes in concentrations. These species include all the air hazards considered in previous emission inventories (e.g., NO_x/VOCs, TACs, and primary PM). These can be done for on-site emissions of selected case studies, where a clear emission boundary can be defined. Modeling and analysis protocol is briefly described below.

- Required inputs:
    - Meteorological data (obtained from national weather service): hourly or daily wind speed and direction, amount of atmospheric turbulence, ambient air temperature, inversion height, cloud cover and solar radiation
    - Emission parameters: source location and height, spatial characteristic of source as in point (i.e., smoke stack), line (i.e., highway), or area (i.e., oil field), and exit velocity and mass flow rate of the plume
    - Terrain elevations and surface characteristics: ground elevations at the source and at the locations where pollutant level are to be computed, surface roughness
- Model simulation: use the meteorological data and surface characteristics to drive the dispersion model for emitted pollutant of interest, accounting for depositional loss and first-order decay.
- Post-model analysis: enhancement in ambient concentrations of the pollutant of interest can be plotted as a function of time and space and summarized by season. The most impacted times and locations can be identified and used for subsequent exposure and health studies.

3.4.3.2.3. Well-Stimulation-Induced Air-Quality Hazard Assessment: Chemically Formed Species

Chemically formed species such as ozone and secondary PM are not directly emitted and thus cannot be tracked by dispersion models, as there is no discrete "source location." Another challenge is that the formation chemistry of these pollutants is often nonlinear. In other words, the amount of pollutant formed cannot be linearly scaled from its precursor emission quantities, but rather depends on the pollution levels present in the air. For example, in a NO_x-rich environment such as a densely populated Los Angeles urban area, additional NO_x emissions from well-stimulation-related activities may actually decrease ambient ozone concentration locally, while affecting downwind regions (Rasmussen et al., 2013). In NO_x-poor areas, such as a remote well pad location in the San Joaquin Valley, the opposite is true, i.e., well-stimulation-related NO_x emissions contribute to an increase in ambient ozone levels (Rasmussen et al., 2013). Chemical transport models (CTMs) are required in this case to simulate the formation process of these species from their precursors. In the case of ozone, CTMs track the production and removal of ozone as its NO_x/VOC precursors disperse from the source location downwind accounting for the nonlinear chemistry. Conducting computer simulation with CTMs is beyond the scope of this study. Many past studies have investigated ozone and secondary PM responses to changes in emissions (Jin et al., 2008; 2013; Rasmussen et al., 2013; Chen et al., 2014) in California.

3.5. Alternative Practices to Mitigate Air Emissions

This section presents a review of alternative practices that reduce emissions of pollutant and GHG related to well stimulation with a focus on direct hazards.

3.5.1. Regulatory Efforts to Prescribe Best Practices

Many states outside of California where hydraulic fracturing takes place have begun to regulate the overall environmental impacts of the oil and gas industry, by requiring emission controls and best practices. As reviewed in Moore et al. (2014), Colorado, Wyoming, Montana, and New York have taken the most aggressive regulatory steps to reduce both pollutant and GHG emissions (Table 3.5-1). In the regulations passed from 2007 to 2009 in Colorado, operators are required to apply alternative practices and controls to reduce VOC emissions, including use of "green completion" or reduced emission completion technologies at oil and gas wells when technically feasible, and control evaporative emissions from condensate and oil storage tanks. In the northeastern Front Range O3 nonattainment area, further actions are required, such as use of no-bleed or low-bleed pneumatic devices.

In California, regulations are set at local air district levels. San Joaquin Valley Air Pollution Control District Rule 4402 regulates the emissions of VOCs from crude oil wastewater sumps. Under this rule, VOC emission control, such as a covering in place, is required for any produced water containing over 35 mg/L of VOCs. Small oil producers and "clean produced water" containing less than 35 mg/L are exempt from the rule. The South Coast Air Quality Management District (SCAQMD), have more stringent regulations for open pits. The SCAQMD (Rule 1176) for example, requires produced water in open pits contain less than 5 mg/L VOC's, compared to the San Joaquin threshold of 35 mg/L.

At the national level, in 2012, the U.S. EPA released a set of new source performance standards (U.S. EPA, 2012) which were phased in starting in late 2012, with full effect in early 2015. The standard requires the use of green completion technologies and reduced VOC emissions from temporary storage tanks during well completion. Historically, the fluids and gases in flowback water are routed to an open-air pit or tank to allow evaporation. Green completion captures liquids and gases during well completion with temporary processing equipment for productive use.

Further VOC and TACs controls are required by the rule (U.S. EPA, 2012), including limiting emissions of VOCs from a new single oil or condensate tank to four tons per year, and limiting the hazardous air pollutants benzene, toluene, ethylbenzene, and xylenes (BTEX) from a single dehydrator to one ton per year. In addition to VOC reduction through vapor controls at temporary storage tanks, green completion also benefits the control of methane emissions by essentially requiring natural gas companies to capture the liquid and gas at the wellhead immediately after well completion instead of releasing it into the atmosphere or flaring it off.

The U.S. EPA also adopted multiple tiers of emissions standards for new diesel engines that may influence emissions incurred by the trucking and pumping processes related to well stimulation. Vehicle and engine pollutant emissions, including NO_x, non-methane hydrocarbons, CO, and PM can be largely controlled if new engines, vehicles, or equipment is used that meet the latest emission tier. However, the long useful life of diesel equipment and vehicles has prompted California to add additional emission requirements for in-use on-road diesel trucks and other diesel equipment. For example, California requires that almost all heavy-duty trucks and buses that currently operate in the state meet stringent particulate matter emission standards now, and meet stringent NO_x emission standards within the next decade.

3.5.2. Control Technologies and Reductions

Many emissions from the above three key processes can be addressed with best controls and alternative practices. U.S. EPA (2012) estimates implementation of green completion will result in a 95% reduction of VOC emissions and a 99.9% reduction in SO2 emissions. These green completions technologies are evolving over time due to relative novelty, and will likely improve with additional deployment (e.g., cost could be reduced). Allen et al. (2013) reported low leakage rates from well completions after some of the controls listed above were implemented compared to uncontrolled processes, and ICF International (2014) analyzed the costs and viability of methane reduction opportunities in the U.S. oil and natural gas industries. Harvey et al. (2012) reviewed 10 technologies with the capability estimated to reduce more than 80 percent of methane emissions in the oil and gas sector. In addition to methane emissions, many of the technologies have the co-benefit of reducing explosive vapors, hazardous air pollutants, and VOCs.

Large reductions in pollutants and GHGs through use of new technologies and compliance with regulations should be interpreted as best-case scenarios and should not be used to estimate real-operation efficacy. For example, current requirement in VOC reductions from tanks in Colorado are 90% in the summertime and 70% in other times of year of the actual annual average reduction in emissions. However, actual reductions were estimated to be 53% (State Review of Oil & Natural Gas Environmental Regulations, 2011). More importantly, in fast-developing areas, increasing numbers of new wells may counter the overall pollution-control benefits resulting from emission controls applied to individual wells. Despite tightening of emission standards for the oil and gas industry in Colorado, the oil and gas-related VOC measurements made in the non-attainment area in Erie showed a continued increase (Thompson et al., 2014). System-wide emission reduction needs careful planning and monitoring, accounting for both technology advances and industry development and expansion.

Table 3.5-1 summarizes the control technologies and alternative practices available in the literature according to their related processes, as reviewed in previous sections. The national- and state-level adoptions of the various practices are also noted. Depending on their attainment status, local air districts may have more stringent regulations of air emissions than the state level, such as permitting programs for new sources. For example, while emission data from the oil and gas industry are collected in Texas, regulation of emissions is limited to the Houston and Dallas—Fort Worth federal ozone standard non-attainment areas. These local regulations can be important, but are not included in the table.

Table 3.5-1. Best control or practices for controlling emissions from key processes.

Process	**Best Control or Practice**	**Description**	**Emissions addressed**	**Regulation adoption**
Trucking and pumping supplies/fluid to the well	U.S. EPA tier 4 diesel engines	Installed with control technologies to reduce emissions from diesel equipment by 90% compared to the one from 1990s.	NO_x, PM	
	Use of newest truck built since 2010	Included exhaust controls	NO_x, PM	
Venting and flaring	Green completion	Capture liquids and gases coming out of the well during completion.	CH_4, VOCs, and TACs	U.S. EPA, Colorado, Wyoming, Montana.
	Plunger lift system	Collect liquids inside the wellbore and capture methane.	CH_4, VOCs, and TACs	
	Dehydrator emission controls	Capture methane with emission control equipment placed on dehydrators	CH_4, VOCs, and TACs	Montana
	Methane capture during pipeline maintenance and repair	Re-route or burn methane, use of hot tap connections, de-pressuring the pipeline etc.	CH_4, VOCs, and TACs	
	Low-bleed or no-bleed pneumatic controllers	Reduce methane release to the atmosphere, or move away from gas-operated devices.	CH_4, VOCs, and TACs	Colorado
Fugitive and/or evaporation of gas and chemicals	Dry seal systems and improved compressor maintenance	Reduce emissions from centrifugal compressors and reciprocating compressors	CH_4, VOCs, and TACs	Montana
	Tank vapor recovery units	Capture gases released from flashing losses, working losses, and standing losses	CH_4, VOCs, TACs.	Colorado, Montana.
	Leak monitoring and repair	Monitoring potential leaks at equipment locations subject to high pressure.	CH_4, VOCs, TACs.	

3.6. Data Gaps

A number of data gaps exist in understanding emissions from well stimulation activities. The challenges that exist include:

- Few studies exist that directly measure emissions from oil and gas activities;

- Even fewer studies exist that directly examine well stimulation activities, none of which occurred in California; and

- It is unclear how applicable results from a given study conducted elsewhere might be to California well stimulation activities, due to significant differences in treatment and regulation of both air emissions and well stimulation between states.

These challenges noted, the available studies that were deemed most relevant to understanding air impacts of well stimulation are reviewed below. These studies can be broken down into studies that will directly measure or assess emissions at a facility or device level (henceforth "bottom-up" studies) and studies that perform indirect or remote measurement of gas concentrations and then estimate emissions from these measurements (henceforth "top-down").

3.6.1. Bottom-Up Studies and Detailed Inventories

A number of experimental studies or bottom-up inventories were performed in regions with significant well stimulation activities. These studies include:

- Study of direct emissions from well stimulation by Allen et al. (2013).

- Study of direct emissions from hydraulically fractured natural gas wells by ERG (Eastern Research Group) and Sage Environmental.

- Study of emissions in the Eagle Ford hydraulically fractured oil basin.

- The Barnett area special inventory.

3.6.1.1. Allen et al. (2013) Study of Hydraulic Fracturing Processes

3.6.1.1.2. Overview of Study and Goals

Aside from the Fort Worth study, the most significant scientific assessment examining the GHG impacts of well stimulation was conducted by Allen et al., funded by the Environmental Defense Fund and with the cooperation of operators (Allen et al., 2013).

3.6.1.1.2. Methodology

This study calculated methane emissions and emissions factors at 190 natural gas facilities across four regions of the country where well stimulation was utilized (Appalachian, Gulf Coast, Mid Continent, and Rocky Mountain). Of these natural gas facilities, they examined 150 production facilities with 489 wells, along with 27 well completion flowbacks, nine well unloadings, and four well workovers across nine different operators.

In order to capture emissions from flowback, completion, unloading, and workover operations, they bagged and diverted all hatches to temporary stacks, where the emissions were analyzed and fluxes calculated. For the production facilities, they utilized an IR camera and recorded leaks for equipment that were detected by the camera. If leaks were detected, they utilized a Hi Flow sampler to measure emissions rates. All of these leaks were reported under the category "Equipment Leaks." (Included in this category are valves, connectors, and well equipment.) In addition, Allen et al. reported detailed results for pneumatic devices, all of which were analyzed with a high-flow sampler.

3.6.1.1.3. Key Findings

The most significant finding was that, overall, methane emissions were found to be slightly lower than the 2011 U.S. EPA inventories. This was the result of measured methane emissions from completion flowbacks that were an order of magnitude lower than the U.S. EPA inventory, offset by higher emissions rates for chemical pumps, pneumatic controllers, and equipment. The overall emissions estimates report a methane leakage rate of 0.42% compared to the U.S. EPA value of 0.47%.

3.6.1.2. City of Fort Worth Air Quality Study

3.6.1.2.1. Overview of Study and Goals

A comprehensive study of direct measurements of emissions from natural gas production in a region of hydraulic fracturing is the "City of Fort Worth Natural Gas Air Quality Study." This was commissioned in 2010 by the city of Forth Worth, TX, and prepared by Eastern Research Group along with Sage Environmental Consulting, LP (ERG/SAGE 2011). The goals of the study were to quantify the environmental and public health and safety impacts of hydrocarbon production activities. They measured leaks from 388 sites, which included 375 well pads with 1,138 wells. The results of this study were published in a report as well as spreadsheets that detail component-level emissions for each site.

3.6.1.2.2. Study Methods

At each of the well locations, leaks were recorded with the following methodology. Initially a FLIR infrared camera was utilized to detect large leaks. The emissions flux for these large emitters was measured with a Hi Flow Sampler. In addition to recording these large leaks, 10% of all valves and connectors were recorded with a toxic vapor analyzer, and any leak greater than 500 ppmv was recorded and measured with a Hi Flow Sampler. Additionally, Summa Canisters were utilized to provide gas speciation.

Leaks were placed into three broad categories: "valves," "connectors," or "other." Leaks were also classified by study authors using detailed categorization with 94 designations. Neither of these categorization schemes align well with U.S. EPA or other established methodologies, making construction of emissions factors difficult from this dataset.

3.6.1.2.3. Study Findings

As is typical for analysis of gas leakage, emissions are driven by a small percentage of leaks. In this case, 6% of wells account for half of the total measured emissions in the study on a well basis, and the average emissions rate ($\sim 1 \times 10^4$ kg/year) aligns with the 75th percentile.

While this study represents a large group of measurements on a significant number of wells that were hydraulically fractured, it is not clear how applicable the observed emissions rates are to California. Also, since the Barnett shale studied in the report is a dry gas region, data would be most applicable to analogous types of environments, such as gas production in the northern SJV region.

3.6.1.3. Alamo Area Council of Governments Eagle Ford Emissions Inventory

3.6.1.3.1. Overview of Study

The Alamo Area Council of Governments (AACOG) conducted an oil and gas emissions inventory for criteria air pollutants. Though some work is still in progress, the bulk of the results were released as a technical report in 2013 (AACOG, 2014). The purpose of the report was to quantify criteria air pollutants (CO, NO_x, and VOCs in particular) from oil and gas drilling, completion, production, and processing (midstream) operations in the Eagle Ford shale formation. Due to regulatory constraints, they did not conduct any measurements of GHG emissions.

3.6.1.3.2. Methods

As part of the study, the authors developed detailed activity counts of drilling rigs, compressors, compressor stations, equipment at production facilities, as well as timelines for production activities (such as drilling and completions). Emissions were calculated from these activity counts with existing emissions factors from the literature or from the Texas Commission on Environmental Quality. Specific emissions factors were calculated for compressor stations as well as drilling rigs, while activity counts and emissions factors for production facilities were aggregated at the county level. They then utilized these aggregated data as inputs to an air-quality impact model, and also provided an uncertainty analysis discussing potential future scenarios of well stimulation air quality impacts in the Eagle Ford.

3.6.1.4. Barnett Shale Special Inventory

3.6.1.4.1. Overview of Study

In response to observing VOC leakage from surface equipment coinciding with the growth of gas production in the Barnett shale, the Texas Commission on Environmental Quality (TCEQ) conducted an emissions inventory of upstream and midstream sources in the twenty-three counties that overlie the Barnett shale. The study was conducted over two phases between 2009 and 2011 and presented county-aggregated emissions factors and activity counts covering the 2009 production year. The pollutants reported in the publicly available summary are NO_x, VOCs, and HAPs (though more detailed information can be requested and are included in the internal TCEQ database).

3.6.1.4.2. Methods

TCEQ collected these data through operator self-reporting. The first phase of the project, which covered activity counts, acquired data from 9,123 upstream and 519 midstream facilities. Results were generated for twenty-two different equipment categories. Produced water storage tanks and piping components are the largest sources of activity, with over 15,000 tanks and 12,500 piping component fugitive areas in the sample data (TCEQ, 2010). Part one of the special inventory included activity counts of higher emissions equipment (TCEQ, 2010).

The second phase of the project was conducted over 2010–2011 and accounted for emissions estimates for sites and equipment at 8,500 sites (TEQ, 2011). The emissions rates for NO_x, VOCs, and HAPs were computed either through taking site-specific samples or through the utilization of TCEQ emissions factors which were provided in the surveys. This allowed for emissions rates as categorized by equipment type across the Barnett region (tons per year). More detailed speciation and some site-level emissions rates can be obtained through contacting TCEQ, but this was determined to be beyond the scope of this work. Maps of results from the TCEQ Barnett inventory are available (TCEQ, 2014).

3.6.2. Top-Down Studies and Experimental Verification of California Air Emissions Inventories

Understanding the accuracy of emissions inventories is an important factor in understanding the impact of well stimulation on air quality in California. If experimental evidence suggests that inventories of the air pollutants of concern (GHGs, VOC/NO_x, TACs, PM) are inaccurate, then this could point to the need for improved understanding of poorly understood or novel contributors to air emissions, such as well stimulation.

Using observations to determine the accuracy of inventories is difficult, and such experimental studies tend to be expensive and performed in a sparse set of locations and time periods. Thankfully, California air quality is the topic of a significant number of experimental studies, over many decades. For this reason, observations that allow assessment accuracy of inventories are numerous in California compared to other regions. A prime example of such activities is the recent large CalNex effort, funded by CARB and NOAA (National Oceanic and Atmospheric Administration), to examine a number of scientific questions at the interface of climate and air quality. CalNex resulted in the publication or submission of approximately 100 peer reviewed scientific papers over a four-year period, with flights and samples occurring in 2010 (Ryerson et al., 2013). A key scientific goal of CalNex was the assessment of CARB inventory accuracy. For more information, the CalNex campaign is introduced in Ryerson et al. (2013) and summary results to scientific questions are presented in a synthesis report (Parrish, 2014).

Using observations to check the accuracy of CO_2 inventories is difficult (Ryerson et al., 2013). This is because CO_2 sources are ubiquitous, and natural diurnal variation in sources and sinks of CO_2 makes discerning a signal challenging. Noting these challenges, CO_2 observations from CalNex agree within experimental error with scaled inventory results (Parrish, 2014, Finding F1). Similar accuracy was found for the CARB CO inventory (Parrish, 2014, Finding F5). NO_x emissions were also found to be in general agreement with CARB inventories, with some caveats about spatial distributions of emissions (Parrish, 2014, Finding F6). No studies of PM or TACs with specific implications for oil and gas or well-stimulation-related emissions were found.

Notably, in CalNex, significant divergence or error was found in comparing CH_4 and VOC observations to inventories. Importantly, in each of these cases, oil and gas sources were examined as specific possible contributors to excess emissions.

Numerous studies, including some before CalNex, examine CH_4 concentrations in California. Some of these studies make explicit comparison to CH_4 inventories, with some specifically examining the role of oil and gas sources in California CH_4 emissions.

CH_4 relevant studies reviewed below are:

- Wunch et al. (2009): Emissions of greenhouse gases from a North American megacity
- Zhao et al. (2009): Atmospheric inverse estimate of methane emissions from Central California
- Hsu et al. (2010): Methane emissions inventory verification in Southern California
- Wennberg et al. (2012): On the sources of methane to the Los Angeles atmosphere
- Peischl et al. (2013): Quantifying sources of methane using light alkanes in the Los Angeles basin, California
- Jeong et al. (2013): A multitower measurement network estimate of California's methane emissions
- Jeong et al. (2014): Spatially explicit methane emissions from petroleum production and the natural gas system in California
- Johnson et al. (2014): Analyzing source apportioned methane in northern California during Discover-AQ-CA using airborne measurements and model simulations

VOC-relevant studies include:

- Gentner et al. (2014): Emissions of organic carbon and methane from petroleum and dairy operations in California's San Joaquin Valley

These studies are reviewed below including their methods and their key results as related to oil and gas CH_4 sources in California. Findings are summarized to determine if there is a consensus regarding the accuracy of California inventories, and whether well-stimulation-associated emissions could be responsible for inventory discrepancy.

Note that top-down atmospheric studies typically report emissions in Tg of CH_4. At typical upstream (production) compositions, 1 Tg of CH_4 (or 52.2 BCF of CH_4) is equal to about 60 BCF (billon cubic feet) of produced natural gas.

3.6.2.1. Wunch et al. (2009)

Wunch et al. (2009) analyzed air column concentrations of CH_4 and CO_2 in the atmosphere of the south coast air basin (SoCAB) in 2007–2008 (Wunch et al., 2009). Fourier transform spectroscopy of sunlight was performed for 131 days of observations. This study cannot reliably partition CH_4 emissions into oil and gas and non-oil and gas sources, due to lack of isotopic sampling and lack of observations of higher alkanes which may provide a chemical "fingerprint" of an oil-and-gas-associated source of CH_4.

Wunch et al. estimate CH_4 emissions in the SoCAB region of 0.6 (+/-0.1) or 0.4 (+/-0.1) Tg CH_4 per year, depending on whether the CARB CO_2 inventory or CARB CO inventory is used to provide temporal scaling of emissions to relate atmospheric concentrations to emissions rates. They compare this result to the CARB CH_4 inventory as follows: CH_4 from all "urban sources" (non-forestry, non-agriculture sources of CH_4) is scaled to the region using the fraction of California population in the SoCAB region. Thus, they argue that the CARB CH_4 inventory underpredicts CH_4 emissions.

3.6.2.2. Zhao et al. (2009)

Zhao et al. (2009) utilized data from a tall tower in the northern SJV, with measurements taken at ~90 m and ~480 m heights. Observations were performed from October to December 2007. A high precision (0.3 ppbv) cavity ring-down spectrometer was used to measure CH_4 concentrations at five-minute intervals. These observations were coupled to an atmospheric transport model. The model was used in an inverse approach to estimate, for a given gas concentration observation, where the gases observed are likely to have been emitted (parcels of air are modeled backward in time for five days). The coupling of tower observations of gas concentrations with simulation allowed an estimate of the likely emissions rates in a spatially resolved manner. They compared their emissions estimates to the Emissions Database for Global Atmospheric Research (EDGAR) spatially resolved emissions inventory (EDGAR v. 3.2).

Zhao et al. find that for the region (central California) and time period (October–December 2007) of analysis, actual emissions are estimated to be 37% +/- 21% higher than annually averaged inventory estimates from the EDGAR inventory. In particular, they believe that livestock emissions are underestimated by an even larger fraction. They do not compare their results to the CARB inventory, as at the time of their study, there existed no spatially resolved version of the CARB inventory (see below for more discussion).

3.6.2.3. Hsu et al. (2010)

Hsu et al. (2010) performed analysis of captured flasks of air from a remote location (Mt. Wilson observatory) to estimate the concentrations of gases in a well-mixed sampling of air from the Los Angeles region. Their study is the Los Angeles County portion of the SoCAB region. In order to estimate CH_4 flux from CH_4 concentrations, they used observed ratios of CH_4 to CO in the atmospheric observations, and coupled this ratio to the CARB CO inventory to estimate CH_4 emissions rate.

Hsu et al. estimated methane emissions of 4.2 +/- 0.12 Mt CO_2 eq. GHGs per year. They then compared this to the CARB inventory of the time, which estimated CH_4 emissions of ~3 Mt CO_2 eq./y from the study region. Thus, they argued that the CARB CH_4 inventory underpredicts CH_4 emissions from the study region.

3.6.2.4. Wennberg et al. (2012)

Wennberg et al. (2012) combined observations of a variety of types to estimate emissions of methane in the SoCAB region. They included air flasks from remote observation locations, aircraft observations from a set of flight campaigns, as well as ground-based Fourier transform spectroscopy to estimate air-column concentrations of CO_2, CO, and CH_4. This study can be seen as an extension and improvement of the work of Wunch et al. (2009) and Hsu et al. (2010). In a novel advance from those previous studies, Wennberg et al. used C_2H_6 concentrations to attempt to partition emissions into various sources.

Wennberg et al. estimated CH_4 emissions in the study region to be 0.44 +/- 0.15 Tg CH_4/y. They compared this to an inventory based largely on CARB sources, which has a scaled emissions estimate for the study region of 0.21 Tg CH_4/y. Thus, they argued that CH_4 emissions may be approximately two times larger than an inventory approach would produce in the region.

3.6.2.5. Peischl et al. (2013)

Peischl et al. (2013) use a variety of sampling methods with aircraft data to estimate CH_4 emissions in the SoCAB region. Similar to other studies noted above, they used CO concentrations and the CO inventory to estimate CH_4 fluxes from CH_4 concentrations.

Peischl et al. created estimates for all sources of CH_4 (0.41 +/- 0.04 Tg CH_4/y) and oil and gas sources (0.22 +/- 0.06 Tg CH_4/y). Their estimate of oil and gas sources was based on concentrations of seven alkanes observed in the air, apportioned to sources using assumed compositions of emissions from those sources in a least-squares-fitting approach. They compared this oil and gas result to a CARB-inventory-estimated quantity of 0.064 Tg CH_4/y. Thus, they estimated that in the SoCAB region, inventory methods underestimate CH_4 emissions by a factor of 3.5 (2.5 to 4.4).

3.6.2.6. Jeong et al. (2013)

Jeong et al. (2013) used observations from five locations in California's central valley (SJV), including one tall tower (samples at ~90 and 480 m) and four small towers (samples at ~10 m). They combined these observations with aircraft observations of the Pacific boundary (i.e., incoming CH_4 concentrations) and urban regions. They compared these observations to a spatially resolved version of the CARB inventory, in which the CARB 2008 inventory was scaled to a detailed spatial emissions model. They also used the EDGAR spatially resolved inventory as a source of comparison emissions estimates, but this report focuses on California Greenhouse Gas Emissions Measurement program (CALGEM) comparisons, as these are more consistent with CARB inventory methods. In this study, atmospheric transport was modeled using an inverse approach with the WRF-STILT model (coupled weather research and forecasting–stochastic time-inverted lagrangian transport model). This approach traced "particles" of air backward through time in a time-inverted weather simulator, to estimate from where gases observed in particular locations were likely to have been emitted. This approach has been used in a number of national and regional atmospheric studies of GHGs.

The "prior" model in the Bayesian analysis of Jeong et al. (2013) is the spatially resolved CALGEM inventory, which predicts CO_2eq. CH_4 emissions of 28 TgCO_2eq./y. The emissions estimated incorporating the observations (the posterior estimate) is 48.3 Tg CO_2eq./y (+/- 6.5 at 1σ level). Thus, they argued that the CALGEM inventory is likely underpredicting California methane emissions.

3.6.2.7. Jeong et al. (2014)

Jeong et al. (2014) generated a much more detailed spatially resolved estimate of emissions from the California oil and gas industry than used in other studies. For example, well-level activity data (production of oil, gas, and water) were compiled from DOGGR data sources, while gas processing data were derived from federal U.S. EPA reporting. Also, pipeline fugitive emissions were modeled using detailed spatial representations of the California oil and gas distribution system. These activity factors were coupled to emissions factors (i.e., emissions per unit of activity) generally derived from U.S. EPA emissions factors. Lastly, they augmented this detailed "bottom-up" approach with data from the SoCAB region collected in atmospheric studies noted above (Wunch et al., 2009; Hsu et al., 2010; Wennberg et al., 2012; Peischl et al., 2013).

Jeong et al. (2014) found that using non-CARB emissions factors with detailed California activity data results in emissions estimates that are significantly larger than either the CARB GHG inventory or the CARB oil and gas survey. For example, the initial bottom-up result from their study was 330 Gg CH_4/y of emissions from all portions of the California oil and gas sector (uncertainty range 220-518 Gg CH_4/y). This compared to CARB GHG inventory and survey results of 210 and 204 Gg CH_4/y respectively. When they scaled their bottom-up approach to better match atmospheric observations, they found that their bottom-up estimate increases to 541 +/- 144 Gg CH_4/y. Thus, Jeong et al. (2014) found that the CARB inventory significantly under-predicts CH_4 emissions compared to what would be expected using existing U.S. EPA emissions factors or using atmospheric data.

3.6.2.8. Johnson et al. (2014)

Johnson et al. (2014) utilized aircraft observations in a series of flights taken in January and February of 2013 in the San Francisco Bay Area and northern San Joaquin Valley. They then coupled these observations to a 3-d atmospheric chemical transport model (GEOS-Chem) to derive flux estimates for CH_4 in the study region. They compared their results to the EDGAR spatially explicit emissions inventory.

They found that the EDGAR emissions inventory must be scaled by a factor of 1.3 to arrive at results that agree with atmospheric observations. They found that increasing oil and gas and waste (landfill) emissions by a factor of two results in a decrease in overall model bias, but degrades the model fit by overpredicting background CH_4 values. They found that increasing livestock emissions between a factor of two to seven would result in reduced overall model-observation bias and decrease overall RMSE (root mean square error). They argued that a correction factor of two for livestock emissions is not sufficient to correct overall underprediction, while a factor of seven is an upper limit. Therefore, Johnson et al. argued that in the SFBA and northern SJV region, it was likely that livestock CH_4 emissions were underestimated in existing spatial inventories. They did not directly compare their results to CARB inventories.

3.6.2.9. Gentner et al. (2014)

Gentner et al. (2014) used ground-based measurements with a meteorological transport model to examine the role of petroleum operations on emissions of hydrocarbon-derived VOCs. The meteorological model was used similarly to other studies above: back trajectories of parcels of air were traced over 6- and 12-hour periods to estimate sources of measured VOCs at the sampling location. These sources were then compared to spatial distributions of petroleum production operations (as well as dairy operations).

Gentner et al. found reasonable agreement between their sampling efforts in Bakersfield and the CARB inventory results. They found that 22% of VOC measured at their site could be attributed to petroleum operations, which was similar to their reported CARB partitioning for the SJV air district of 15%. Dairy sources were found to contribute 22%

(compared to 30% for CARB inventory) and motor vehicles 56% (compared to 55% for CARB inventory). In contrast, a smaller inventory comparison to just the Kern County portion of the SJV air district implies less petroleum emissions observed than expected in the inventory (as should be expected, given large petroleum operations in Kern County).

Gentner's explained fraction of ROG emissions in the SJV region, partitioned 15% to petroleum operations, is not in alignment with our computed value of 8% above. The causes for these differences were unable to be determined.

3.6.2.10. Summary Across Studies: How do Experimental Observations Align With California Inventory Efforts?

Taking the above experimental efforts in the aggregate, some general conclusions can be drawn about the California GHG inventory:

- Experimental evidence points to CARB inventories generally underpredicting CH_4 emissions in California. The degree of estimated underprediction varies by study, and no scientific consensus has yet emerged.

- Uncertainties are not reported for CARB inventories, and uncertainties for experimental studies are typically on the order of 15–30%.

- Studies point to livestock and oil and gas sources as drivers of these excess CH_4 emissions. There may be a regional effect observed here: livestock underprediction may be more important in studies focused on the northern SJV, while oil and gas under-prediction may be more important in studies focused on southern SJV and SC air districts.

- There is still considerable uncertainty in the observational literature about the precise level of CH_4 emissions from the California oil and gas industry.

- None of the experimental studies performed in California targeted well stimulation activities, so none of these studies provides evidence as to the accuracy of potential inventory treatment of well stimulation activities.

3.7. Findings

- Fields that are currently produced with well stimulation technologies in California have, on average, lower greenhouse gas emissions from oil production than a typical California oil field, and lower than fields produced without well stimulation.

- Because California produces a significant amount of high carbon intensity heavy crude oil in non-stimulated fields, reducing the use of well stimulation could result in an increasing reliance on more GHG-intensive sources of crude oil. More analysis involving market-based life cycle analysis is required to understand the potential impacts of removing hydraulic-fracturing-induced oil from California's oil supply.

- Current California air quality inventory methods likely include at least some well-stimulation-related emissions in their results. Inventory methods are not designed to estimate well stimulation emissions directly, and it is not possible to determine well stimulation emissions from current inventory methods.

- Using current inventory methods, the oil and gas sector is a minor contributor to GHG, emissions in California, contributing about 4% to state emissions.

- In the San Joaquin Valley, the oil and gas sector is a material contributor to TAC emissions, especially hydrogen sulfide, which is emitted mostly from oil and gas sources. In the San Joaquin Valley, the oil and gas sector contributes 30% of SO_x and 8% of ROG emissions.

- In the South Coast region, the oil and gas sector emits less than 1% of all studied species.

- Due to the fact that about 20% of California production is induced by well stimulation, direct and indirect impacts from well stimulation should be approximately 1/5 of above impacts.

- Local effects of air emissions can be more significant than the above analyses at the air basin scale. See Volume II, Chapter 6 for more discussion of local air impacts and impacts on populations that live near production sites.

- More research is required on overall leakage rates from oil and gas systems to better understand the breakdown of VOC and TAC emissions between sources (e.g., produced hydrocarbons, solvents, other process chemicals).

- Regulatory processes are currently in flux in a number of U.S. states, as well as federally. Current regulatory processes (e.g., federal EPA regulations) will greatly reduce some previously large emissions sources from well stimulation.

- Technologies exist to greatly reduce GHG and VOC emissions from well stimulation. Well stimulation direct emissions can be controlled through reduced emissions completions technologies.

- There are currently a number of significant gaps in the scientific literature with respect to the air emissions from well stimulation in particular, as well as in understanding air emissions from the oil and gas sector more generally.

3.8. Conclusions

Well stimulation is a potential source of air quality impacts in California. The oil and gas industry in general is a minor source of California's GHG emissions. In regions with large oil and gas sectors, such as the SJV region, the oil and gas industry is a major contributor to some TAC emissions and to SO_x emissions. The oil and gas industry materially contributes to ROG emissions as well. Because current inventory methodologies used in California were not designed to differentiate well stimulation emissions from other oil and gas emissions, it is not currently possible to estimate direct air emissions from well stimulation in California.

A number of regulatory and technical approaches to reducing emissions from well stimulation (and oil and gas production more generally) are available and currently used in at least some jurisdictions. The regulation of well stimulation emissions is still in flux at state and federal levels, and California is no exception.

The few studies that have examined well stimulation emissions directly have found that emissions are generally small, especially if control technologies are applied (as required by federal regulations for stimulated natural gas wells). These studies are few in number, so uncertainty still remains about the sources of air emissions from well stimulation. Given the importance of the California oil and gas sector for some emissions sources (e.g., TACs in the San Joaquin Valley), a significant induced increase of oil and gas production due to well stimulation could result in meaningful additional indirect air impacts. For other air quality concerns, or for smaller induced production volumes, it is unlikely that well stimulation will materially affect air quality.

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Chapter Four

Seismic Impacts Resulting from Well Stimulation

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4.1. Abstract

Induced seismicity refers to seismic events caused by human activities. These activities include injection of fluids into the subsurface, when elevated fluid pore pressures can lower the frictional strengths of faults and fractures leading to seismic rupture. The vast majority of induced earthquakes that have been attributed to fluid injection were too small to be perceptible by humans. However, events induced by injection have on several occasions been felt at the ground surface, and in extremely rare cases have produced ground shaking large enough to cause damage. These larger events can occur when large volumes of water are injected over long time periods (months to years) into zones in or near potentially active earthquake sources.

The relatively small fluid volumes and short time durations (hours) involved in most hydraulic fracturing operations are generally not sufficient to create pore-pressure perturbations of large enough spatial extent to generate induced seismicity of concern. Current hydraulic fracturing activity is not considered to pose a significant seismic hazard in California. To date, only one felt earthquake attributed to hydraulic fracturing in a California oil or gas field has been documented, and that was anomalous because it was a slow-slip event that radiated much lower energy at much lower dominant frequencies than ordinary earthquakes of similar size.

In contrast to hydraulic fracturing, earthquakes as large as magnitude 5.7 have been linked to injection of large volumes of wastewater into deep disposal wells in the eastern and central United States. Compared to states that have recently experienced large increases in induced seismicity, water volumes disposed per well in California are relatively small.

Despite decades of production and injection in oil and gas fields, extensive seismic monitoring, and vigorous seismological research in California, there are no published reports of induced seismicity associated with wastewater disposal related to oil and gas operations in the state. However, the potential seismic hazard posed by current water

disposal in California is uncertain because possible relationships between seismicity and wastewater injection have yet to be studied in detail. Injection of larger volumes of produced water from increased well stimulation activity and the subsequent increase in oil and gas production could conceivably increase the hazard. Given the active tectonic setting of California, it would be prudent to carry out assessments of induced seismic hazard and risk for future injection projects, based on a comprehensive study of spatial and temporal relationships between wastewater injection and seismicity.

The closest wastewater disposal wells to the San Andreas Fault (SAF) are located in oilfields just over 10 km (6.2 mi) away in the southern San Joaquin Valley. It is unlikely that current wastewater injection in these wells would induce earthquakes on the fault. If in the future significantly higher-volume injection were to take place in or close to these existing oilfields, then it is plausible that the likelihood of inducing earthquakes on the SAF could increase.

The probability of inducing larger, hazardous earthquakes by wastewater disposal could likely be reduced by following protocols similar to those that have been developed for other types of injection operations, such as enhanced geothermal. Even though hydraulic fracturing itself rarely induces felt earthquakes, application of similar protocols could protect against potential worst-case outcomes resulting from these operations as well.

4.2. Introduction

Induced seismicity refers to seismic events caused by human activities, which can include injection of fluids into the subsurface. The vast majority of induced earthquakes that have been attributed to fluid injection were too small to be perceptible by humans. However, seismic events induced by fluid injection have on several occasions been felt at the ground surface, and in extremely rare cases have produced ground shaking large enough to cause damage. This chapter reviews the current state of knowledge about induced seismicity, and discusses the data and research that would be required to determine the potential for induced seismicity in California, including along the SAF. Measures to assess and, if necessary, to reduce the risk from induced seismicity are also discussed.

4.2.1. Chapter Structure

This introductory section provides a brief overview of the general characteristics of earthquakes and the basic cause of earthquakes induced by subsurface fluid injection, followed by a summary of observed cases of induced seismicity related to well stimulation activities. Section 4.3 first discusses the potential impacts of induced seismicity in terms of the risks of nuisance and structural damage caused by ground shaking, and then describes the mechanics of fluid-induced earthquakes and the characteristics of seismicity sequences related to well stimulation. Section 4.4 considers factors that could influence the potential for well stimulation in California to induce seismicity, and describes the studies needed to assess that potential. Suggested measures to lower the likelihood of induced earthquakes occurring and hence reduce the risks are described in Section 4.5. Section 4.6 identifies gaps in the available data that presently limit our ability to evaluate induced seismicity in California, and then discusses potential actions to address those gaps. A summary of findings and conclusions are presented in Sections 4.7 and 4.8, respectively.

4.2.2. Natural and Induced Earthquakes

An earthquake is a seismic event that involves sudden slippage along an approximately planar fault or fracture in the Earth. This process occurs naturally as a result of stresses that build up owing to deformation within the Earth's crust and interior. The size of an earthquake depends primarily on the area of the patch on the fault that slips and the amount of relative displacement across the slip patch. Earthquake sizes range over many orders of magnitude. There are many more small events than large events; a decrease of one unit in the magnitude scale (see below and Appendix 4.A) corresponds roughly to a ten-fold increase in the number of events. As a result, the vast majority of earthquakes can only be detected by sensitive instruments. If, however, the slip area is sufficient to generate an earthquake larger than magnitude 2 to 3, the energy released during the event can generate seismic waves sufficient to produce ground motions that can be felt by humans, and larger events (usually about magnitude 5 and above) can in some cases cause structural damage. Over one million natural earthquakes of magnitude 2 or more occur worldwide every year (National Research Council (NRC), 2013).

As discussed in Appendix 4.A, several alternative magnitude scales are commonly used to express earthquake sizes. These employ different methods to compute magnitude, but all of the scales are roughly consistent with each other (within one-half magnitude unit) for earthquakes smaller than about magnitude 7. Henceforth in this report, we use published moment magnitudes, M_w. When discussing specific earthquakes for which M_w was not reported, we use the published magnitude, which, for the earthquakes discussed below, include only local magnitude, M_L and body-wave magnitude, m_b. In published cases when the scale was not specified, or to refer to magnitude in a general sense, we use the designation "M". Definition of the term "microseismicity" is somewhat arbitrary; for example, in earthquake seismology microseismicity usually refers to earthquakes smaller than M_w2-3, whereas in hydrofracture monitoring it commonly refers to events smaller than M_w0. In this report, we use microseismicity to describe earthquakes having magnitudes less than M_w3.

Earthquakes caused by human activity are termed *induced seismicity*. Activities that can induce earthquakes include underground mining, reservoir impoundment, and the injection and withdrawal of fluids as part of energy production activities (NRC, 2013). Note that some authors distinguish between "induced" and "triggered" events according to various criteria (e.g., McGarr et al., 2002; Baisch et al., 2009). In this report we do not make this distinction, but refer to all earthquakes that occur as a consequence of human activities as induced seismicity.

4.2.3. Induced Seismicity Related to Well Stimulation

Induced earthquakes related to well stimulation can be caused by injection of fluids into the subsurface, both for hydraulic fracturing stimulation itself and for disposal of recovered fluids and produced wastewater during stimulation and subsequent production. The predominant mechanism responsible for a fluid injection-induced earthquake is an increase in the pore-fluid pressure within a fault that reduces the confining stress that holds the two sides of the fault together, thus reducing its frictional resistance to slip (Hubbert and Rubey, 1959). Applying this mechanism to estimate the probability that seismic events of concern will be caused by a particular operation requires measurement or calculation of (1) the development of the subsurface pore-pressure perturbation in time and space, (2) characterization of faults likely to experience elevated pressures, and (3) characterization of rock material properties and *in situ* stress conditions. Because in practice these input parameters are often known only within broad bounds, an important part of the analysis is to properly constrain the uncertainties in order to correctly determine uncertainty bounds on the calculated event probabilities.

To date, the largest observed event attributed to hydraulic-fracture well stimulation is an M_L3.8 earthquake that occurred in the Horn River Basin, British Columbia, in 2011 (BC Oil and Gas Commission, 2012). The generally lower magnitudes of events associated with hydraulic fracturing relative to those induced by wastewater disposal are usually attributed to the short durations, smaller volumes, and flowback of injection fluids following stimulation, which result in smaller regions affected by elevated fluid pressures compared with the longer time periods and much higher volumes of wastewater injection. None of the events related to hydraulic fracturing reported in the literature have occurred in California and (with the possible exception of one paper that discusses an abnormal slow earthquake) we have found no published study that addressed this topic in California. If hydraulic fracturing operations carried out in California to date have, in fact, not induced normal seismic events above M2, possible explanations are that most of the well stimulation takes place in vertical wells at relatively shallow injection depths and employs relatively small injected volumes (Chapter 2). Volume I of this report concludes that salient features of hydraulic fracturing in California in the near- to mid-term are expected to be similar to those experienced thus far. If in the longer term hydraulic fracturing in the state shifts to larger injected volumes and deeper stimulation, then the likelihood of induced seismicity from hydrofracturing could increase.

The largest observed earthquake suspected to be related to wastewater disposal in the U.S. to date is a 2011 M_w5.7 event near Prague, Oklahoma (Keranen et al., 2013; Sumy et al., 2014), although the cause of this event is still under debate (Keller and Holland, 2013; McGarr, 2014). The largest earthquake clearly linked to stimulation-related wastewater injection is a 2011 M_w5.3 event in the Raton Basin of Colorado and New Mexico (Rubinstein et al., 2014). Despite decades of oil and gas production and wastewater injection, extensive seismic monitoring and exceptional in-depth research into the occurrence and mechanics of regional and local earthquakes, there are no published reports of induced seismicity caused by wastewater disposal related to oil and gas operations in California. However, there has been no comprehensive, in-depth study of the relationship between seismicity and disposal operations in the state.

Typical wastewater volumes injected per well in California are generally less than those associated with well stimulation operations in other parts of the country where induced seismicity has occurred. For example, typical wastewater volumes injected in Kern County to date have been about one fourth of those resulting from well stimulation in the Barnett shale and injected in the Dallas-Fort Worth area in Texas, where induced seismicity has been reported from ongoing observational studies. This might suggest that at the present time the potential for induced seismicity related to wastewater disposal in California may be relatively low compared with some other regions in the U.S. However, because the possible relationship between injection and seismicity in California has yet to be investigated, the potential seismic impact is at present unknown. Expanded well stimulation activity would require disposal of larger volumes of fluid, which would potentially increase the impact. Given the active tectonic setting of California, it will be prudent to carry out an assessment of induced seismic hazard and risk as part of the permitting process for future injection projects, particularly in areas where there are active faults and that experience naturally occurring seismicity. A comprehensive study of spatial and temporal relationships between wastewater injection and seismicity is necessary to provide a basis for such assessments. The chance of inducing larger, hazardous earthquakes would most likely be reduced by following protocols similar to those that have been developed for other types of injection operations, such as those for enhanced geothermal energy production (e.g. Majer et al., 2012).

4.3. Potential Impacts of Induced Seismicity

Induced seismicity can produce felt or even damaging ground motions when large volumes of water are injected over long time periods into zones in or near potentially active earthquake sources. The relatively small fluid volumes and short time durations involved in most hydraulic fracturing operations themselves are generally not sufficient to create pore-pressure perturbations of large enough spatial extent to generate induced seismicity of concern. In contrast, earthquakes as large as M_w5.7 have been linked to injection of large volumes of wastewater into deep disposal wells in the eastern and central United States (Keranen et al., 2013; Sumy et al., 2014).

Seismic hazard is defined as the probability that a specific level of ground shaking will occur at a particular location during in a specified interval of time. This formal definition is a departure from the meaning of the more general term “hazard”, which refers to possible negative outcomes or impacts. In this chapter, the word hazard alone indicates the more general possibility of impact, while the term *seismic hazard* will be used to refer to the formal definition used by the seismic hazard community. *Seismic risk* is the probability of a consequence, such as deaths and injuries or a particular degree of building damage, resulting from the shaking. Risk, as defined with regard to seismic ground motion, therefore combines the seismic hazard with the vulnerability of the population and built infrastructure to shaking, so that for the same seismic hazard, the risk is higher in densely populated areas. This use of the word risk is consistent with that used in other fields and involves both likelihood (probability of occurrence) and impact severity.

4.3.1. Building and Infrastructure Damage

Conventional seismic hazard and risk assessment deal with building and infrastructure damage—and the possible resulting injuries and loss of life—caused by strong ground shaking generated by naturally occurring earthquakes. The threshold magnitude for earthquakes to be capable of causing structural damage is generally considered to be about M_w5. Ground shaking from induced seismicity poses a potential incremental hazard above the natural background that needs to be considered in assessing the overall risk of an injection operation.

4.3.2. Nuisance from Seismic Ground Motion and Public Perception

Unlike assessing risk from naturally occurring seismicity, in the case of induced seismicity the likelihood of causing public nuisance from small events that are felt in nearby communities also has to be considered. This seismic risk includes minor cosmetic damage such as cracked plaster, as well as annoyance, alarm, and other adverse effects such as disrupted sleep. The magnitude threshold for felt events can be as low as M1.5–2.0 for the shallow depths of seismicity that are typically associated with fluid injection. In general, small earthquakes occur more frequently than large ones (see Section 4.2.2). Therefore, the frequency of occurrence of felt events can be relatively high, so that they may pose an ongoing impact on the quality of life in nearby communities.

4.3.3. Mechanics of Earthquakes Induced by Subsurface Fluid Injection

This section summarizes the physical mechanisms responsible for earthquakes induced by fluid injection. Fluid injection related to well stimulation takes place both for hydraulic fracturing and for wastewater disposal. In general, induced seismicity related to well stimulation is dominated by perturbations in fluid pore pressure, rather than by changes in *in situ* principal stresses (NRC, 2013). The characteristics of pore-pressure perturbations and induced seismicity resulting from hydraulic fracturing and wastewater disposal and their potential impacts are discussed in Sections 4.3.4 and 4.3.5, respectively.

During fluid injection there can be two types of rock failure, tensile and shear. Below we describe these two types of failure in the context of injection operations related to well stimulation.

4.3.3.1. Tensile Fracturing

The primary objective of hydraulic fracturing is to inject fluid into the earth to create a new fracture that connects the pores and existing fractures in the surrounding rock with the well, thus forming a permeable pathway that enables the oil and/or gas (and water) in the pores and fractures to be recovered. Hydraulic fractures are created by the rock failing in tension when the fluid pressure exceeds the *in situ* minimum principal stress (see Appendix 4.B). In this type of failure, a roughly planar fracture forms in the rock, and the walls of the fracture move apart perpendicular to the fracture plane at the same time as the fracture propagates (grows) at the crack tip in the direction parallel to the fracture plane. While there may be bursts of fracturing over short length scales at the crack tip, large-scale hydraulic fractures form slowly (hours) and can extend up to hundreds of meters away from the well. Although the physical processes at the crack tip are not yet fully understood, it appears that the amount of seismic energy radiated as the tensile fracture propagates is small and difficult to detect. Therefore, hydraulic fracture growth itself is responsible for little, if any, of the seismicity recorded in the field, and it probably makes little or no contribution to seismic hazard.

4.3.3.2. Shear Failure on Pre-existing Faults and Fractures

Shear failure on existing faults and fractures can occur both during stimulation by hydraulic fracturing and during wastewater disposal. During stimulation, shear events serve to enhance the permeability of small, existing fractures and faults and to link them up to create conductive networks connected to the main hydraulic fracture. Shear slip is the type of failure that occurs in most natural tectonic earthquakes, and it is shear events on larger faults that can produce perceptible or damaging ground motions at the Earth’s surface.

During a shear event the two faces of the fault slip in opposite directions to each other parallel to the fault surface. The conditions for the initiation of shear slip are governed by the balance between the shear stress applied parallel to the fault surface, the cohesion across the fault, and the frictional resistance to sliding (shear strength). Assuming that the cohesion is negligible, these conditions are summarized in the Coulomb criterion,

$$\tau = \mu\,(\sigma - p) \qquad (4\text{-}1)$$

in which an applied shear stress (τ) is balanced by the shear strength, which is the product of the coefficient of friction (μ) and the difference between normal stress (σ) and pore-fluid pressure (p). Shear stress is directed along the fault plane, while normal stress is directed perpendicular to the plane. The quantity (σ - p) is called the effective stress. Effective stress represents the difference between the normal stress, which pushes the two sides of the fault together and increases the frictional strength, and the fluid pressure within the fault, which has the opposite effect. The Coulomb criterion states that slip will occur when the shear stress (τ) exceeds the strength of the fracture (right-hand side of Equation 4-1). The shear stress that drives earthquake slip results from strain that accumulates in the Earth's crust, primarily as a result of tectonic and gravitational loading. An earthquake occurs when a fault fails in shear, releasing stored strain energy. In a tectonic earthquake, fault failure occurs when the accumulated shear stress reaches the critical value. Fault failure can also be initiated by decreasing the effective stress either by decreasing the normal stress (σ) that holds the fault closed and unable to slip, or by increasing the fluid pressure, which tends to push the sides of the fault apart, enabling slip.

4.3.3.3. Factors Influencing the Probability of Occurrence of Induced Earthquakes

If elevated pore pressures produced by either hydraulic fracturing or wastewater injection reach nearby faults or fractures, the resulting decrease in effective stress on the fault/fracture planes can cause induced seismicity. Therefore, in both activities, one consideration in developing an injection strategy should be to prevent the pressure perturbation from reaching larger faults capable of generating significant seismic events. This would help to minimize the seismic hazard and, in the case of well stimulation, to inhibit the fracture from propagating beyond the bounds of the hydrocarbon reservoir and providing a potential leakage pathway.

The primary factors that determine the probability of inducing seismic events are the volume of injected fluids, the spatial extent of the affected subsurface volume, ambient stress conditions, and the presence of faults that are well oriented for slip and are near-critically stressed (Appendix 4.B). The primary factors affecting the magnitude and extent, shape, and orientation of a pore-pressure perturbation include the injection rate and pressure, which are generally interdependent, the total volume injected, the hydraulic diffusivity (a measure of how fast a pore-pressure perturbation propagates in the fluids in the pore space), and the stress state and natural fracture orientation and conductivity under injection conditions. At early stages of an injection, the extent of the pressure perturbation depends on the hydraulic diffusivity and the duration of the injection, while the maximum pore pressure depends on the product of injection rate and duration divided by the permeability (NRC, 2013). At later stages, the induced pore-pressure field does not depend on the injection rate or permeability, but becomes proportional to the total volume of fluid injected.

4.3.3.4. Maximum Magnitude of Induced Earthquakes

The vast majority of earthquakes induced by fluid injection in general do not exceed M1 (e.g., Davies et al., 2013; Ellsworth, 2013). However, larger magnitude earthquakes (M > 2) have resulted from both wastewater injection and hydraulic fracturing. McGarr (2014) proposed estimating upper bounds on induced earthquake magnitudes based on net total injected fluid volume, observing that such a relationship is found to be valid for the largest induced earthquakes that have been attributed to fluid injection. Shapiro et al. (2011) proposed a similar approach to estimating maximum magnitude, based on the dimensions of the overpressurized zone deduced from observed microseismicity. Brodsky and Lajoie

(2013) also concluded that induced seismicity rates associated with the Salton Sea geothermal field correlate with net injected volume. However, the approaches proposed by both McGarr (2014) and Shapiro et al. (2011) appear to imply that fault rupture induced by the injection occurs only within the volume of pore-pressure increase. An alternative hypothesis is that a rupture that initiates on a fault patch within the overpressured volume can continue to propagate beyond its boundaries, in which case the possible maximum magnitude is determined by the size of the entire fault. Indeed, McGarr (2014) does not regard that his relationship determines an absolute physical limit on event size.

4.3.4. Induced Seismicity Resulting from Hydraulic Fracturing Operations

Because hydraulic fracture treatments are carried with relatively small injected volumes over short time periods and a proportion of the fluid flows back up the well following stimulation, the volume of the subsurface affected by pressure perturbations is usually confined within a few hundred meters of the wellbore, as shown by microseismic and tiltmeter fracture mapping results (e.g., Shemeta et al., 1994; Shapiro and Dinske, 2009; Davies et al., 2012; Fisher et al., 2002; Fisher et al., 2004). Davies et al. (2013) cite evidence to suggest that induced shear events in the vicinities of stimulation zones are mainly caused by fluids leaking off into preexisting faults and fractures intersected by the hydraulic fracture. Shear failure may also occur on nearby, favorably oriented faults and fractures isolated from the zone of increased pressure due to perturbation of the local stress field near the tip of the propagating hydraulic fracture (e.g., Rutledge and Phillips, 2003).

There can be a time delay between the beginning of injection and the occurrence of larger (M > 2) events, and in several cases the largest event has occurred after injection ceases. The longest time delay observed to date following a well stimulation injection was almost 24 hours before the occurrence of the largest (M_L3.8) event at the Horn River Basin, BC site (BC Oil and Gas Commission, 2012). A 2011 M_L2.3 earthquake in Blackpool, UK, occurred about 10 hours after injection ceased at the Preese Hall 1 stimulation well (de Pater and Baisch, 2011).

Overall, because of the relatively small volumes of rock that experience elevated pressures, there is a lower potential seismic hazard from short-duration hydraulic fracture operations than from disposal of large volumes of wastewater. The fact that, to date, the maximum magnitudes of events caused by hydraulic fracturing have been well below those usually considered to be capable of causing damage suggests that the likelihood of damaging events being induced by hydraulic fracturing is very low.

Published cases of known or suspected fluid injection-induced seismicity resulting from well stimulation and wastewater disposal that included events greater than M1.5 are described in Appendix 4.C. Five out of the six seismicity sequences listed in Table 4.C-1 attributed to hydraulic fracturing worldwide included felt earthquakes, and in all but one of these five cases, only one or two events were reported felt. This suggests that the risk of nuisance is also quite low. However, it is pertinent that all but one of the cases involving felt earthquakes have occurred during the major upsurge in well stimulation activity since 2010, so that a further increase in activity in a particular region may increase the overall seismic hazard and risk there beyond past experience.

4.3.5. Induced Seismicity Resulting from Wastewater Disposal

Large-scale, continuous injection of wastewater into a single formation over time periods of months to years commonly generates overpressure fields of much larger extent than those resulting from well stimulation. For example, at the Rocky Mountain Arsenal, Colorado significant earthquakes caused by fluid injection occurred 10 km (6.2 mi) away from the injection well (Healy et al., 1968; Herrmann et al., 1981; Nicholson and Wesson, 1990). Hydrologic modeling of injection into the deep well at the site indicated that the seismicity tracked a critical pressure surface of 3.2 MPa (Hsieh and Bredehoeft, 1981). Long time delays between the cessation of injection and the occurrence of larger events have also been observed in several cases. For example, at the Rocky Mountain Arsenal, the largest earthquake (M_w4.8) occurred 17 months after injection ceased (Herrmann et al., 1981).

Generally, the likelihood of inducing larger events increases as the volume of injected wastewater increases. The largest earthquake suspected of being related to wastewater disposal is the 2011 M_w5.7 Prague, Oklahoma event (Keranen et al., 2013; Sumy et al., 2014), but the causal mechanism of this event is still the subject of active research, and the possibility that it was a natural tectonic earthquake cannot confidently be ruled out at present. The largest earthquake for which there is clear evidence for a causative link to stimulation-related wastewater injection is the 2011 M_w5.3 event in the Raton Basin of Colorado and New Mexico (Rubinstein et al., 2014). It is important to note, however, that significant induced seismicity has occurred at very few of the tens of thousands of wastewater disposal wells currently or formerly active in the U.S. (e.g., NRC, 2013; Ellsworth, 2013; Weingarten and Ge, 2014).

In most of the reported cases of induced seismicity associated with wastewater disposal listed in Table 4.C-1, events occurred both in the sedimentary formation into which the injection took place and, except in the Dallas-Fort Worth and Cleburne, Texas sequences, in the underlying crystalline basement rocks. In all of the cases, the seismicity illuminated planar features that were interpreted as favorably oriented faults reactivated by injection. Most of the faults interpreted from the seismicity had not been mapped on the ground surface. Reactivation of faults well below the injection interval can occur if there is hydraulic communication between them and the well (Horton, 2012; Justinic et al., 2013), and although the matrix permeability of basement rock is generally very low, critically stressed faults and fractures in this part of the brittle crust can serve as high permeability channels (Townend and Zoback, 2000; Fehler et al., 1998; Shapiro et al., 2003). The maximum depth of seismicity in the cases listed in Table 4.C-1 ranged from about 4 to 8 km (2.5 to 5 mi).

All seven of the M4 and larger earthquakes that occurred within the fluid injection-induced seismicity sequences listed in Table 4C-1 and that have relatively accurate hypocentral locations constrained by local seismic networks nucleated at depths between 3 and 6 km (1.9 and 3.7 mi). This depth range is assumed to correspond to the zone some distance below the injection interval where high fluid overpressures over relatively large fault areas coincide with stresses that put favorably oriented faults into a near-critical state; i.e. where the pressure reaches the critical value needed to nucleate a larger event. Deeper seismicity corresponds both to aftershocks of the larger events and to smaller magnitude events perhaps triggered at lower pressures.

Relatively high seismic hazard from earthquakes below M_w4.5 translates into a greater risk of nuisance if the seismicity occurs close to inhabited areas. Of the 14 events in Table 4.C-1 attributed to wastewater disposal, five were larger than M_w4.5. Only three of these, the M_w4.8 1967 Rocky Mountain Arsenal, the M_w5.7 2011 Prague, and the M_w5.3 2011 Raton Basin events, caused anything more significant than localized minor damage. However, as noted above, events as small as about M_w5 are generally considered to be capable of causing significant damage under certain circumstances (shallow focal depth, construction that is not seismically resistant, etc.), at least in the vicinity of the epicenter. Therefore, although it may be low in absolute terms, the seismic risk of damage associated with wastewater injection is relatively much greater than that associated with well stimulation. In view of the dramatic increase in seismicity—including all but one of the events greater than M_w4.5 in Table 4.C-1—that has accompanied the upswing in wastewater disposal in some parts of the U.S. beginning in 2010 (see U.S. Geological Survey, 2015), a future increase in the rate of operations in a particular region may increase the likelihood of damage there, as well as nuisance.

4.4. Potential for Induced Seismicity in California

All of the U.S. cases of induced seismicity related to fluid injection discussed in Appendix 4.C occurred within the continental interior, where tectonic deformation rates are very low. California, on the other hand, is situated within an active tectonic plate margin, where the rapid buildup of shear stress on the numerous active faults (Figure 4.4-1) results in much higher seismicity rates in many areas of the state than in the continental interior, as can be seen in Figure 4.4-2. If, as discussed in Appendix 4.B, the Earth's upper crust is generally in a near-critical stress state, then the high loading rates would imply that a relatively high proportion of faults in California will be close to failure at any given time, and hence susceptible to earthquakes triggered by small effective stress or shear stress perturbations.

4.4.1. California Faults and Stress Field

Unlike the central and eastern U.S., a large number of active faults have been mapped at the Earth's surface and characterized in California. Figures 4.4-1 and 4.4-2 show the surface traces of active faults in central and southern California contained in the U.S. Quaternary Fault and Fold (USQFF) database (http://pubs.usgs.gov/fs/2004/3033/fs-2004-3033.html). This database contains descriptions of faults known or believed to have been active during the Quaternary period (the last 1.6 million years). While particular attention should be paid to these faults in assessing the potential for induced seismicity and in siting injection operations, local faults that are suitably oriented for slip in the prevailing *in situ* stress field (see Appendix 4.B) also need to be taken into account, as does the possible presence of unmapped faults like the basement faults activated in some of the recent cases of mid-continent induced seismicity discussed above. This is further discussed in Section 4.6.3 below.

Figure 4.4-1 shows the relationship of faults to the higher-quality (quality A-C) stress measurements in central and southern California taken from the World Stress Map database (Heidbach et al., 2008), which is the most recent compilation of tectonic stress orientations, and in some cases the magnitudes of principal stress components. These measurements are derived from observations of wellbore breakouts, earthquake focal mechanisms, pressure and tiltmeter monitoring of hydraulic fractures, and geological strain indicators.

Figure 4.4-1. High-quality stress measurements for central and southern California from the World Stress Map (Heidbach et al., 2008), plotted with mapped faults from the USQFF database. The line plotted at the location of each stress measurement shows the orientation of the maximum horizontal compressive stress direction, color-coded according to stress regime.

4.4.2. California Seismicity

The generally low magnitude earthquake detection threshold in California, discussed in Appendix 4.A, means that California earthquake catalogs provide a relatively high-resolution picture of seismicity in the state as a whole. Figure 4.4-2 shows high-precision, relocated epicenters of California earthquakes M≥3 recorded in central and southern California between 1981 and 2011 contained in the Southern California Earthquake Data

Center (SCEDC) 2013 catalog (Hauksson et al., 2012). Intense seismicity occurs along segments of the major fault systems like the SAF zone in central California, in addition to relatively frequent (10s to 100s of years), large ($M_w \geq 6$) earthquakes. Large events accompanied by aftershock sequences have also occurred during this 30-year time period under the western slopes of the Central Valley near Coalinga (1983) and Kettleman Hills (1985), near Northridge (1994) and Whittier (1987, M5.9) north of Los Angeles, and along the coast near San Simeon (2003). Elsewhere, lower-magnitude seismicity is generally more diffuse. In addition to the Los Angeles Basin, oil-producing areas of the southernmost San Joaquin Valley and the Ventura Basin have relatively high rates of seismicity in the M2-5 range.

The vast majority of earthquakes in California are naturally occurring, but we can still question whether some of them may have been induced by fluid injection related to oil and gas recovery. The bulk of the seismicity that occurs in California is located at depths below about 2-3 km (1.2 – 1.9 mi). Therefore, the upper boundary of the main seismogenic zone is within about the same depth range as the deepest wastewater disposal wells for which depth information is available in the DOGGR (2014a) database, and about 1 km (0.6 mi) deeper than the depths of the wells having the highest cumulative injected volumes (see Section 4.4.3.1). Based just on the observed depths of earthquakes relative to injection depths in the reported cases of induced seismicity discussed in Section 4.3.5, it would appear that the overall potential for seismicity to be induced by wastewater injection may be at least as high in California as in the central U.S. Furthermore, some M5-6 events are observed to occur at relatively shallow depths in California, which suggests that induced earthquakes could be at least as large as those experienced to date in the continental interior. For example, ten (out of a total of 98) M5-6 earthquakes in the Hauksson et al. (2012) 1981–2011 catalog have focal depths between 3 and 6 km (1.9 and 3.7 mi), the depth range of M4 and larger induced events in the mid-continent (Section 4.3.5).

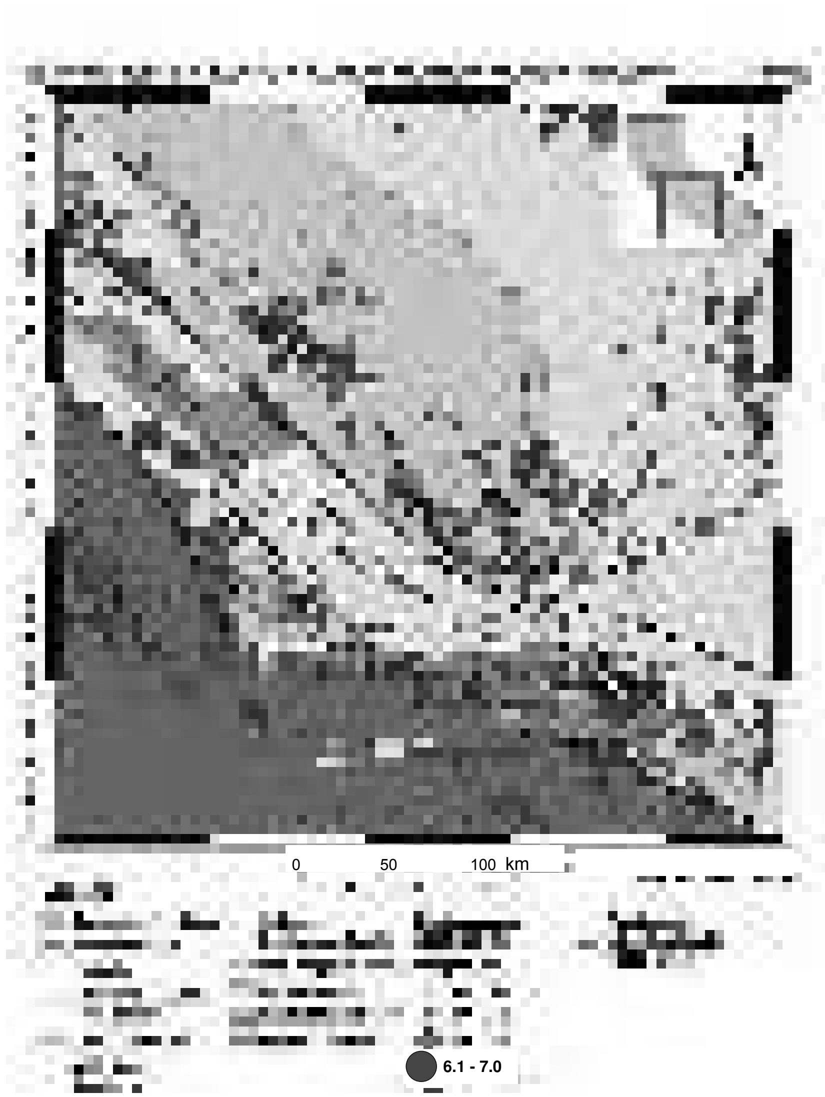

Figure 4.4-2. High-precision locations for earthquakes M≥3 in central and southern California during the period 1981-2011 (Hauksson et al., 2012), and active and previously active water disposal wells from DOGGR (2014a). Faults as in Figure 4.4-1.

While the above argument suggests that induced seismicity could potentially be caused by wastewater disposal in California, analysis of the relationship of seismicity to injection operations in the state is necessary to find out if that is indeed the case and, if so, to assess the resulting seismic hazard. Despite decades of injection in Californian oil and gas fields and one of the most active seismological monitoring and research programs in the world, no systematic study to explore possible associations between seismicity and fluid injection related to oil and gas production in the state has yet been completed. Although there have been numerous studies of induced seismicity associated with injection and production in geothermal fields in California (e.g., Eberhart-Phillips and Oppenheimer, 1984; Majer et al., 2007; Kaven et al., 2014; Brodsky and Lajoie, 2013), and microseismic monitoring is routinely used to monitor hydraulic fracturing in oilfields (e.g., Murer et al., 2012; Cardno ENTRIX, 2012), we have found only one published paper (Kanamori and Hauksson, 1992) in which a California earthquake greater than M2 was linked to oilfield fluid injection. In that case, the authors attributed the occurrence of a very shallow M_L3.5 slow-slip event to hydraulic fracturing at the Orcutt oilfield in the Santa Maria Basin. This event was anomalous in that it radiated much lower energy at much lower dominant frequencies than normal earthquakes of similar size.

4.4.3. Correlation of Seismicity and Faulting with Injection Activity in California

One of the reasons that there have been no detailed studies of possible links between fluid injection in Californian oilfields and seismicity until recently is that small, naturally occurring earthquakes are very frequent in many regions of California, making it difficult to discriminate induced events in the M2-4 range from natural events (e.g., Brodsky and Lajoie, 2013). In contrast, natural seismicity rates are very low over most of the central and Eastern U.S., so if an earthquake does occur it is much easier to investigate whether the cause could be anthropogenic. However, Goebel et al. (2014) have reported initial results of a study that suggests that wastewater injection contributes to seismicity in Kern County, and Hauksson et al. (2014) have begun to study the relationship of seismicity to injection and production in the Los Angeles basin.

The most direct way to identify potential injection-induced seismicity on a statewide basis would be to conduct a comprehensive, systematic search for statistically significant spatial and temporal correlations between earthquake occurrence and injection rate, pressure, depth and distance from suitably oriented faults at a local scale within each oil-producing basin. A complete correlation analysis is beyond the scope of the present review. What this section does include is a summary of injection depths and volumes in California and an overview of the locations of injection wells relative to mapped faults and seismicity. Then a preliminary example of exploratory data analysis that seeks to identify relationships between injection and seismicity is presented. Given its generally higher potential for inducing seismicity of concern, we focus on wastewater disposal in California since 1981.

4.4.3.1. Depths and Volumes of California Wastewater Injection

The basic data required to carry out detailed correlation analyses include comprehensive records of the volume and pressure time histories and depths of injection in wastewater disposal wells in California. However, in the California Division of Oil, Gas and Geothermal Resources (DOGGR) database (DOGGR, 2014a), depth information is given for only 13% (329) of water disposal wells active since 1981. Reported depths range from 60 m (197 ft) to 4.42 km (14,500 ft). Of these, 21 currently active water disposal wells in their present configurations have recorded depths greater than 1.8 km (5,905 ft). The depth range for the ten highest-volume injection wells for which depth information is available is

732–838 m (2,400–2,750 ft). Compared with, for example, permitted injection intervals of 3.3–4.2 km (10,827–13,780 ft) in the Ellenberger Formation underlying the Barnett shale (Frohlich et al., 2010), the available data suggest that typical wastewater injection depths in California are about 1.5–3 km (4,921–9,842 ft) shallower than in Tarrant County in Texas, where the 2008–2009 Dallas-Fort Worth induced seismicity sequence occurred (see Appendix 4.C). However, this comparison is based on the very limited sample of California disposal wells for which depths are available.

Previous case studies show that the occurrence of induced earthquakes is usually closely associated with short-term changes in injection volume and pressure. Therefore, volume and pressure time histories sampled at intervals minutes to hours are ideally required to carry out detailed correlation analyses. However, volumes and pressures are reported on a monthly basis in the DOGGR database. The reported volume rates and pressures are assumed to be monthly averages.

Currently, average annual wastewater disposal volumes per well in California are generally less than in other regions in the U.S. where well stimulation is taking place. According to DOGGR (2010) (the most recent annual report available), total annual wastewater injected in 2009 in Kern County was approximately 79.4 million m^3 (21 billion gal) into 611 active wells, or an average disposal rate of about 360 m^3 (95,100 gal) per well per day. This, for example, is less than one-fourth of typical water disposal rates of 1,590 m^3 (420,000 gal) per well per day in Tarrant and Johnson Counties, Texas (Frohlich et al., 2010). In the Raton Basin of Colorado and New Mexico, an increase in the average daily rate of fluid injection to 300 m^3 /day (79,250 gal/day) per well, comparable to California's average daily disposal rate, was linked to a significant increase in the number of earthquakes greater than M3 (Rubenstein et al., 2014) (see Appendix 4.C), but in this case the increase in injection rate took place simultaneously in 21 wells within the basin.

In terms of cumulative volume, there are 27 wells in California that have cumulative injected volumes since 1977 greater than 16 million m^3 (4.2 billion gal), 13 of which are located on the eastern side of the southern San Joaquin Valley near Bakersfield. Further investigation is needed to determine if these 13 high-volume wells were injecting into the same pool. If this is the case, and if the wastewater was not injected into the same interval as it was produced from, then the aggregate injected volume into the pool between 1977 and 2013 was 334 million m^3 (88.2 billion gal). This is only one-half the aggregate injected volume reported for the Raton Basin during the main period of induced seismicity there between 2006 and 2013, when the 21 injection wells each disposed of 33 million m^3 (8.7 billion gal) (Rubenstein et al., 2014). The reported aggregate volume for the Raton Basin does not include the volume injected between 1995 and 2006, when the field was under development.

4.4.3.2. Locations of Wastewater Injection Wells Relative to Mapped Faults and Seismicity

Many active faults in California are not confined to the basement or deeper sedimentary layers but extend all the way to the Earth's surface. This means that in many cases the lateral distance from a disposal well to a fault is likely as important as the depth of injection in determining whether a hydraulic connection is established that allows injection-induced pressure changes to reach the fault. Although cases like the Rocky Mountain Arsenal and Raton Basin indicate that pressure perturbations large enough to induce earthquakes can travel distances up to10 km (6.2 mi) or more along fault zones, in all but one of the cases of mid-continent induced seismicity discussed in Section 4.3.5 the injection wells were located less than 3 km (1.9 mi) laterally from the fault defined by the seismicity. The exception was Paradox Valley, Colorado, where the largest event (M_w4.0 in January, 2013) induced by 17 years of continuous high-rate injection occurred on a fault located 8 km (5 mi) away from the well (Block et al., 2014). The cumulative volume injected in the Paradox Valley well between 1996 and 2012 was about 8.5 million m^3 (2.2 billion gal), about half of typical cumulative volumes injected into the 27 highest-volume wastewater disposal wells in California since 1977. It is important to note that there is a high-permeability pathway between the Paradox Valley well and the fault activated in the 2013 event, which apparently corresponds to a regional-scale fracture zone (King et al., 2014).

These well-fault distances provide the context for the following brief summary of spatial relationships between wastewater injections wells and surface faults and seismicity in oil-producing basins in California.

Figure 4.4-3 summarizes the distribution of distances between wastewater disposal wells active since 1981 and faults in the USQFF database in six oil-producing basins in California. Across all six basins, over 1,000 wells are located within 2.5 km (1.5 mi) of a mapped active fault, and more than 150 within 200 m (656 ft).

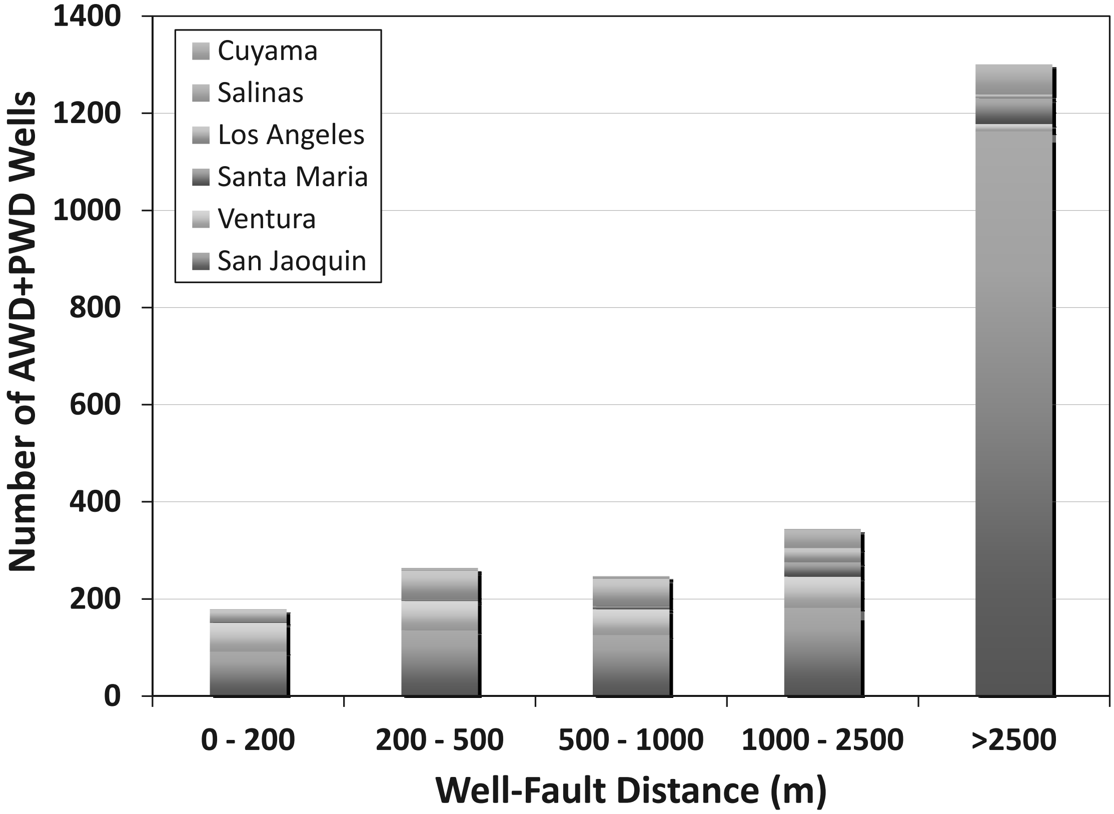

Figure 4.4-3. Distribution of distances between wastewater disposal wells active during the period 1981-present (DOGGR, 2014a) and Quaternary active faults in the six major oil-producing basins in California.

The maps in Figures 4.4-4 to 4.4-9 show the locations of disposal wells relative to mapped faults and seismicity in four of the largest oil-producing basins. The faults are colored according to the estimated time of their last earthquake activity as follows: historic (red), <~150 years; Holocene/latest Pleistocene (orange), <15,000 years; latest-Quaternary (yellow), <130,000 years; Quaternary (blue), <1.6 million years. The most recently active faults and those with the highest long-term slip rates are considered to be the ones most likely to experience future earthquakes. Long-term slip rates of California faults range from less than 0.1 mm/yr to 34 mm/yr on the SAF.

The historically active trace of the White Wolf fault (slip rate 2 mm/yr) delineates the southeastern boundary of the San Joaquin Valley (Figures 4.4-4 and 4.4-5). This fault last ruptured in the 1952 M7.3 Kern County earthquake. (Other red traces on Figures 4.4-5 and 4.4-6 are ground fractures mapped following the 1952 earthquake or have been linked to oilfield subsidence, and so they might not correspond to active faults.) The closest well to the White Wolf fault is about 5 km (3.1 mi) south the surface trace (Figure 4.4-5). The densest concentration of seismicity is located to the southwest, where two

Quaternary faults continue the trend of the historic White Wolf trace, and Holocene and Quaternary traces of the Pleito fault system are also mapped. In addition to abundant microseismicity, M4.7 and M5.1 earthquakes occurred in this area in 2005. Several injection wells are located within 1 km (0.6 mi) of a Quaternary strand of the Pleito system. Clusters of microearthquakes have occurred close to several of the injection wells in this area, but others are located away from the wells.

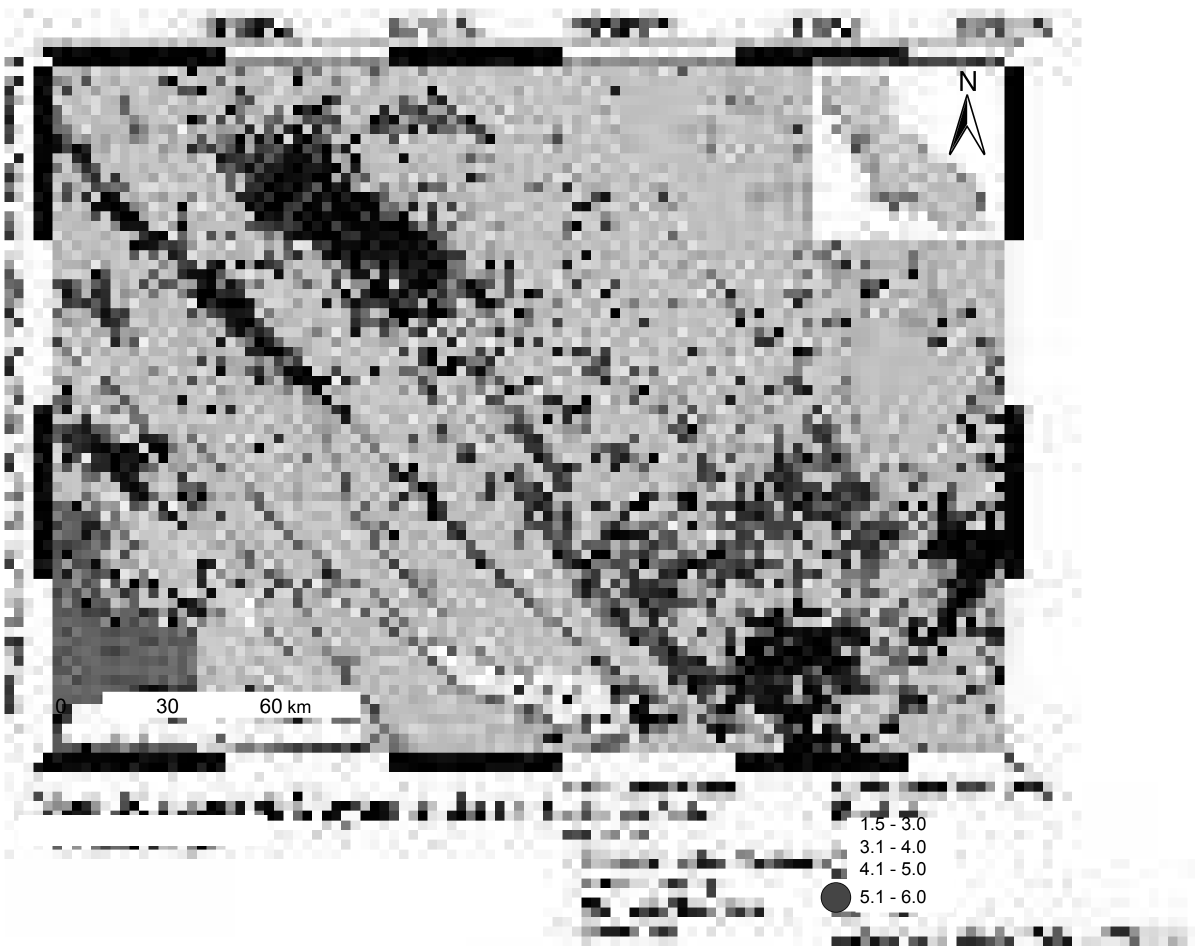

Figure 4.4-4. Earthquakes M≥1.5 in the southern San Joaquin Valley and Cuyama Basin from Hauksson et al. 2012, plotted with active and previously active water disposal wells from DOGGR (2014a) and faults from the USQFF database. Faults colored according to the time of most recent activity. The White Wolf fault is the red trace in the southeast corner of the Valley.

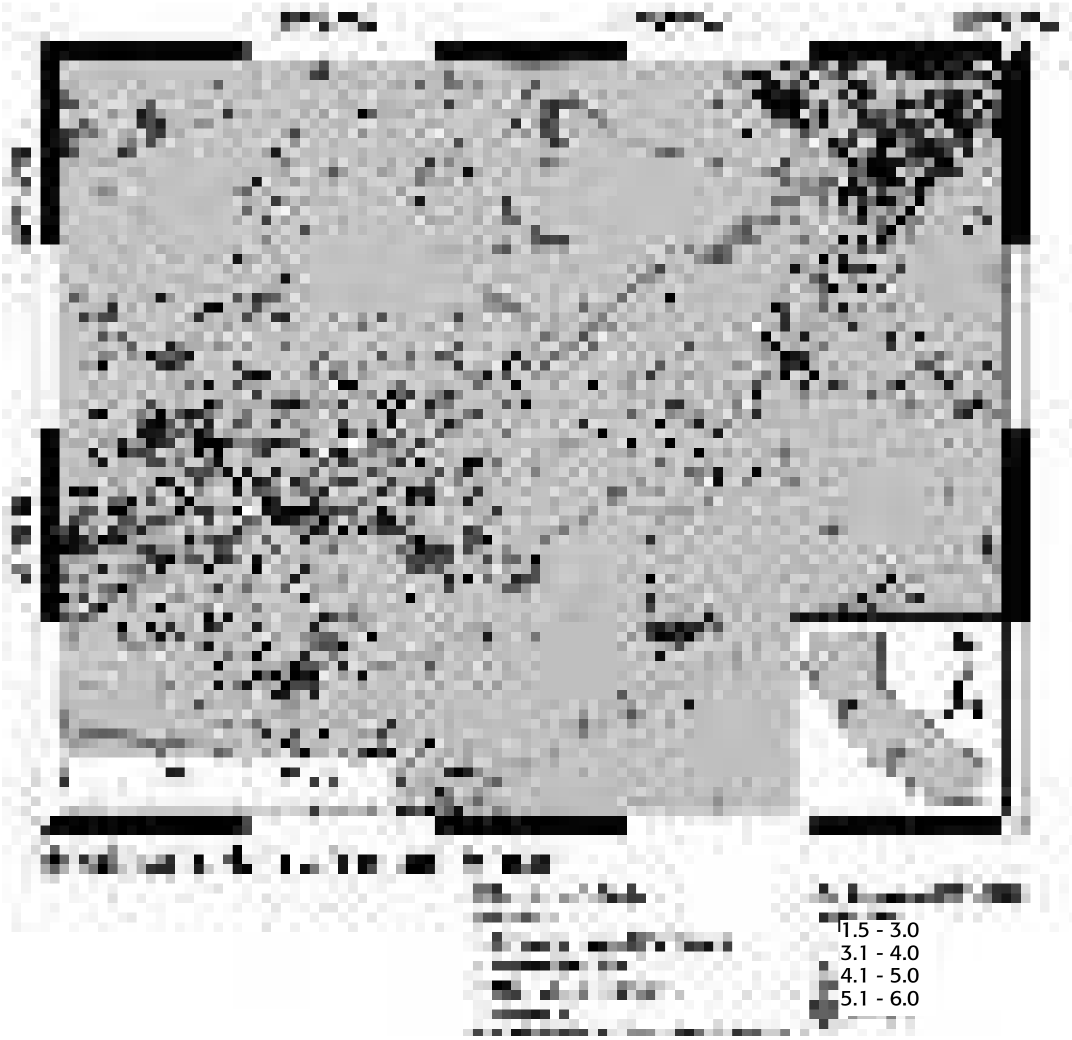

Figure 4.4-5. Earthquakes M≥1.5 in the southernmost San Joaquin Valley from Hauksson et al. (2012). Wells and faults as in Figure 4.4-4.

Quaternary and latest-Quaternary faults are mapped at the surface near the dense concentrations of disposal wells towards the eastern margin of the San Joaquin Valley in the vicinity of Bakersfield (Figure 4.4-6). Many of the Quaternary faults strike roughly north-south and are not favorably oriented for reactivation within the prevailing stress field (Figure 4.4-1). The green triangles show the locations of 13 of the 27 disposal wells in California having cumulative injected volumes greater than 16 million m^3 (4.2 billion gal). Earthquakes are observed only infrequently in this area. There is also only sparse, scattered seismicity near the long chain of disposal wells along the southwestern margin of the San Joaquin Valley (Figure 4.4-4). Most of the earthquakes in the dense cluster further northwest are aftershocks of M_w6.5 and M_w6.1 earthquakes that occurred in 1983 and 1985, respectively on deeply buried (blind) faults (U.S. Geological Survey, 1990; Ekström et al., 1992).

In the Santa Maria Basin, numerous wastewater disposal wells are located within 1–2 km (0.6–1.2 mi) of the surface traces of favorably oriented northwest-striking latest-Quaternary and Quaternary fault systems (Figure 4.4-7). All of the faults close to oilfields in the Santa Maria Basin have estimated slip rates less than 1 mm/yr (see California Geological Survey, 1996). The only dense cluster of seismicity is in the vicinity of the group of wells in the east-central part of the basin located in the Zaca oilfield. This cluster is discussed in Section 4.4.3.3 below. Numerous disposal wells in the Ventura Basin are sited very close to mapped Holocene-active faults, most notably along the major, west-striking Holocene San Cayetano system (slip rate 6 mm/yr) in the northern part of the basin, and to latest-Quaternary faults (Figure 4.4-8). Pockets of dense seismicity are located both close to and remote from injection wells. Most of the events in the dense cloud of seismicity at the eastern end of the basin are aftershocks of the deep (21 km; 13 mi) 1994 Northridge earthquake.

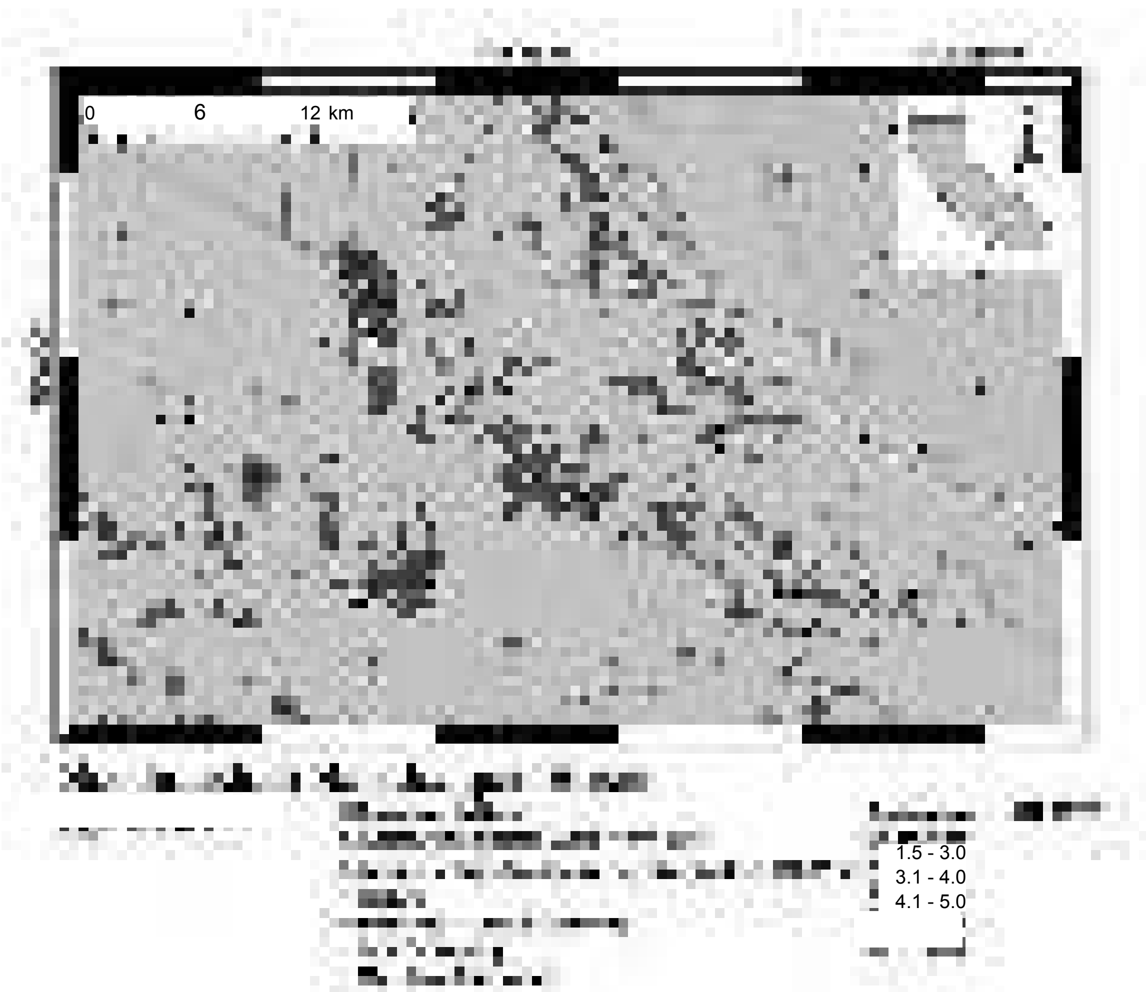

Figure 4.4-6. Earthquakes M≥1.5 in the southeastern San Joaquin Valley near Bakersfield from Hauksson et al. (2012). Wells having cumulative injected wastewater volumes >16 million m³

(4.2 billion gal) shown in green. Other wells and faults as in Figure 4.4-4.

In the Los Angeles Basin (Figure 4.4-9), disposal wells are concentrated mainly in oilfields located along the Holocene Newport-Inglewood fault zone (slip rate 1.5 mm/yr), a segment of which was the source of the destructive 1933 M_w6.4 Long Beach earthquake, and in the Wilmington oilfield. Several wells in the Wilmington field are located within 4 km (2.5 mi) of the Holocene Palos Verdes fault (slip rate 3 mm/yr). Only scattered seismicity has occurred near any these fields except Inglewood and Cheviot Hills at the northwestern end of the Newport-Inglewood trend. As in the Ventura Basin, clusters of seismicity are located close to some disposal wells but also elsewhere. The cluster at the top-center of the figure are aftershocks of the 2014 La Habra earthquake.

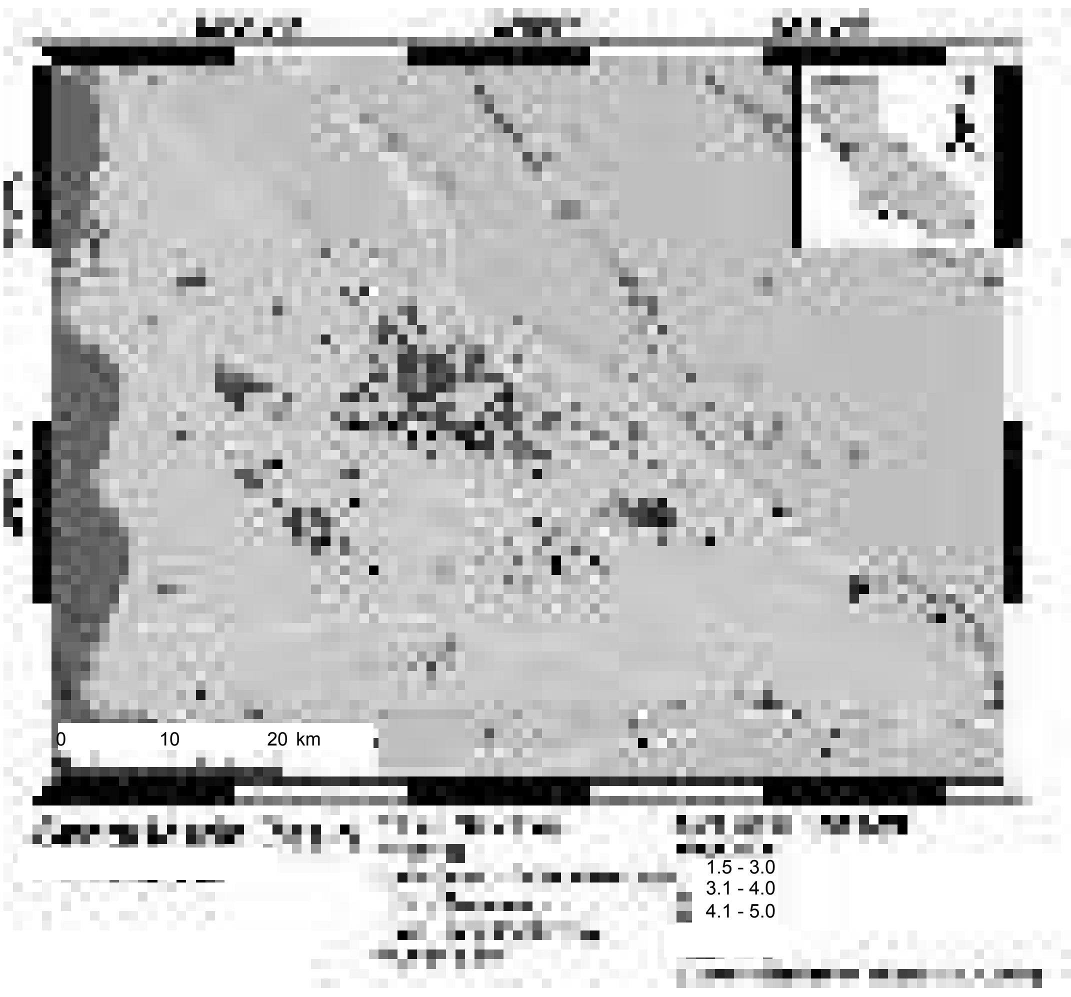

Figure 4.4-7. Earthquakes M≥1.5 in the Santa Maria Basin from Hauksson et al. (2012). Wells and faults as in Figure 4.4-4.

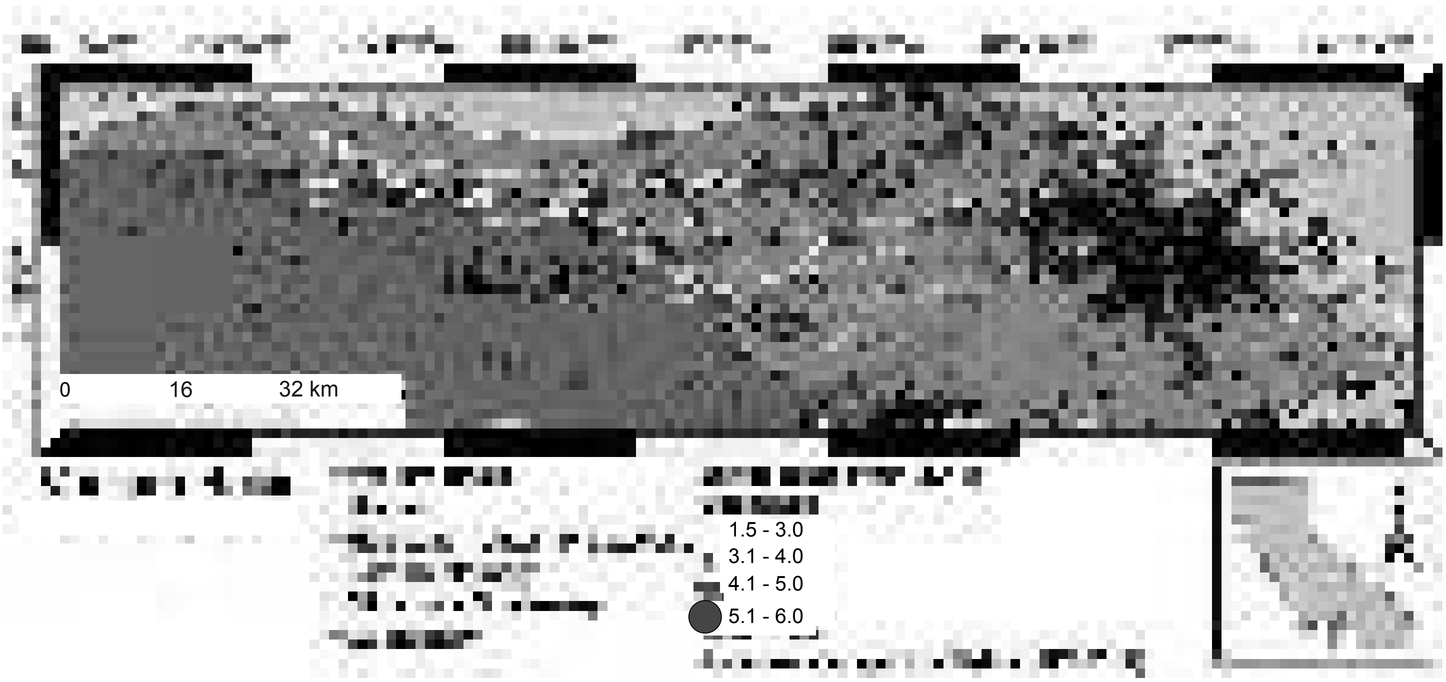

Figure 4.4-8. Earthquakes M≥1.5 in the Ventura Basin from Hauksson et al. (2012). Wells and faults as in Figure 4.4-4.

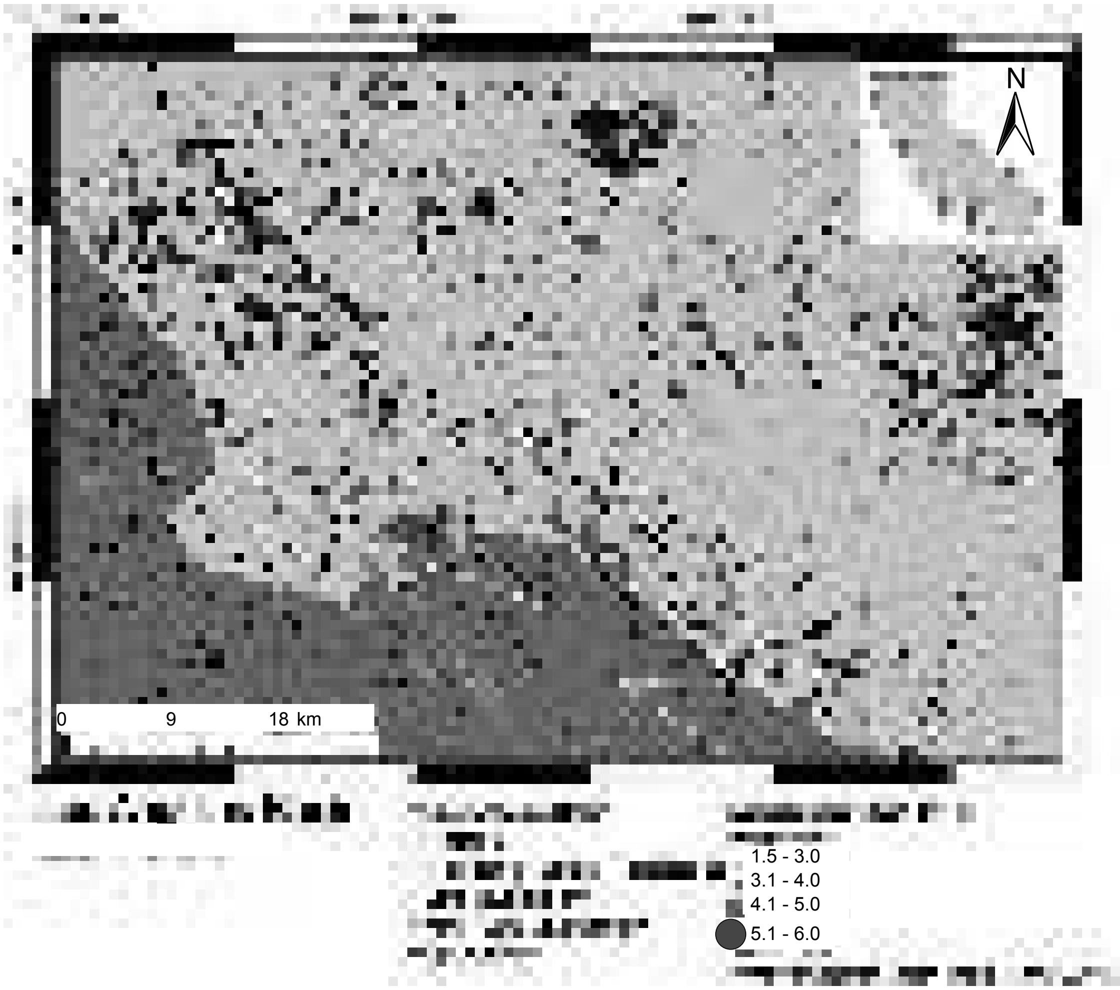

Figure 4.4-9. Earthquakes M≥1.5 in the Los Angeles Basin from Hauksson et al. (2012). Wells and faults as in Figure 4.4-4.

While numerous disposal wells in some of the basins are located very close to active faults, not all of those necessarily have the potential for inducing seismicity. In some cases injection may be into a depleted zone, in which years of oil production has reduced the pressure below its pre-drilling state, thus increasing the resistance to slip on faults in hydraulic connection with the reservoir (NRC, 2013). (Note that disposal into depleted reservoirs is distinct from reinjection of wastewater for enhanced oil recovery by water flooding; waterflood wells are listed separately in the DOGGR database.) In these cases, the potential for induced seismicity will not exist until the pressure buildup resulting from injection exceeds the original reservoir pressure. The DOGGR Online Well Record Search (DOGGR, 2014b) tool details the pool(s) into which each disposal well injects, so it should be possible to determine which wells inject into depleted zones by examining the production records for the same pool.

4.4.3.3. Preliminary Example of a Spatiotemporal Correlation Analysis

To analyze potential correlations of seismicity with water injection, we first identify clusters of earthquakes and then examine the relationships of the clusters to injection volumes and pressures. This is illustrated for the Santa Maria Basin in Figure 4.4-10. Figure 4.4-10a shows 1981–2011 Santa Maria Basin earthquake epicenters in the Hauksson (2012) catalog. To easily identify event clusters, each epicenter is color coded according to the slant distance (i.e., including event depth) of the event hypocenter to its nearest neighbor. Figure 4.4-10b shows the highly clustered seismicity contained in the green rectangle in 4.4.10a at expanded scale and the spatial relationship of the events to the locations of injection wells in the Zaca oilfield. Figures 4.4-10c and 4.4-10d compare the occurrence history of these 66 earthquakes with injected fluid volume and pressure histories for the four injection wells shown colored in Figure 4.4-10b. All of the events occurred between October 1984 and March 1987, and all but a few are clustered in two bursts of activity in October 1984 and October-November 1986. Both bursts include one event greater than M4.

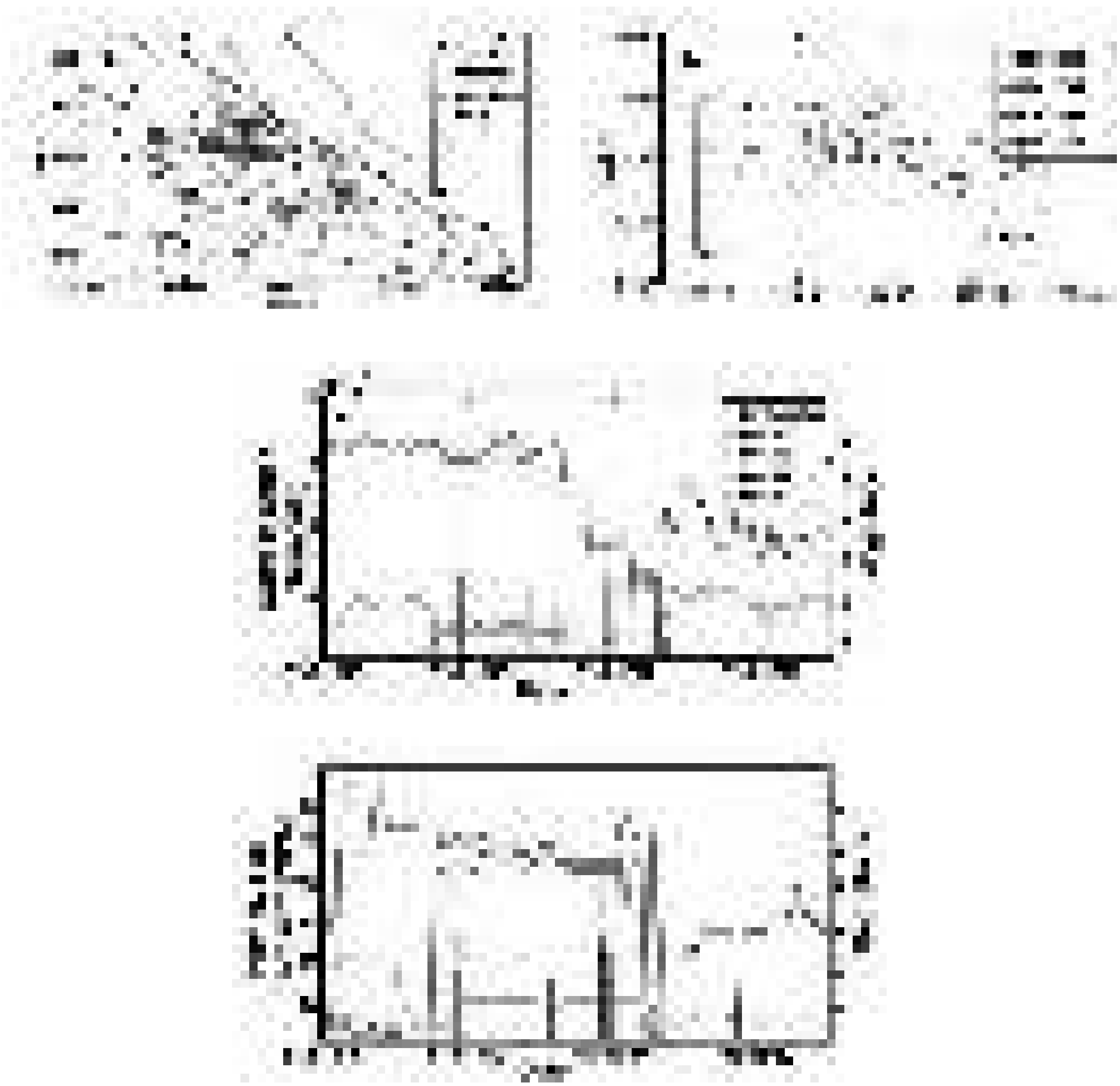

Figure 4.4-10. Spatiotemporal analysis of a seismicity cluster in the Santa Maria Basin. Earthquakes shown by solid circles in a and b are color-coded to show their closest slant distances to neighboring events. Wastewater injection wells are shown as triangles. Events and wells within the green rectangle in a are shown in b. Monthly injected volumes and wellhead pressure taken from the DOGGR (2014a) database for the four wells colored in b and identified by API number are plotted in c and d, respectively, along with the earthquakes in b shown in black.

The first burst of seismicity occurred about one month after the pressure in well 8301777 (magenta) reached its first peak following the abrupt increase in injection rate and pressure that began in June 1984, and also coincides with the beginning of a modest increase in pressure in well 8301784 (light green). These correlations suggest a relationship between the event sequence and the combined effect of the pressure increases in these two wells, which are the wells closest to the most densely clustered seismicity in Figure 4.4-10b. The second burst of activity was not associated with pressure changes apparent in the DOGGR database, but occurred shortly after a major decrease in injection rate in well 8304562 (dark green). In this case, no immediate correlation with changes in pressure in any of the wells is evident. However, all 66 earthquakes occurred during a period when the pressures in wells 8301777 and 8301784 (and also in well 8304562) were high, which further suggests a relationship between local seismicity and elevated fluid pressure. These evident relationships merit further detailed analysis that includes tests of statistical significance to investigate whether there is a causal link between the seismicity and pressure changes. Note that the flat portions of the pressure histories for the three wells mentioned above between May and December 1986 suggest missing data for this period, so that the pressure increase in well 8304562 (and in 8300260) evident after December 1986 may have begun earlier, perhaps following a pressure decrease sometime after May 1986.

This simple example demonstrates that analysis of the spatial and temporal relationship of earthquakes to wastewater injection has the potential to detect and characterize induced seismicity in California. However, the apparent gaps in the pressure data for several periods evident in Figure 4.4-10d, and the lack of depth information for any of the disposal wells in the Zaca field, illustrate two of the deficiencies in the present DOGGR well database that impede this kind of correlation analysis (see Section 4.6.1 below).

4.4.4. Potential for Induced Seismicity on the San Andreas Fault

The existing oilfields and disposal wells closest to the SAF are located just over 10 km (6.2 mi) away along the western margin of in the San Joaquin Valley (Figure 4.4-4). This is significantly greater than typical lateral well-fault distances of less than 3 km (1.9 mi) for the fluid injection-induced seismicity cases observed in the continental interior (see Section 4.4.3.2). It is similar to the 8 km (5 mi) distance between the Paradox Valley injection well and the fault that was the source of the 2013 M_w4.0 earthquake, but in that case a high-permeability pathway connects the well to the fault (Section 4.4.3.2). Therefore, while the possibility that current, relatively low-volume, wastewater injection in the San Joaquin Valley could induce earthquakes on the SAF cannot be entirely discounted, we judge that that it is unlikely.

Using the Paradox Valley case as a benchmark, it is plausible that the likelihood of triggering earthquakes on the SAF could increase if future high-volume wastewater injection took place in or close to existing disposal wells along the western margin of the San Joaquin Valley. Future injection projects that could potentially alter fluid pressures in the SAF or the other most active (high slip rate), major fault zones should be subject to particularly rigorous screening and permitting procedures, as described in the following section.

4.5. Impact Mitigation

Even if a comprehensive investigation of the relationship of seismicity to oilfield injection were to conclude that the overall potential for induced seismicity in California is low, it would be prudent to adopt measures to mitigate the risks from induced seismicity that may be associated with new stimulation-related injection projects. It will be particularly important to adopt such measures if there is an increase in stimulation activity and expanded production, resulting in higher per-well volumes of injected wastewater approaching those employed elsewhere in the U.S. In this section, we discuss measures that should be considered before injection begins to reduce the likelihood of induced earthquakes, and to manage seismicity during and following injection.

Initial, low-level hazard and risk assessment during site screening could be used to place each site into one of a few risk categories (e.g., low, moderate, high), based on the following recommended criteria:

- Planned injection rate, cumulative volume, duration, and depth.
- Distance from active or potentially active faults, and recency and rate of fault activity.
- Existence of potential high-permeability pathways between the well and faults
- Estimation of pressure changes on nearby faults.
- Background seismicity.
- Proximity to population centers and critical facilities.

Decisions regarding permitting and regulation of a site in one of the higher risk categories could then be based on a level of probabilistic seismic hazard and risk assessment determined to be appropriate for that category. The final permit would specify operating parameters such as maximum injection rate and pressure adjusted to achieve an acceptable level of risk. An important part of the permit would be specifications for monitoring requirements and operating procedures to manage and, if necessary, mitigate induced seismicity during injection, and perhaps for a period after the well is shut down. Methods for induced seismicity hazard and risk assessment and management are discussed in Section 4.5.1 below.

If future large-volume wastewater disposal were to be planned at sites along the western margin of the San Joaquin Valley, and especially if new injection locations closer to the SAF and other major active faults were contemplated, these wells should be subject to the most stringent risk assessment and permitting requirements. These should include detailed modeling to estimate the probability that the pressure changes on the fault over time would remain below a predetermined, conservative maximum bound.

4.5.1. Induced Seismic Hazard and Risk Assessment

Maps of seismic hazard from naturally occurring earthquakes in California are developed by the U.S. Geological Survey (USGS) and California Geological survey (CGS) as part of the National Seismic Hazard Mapping Project. The hazard maps and technical details of how they are produced can be found at http://earthquake.usgs.gov/hazards/index.php. Of the areas in which water disposal wells are currently active, seismic hazard from naturally occurring earthquakes is high in the Los Angeles, Ventura, Santa Maria, Salinas and Cuyama Basins and in the Santa Clarita Valley, and moderate to high along the western and southern flanks of the southern San Joaquin Valley. The hazard is moderate in the Bakersfield area and decreases towards the center and north of the San Joaquin Valley.[1]

Approaches to assessing induced seismicity hazard can be developed by adapting standard probabilistic seismic hazard assessment (PSHA) methods, such as those used by the USGS and CGS. The standard methods cannot be applied directly, however, because conventional PSHA usually is based only on mean long-term (100s to 1,000s of years) earthquake occurrence rates; i.e., earthquake occurrence is assumed to be time-independent. Induced seismicity, on the other hand, is strongly time- and space-dependent because it is dependent on the evolution of the pore pressure field, which must therefore be considered in estimating earthquake frequencies and spatial distributions.

Developing a rigorous PSHA method for short- and long-term hazards from induced seismicity presents a significant challenge. In particular, no satisfactory method of calculating the hazard at the planning and regulatory phases of a project is available at the present time; whereas in conventional PSHA earthquake frequency-magnitude statistics for a given region are derived from the record of past earthquakes, obviously no record of induced seismicity can exist prior to well stimulation or wastewater disposal. Using seismicity observed at an assumed “analog” site as a proxy (e.g., Cladouhos et al., 2012) would not appear to be a satisfactory approach, as induced seismicity is in general highly dependent on site-specific subsurface structure and rock properties.

1. Moderate and high seismic hazard are defined here as a 2% probability of exceeding peak ground accelerations of 0.1-0.3g and greater than 0.3g, respectively, in 50 years, where g is the acceleration due to gravity. The threshold of damaging ground motion is about 0.1g.

Physics-based approaches to generate simulated catalogs of induced seismicity at a given site for prescribed sets of injection parameters are under development (e.g., Foxall et al., 2013). Such approaches rely on adequate characterization of the site geology, hydrogeology, stress, and material properties, which are inevitably subject to significant uncertainties. However, large uncertainties in input parameters are inherent in PSHA in general, and techniques for propagating them to provide rigorous estimates of the uncertainty in the final hazard have been developed (e.g. Budnitz et al., 1997).

There has been more progress in developing methods for short-term hazard forecasting based on automated, near-real time empirical analysis of microseismicity recorded by a locally deployed seismic network once injection is under way (e.g., Bachmann et al., 2011; Mena et al., 2013; Shapiro et al., 2007). Continuously updated hazard assessments can form the input to a real-time mitigation procedure (Bachmann et al., 2011; Mena et al., 2013), as outlined in Section 4.5.2. Using two different time-dependent empirical models, Bachmann et al. (2011) and Mena et al. (2013) retrospectively were able to obtain acceptable overall fits of forecast to observed seismicity rates induced by the 2006 Enhanced Geothermal System (EGS) injection in Basel, Switzerland, over time periods ranging from 6 hours to 2 weeks. However, the models performed relatively poorly in forecasting the occurrence of the largest event (M_L3.4), which occurred after well shut-in; this event was forecast with a probability of only 15%, and the forecast probability of exceeding the ground motion it produced was calculated at only 5%. The performance of this empirical method could probably be improved by incorporating a more physically based dependence on injection rate or pressure.

4.5.2. Protocols and Best Practices to Reduce the Impact of Induced Seismicity

In 2004, the U.S. Department of Energy (DOE) and the International Energy Agency (IEA) sponsored an effort to develop a protocol and best practices to monitor, analyze, and manage induced seismicity at geothermal projects (Majer et al., 2007; 2012; 2014). The protocols/best practices are not intended to be either regulatory documents or universally prescribed sets of procedures for induced seismicity management, but rather to serve as a guide to enable stakeholders to tailor operating procedures to specific projects. Many geothermal operators in the western U.S. are implementing either all or parts of the most recent U.S. DOE protocol (Majer et al., 2012), and the U.S. Bureau of Land Management (BLM) has adopted it as the basis for developing criteria for geothermal project permitting on the federal lands administered by them.

Largely spurred by the dramatic increase in seismicity in the mid-continent discussed in Appendix 4.C, oil-producing states and the petroleum industry are beginning to develop similar protocols, such as those being developed by the Oklahoma Geological Survey and by a consortium of member companies in the American Exploration and Production Council (AXPC) (see Appendix 4.D). Zoback (2012) also describes a series of mitigation steps that operators could use as a guide. All of the protocols currently under development contain, in some combination, the steps that comprise the U.S. DOE geothermal protocol, described in Appendix 4.D.

Current real-time induced seismicity monitoring and mitigation strategies used by most enhanced geothermal system (EGS) operators employ a “traffic-light” system similar to the one implemented by Bommer et al. (2006). The traffic-light system may incorporate up to four stages of near-real time response to recorded seismicity, ranging from normal operation (green) to bleeding off to minimum wellhead pressure and shutting down the well (red). The response trigger criteria are generally based on some combination of maximum observed magnitude, measured peak ground velocity and public response, although definition of the criteria is usually somewhat ad hoc and depends on the project scenario. The traffic-light procedure implemented at the 2006 Basel EGS project was not successful in preventing the occurrence of the M_L3.4 earthquake that led to the eventual abandonment of the project, even though the well was shut down following an earlier M_L2.7 event. The EGS community is beginning development of traffic-light methods that employ near-real time hazard updating like that reported by Bachmann et al. (2011) and Mena et al. (2013). These will provide risk-based forecasting based on the evolving seismicity and state of the reservoir to inform decision-making.

4.6. Data Gaps

4.6.1. Injection Data

There are two important gaps in the current DOGGR (2014a) injection database that seriously limit its usefulness for investigating induced seismicity in California. First, injection rates and wellhead pressures are reported monthly. These are presumably monthly averages, since water disposal rates and pressures are rarely constant over month-long intervals. Significant short-term variations in peak pressures and injection rates are relevant to detecting the effects of fluid injection on seismicity in the vicinity of the well, in addition to long-term rates and cumulative volumes that can potentially impact seismicity on more distant faults. Therefore, monthly averages are usually too coarse to carry out correlation analyses against incremental increases in seismicity above the high seismic background in many areas of California.

The second data gap is consistent and accurate reporting of injection depth and geological interval. Currently, depth information of any kind is provided for less than 15% of active and plugged wastewater disposal wells in the database. Furthermore, currently available information is ambiguous because the parameter “WellDepthAmount” in the database can refer to injection depth, top or bottom of the perforation interval, or the total vertical depth of the well. Correlating injection depth with stratigraphy and the depth of seismicity has been shown to be critical in identifying induced events (e.g., Keranen et al., 2013).

Although it may be feasible to conduct spatiotemporal correlation analyses to identify and provide a basic characterization of more prominent cases of potentially induced seismicity using the current DOGGR (2014a) database, filling these two data gaps to some extent in the existing catalog would permit a much more comprehensive analysis. More complete reporting in the future would enable risk assessment and mitigation of induced seismicity for new stimulation-related injection operations.

4.6.2. Seismic Catalog Completeness

Although only earthquakes greater than about M2 are generally relevant to seismic hazard, M1 or even smaller earthquakes are important in analyzing potential induced seismicity. As discussed in Appendix 4.A, the estimated minimum magnitude of complete detection (M_c) of the USGS Advanced National Seismic System (ANSS) network is M1 or less in large areas of California, and less than M2 over most of the state. However, Figure 4A-1 shows that M_c is between 2 and 2.5 in the interior of the southern San Joaquin Valley and at some locations along the coast of southern California. Estimated mean, minimum and maximum M_c values in the main onshore oil-producing basins are summarized in Table 4.6-1; note that these values have not been adjusted to account for the tendency of the calculation method employed to underestimate M_c (see Appendix 4.A). Some wells in the southern San Joaquin Valley and the Los Angles and Ventura Basins are within areas having M_c2 or greater, so that microseismicity that may have been induced by injection into those wells might not have been recorded.

Ideally, a sensitive local seismic network comprising five or more seismic recording stations deployed at a spacing on the order of one kilometer or less is required to provide an adequate characterization of both the background activity and any induced seismicity at an injection site. Deploying sensors in deep boreholes is relatively expensive, but greatly enhances the signal-to-noise ratio, enabling very small earthquakes (often $M < 0$) to be recorded. While installation of a local network may not be feasible or necessary at many injection sites, it should be considered for sites in higher risk categories (Section 4.5).

4.6.3. Fault Detection

The USQFF fault inventory described in Section 4.4.1.2 contains the parameters of Quaternary-active faults in California. While it will be important to consider these faults in siting possible new injection operations, smaller local faults in the site vicinity will likely be of more direct relevance in assessing the potential for induced seismicity. These include faults having lengths on the order of 1 to 10 km (0.6–6.2 mi) capable of producing earthquakes between about M_w3.5 and 5, and even smaller ones that are potential sources of felt earthquakes. The fault inventory should also include inactive faults (i.e., activity predates the Quaternary) that are suitably oriented relative to the *in situ* stress field for shear failure. Both major and local faults that outcrop at the surface are shown on published geologic maps at scales as large as 1:24,000 (USGS 7.5 minute quadrangles). Unmapped faults on the kilometer scale, including buried structures, may be detectable in seismic and well data acquired during field exploration or characterization of specific injection sites. Faults on the 100-meter scale may be detectable depending on specific circumstances, but in general present a greater challenge. Finally, faults that are potential sources of induced earthquakes of concern and that escape detection during site characterization may often be illuminated by low-magnitude microearthquakes recorded during the initial stages of injection.

Table 4.6-1. Summary of minimum magnitudes of complete detection, M_c, in onshore oil-producing basins. M_c values not adjusted to accountt for underestimation bias (see Appendix 4.A).

Basin	Mean M_c±1s	Min M_c	Max M_c
Los Angeles	1.5±0.2	1.1	2.0
Ventura	1.5±0.3	0.8	2.1
Santa Maria	1.6±0.3	1.1	2.1
Cuyama	1.4±0.2	0.9	1.7
San Joaquin	1.6±0.3	0.6	2.0
Salinas	1.0±0.3	0.3	1.3

4.6.4. In-situ Stresses and Fluid Pressures

Although there are a large number of stress measurements in California compared with other regions of the U.S., the point measurements in the World Stress Map database provide only a sparse sampling of the stress field. While overall trends in Figure 4.4-1 appear relatively uniform, significant variations are to be expected because stress states at the local scale are influenced by heterogeneously distributed fractures of varying orientation and by changes in lithology and rock material properties (e.g., Finkbeiner et al., 1997). Ideally, stress measurements at a given injection site are needed to assess the potential for induced seismicity. To achieve this, it may be possible to employ other measurement techniques in addition to borehole data and analysis of hydraulic fracture breakdown and shut-in pressures. For example, in a hydraulic fracturing experiment in the Monterey formation, Shemeta et al. (1994) studied the geometry of the hydrofracture using continuously recorded microseismic data, regional stress information, and well logs. They found that the microseismic and well data were consistent with both the regional tectonic stress field and fracture orientations observed in core samples and microscanner and televiewer logs. The results of this study suggest that observations of the natural fracture system can be used as indicators for the orientations of induced fractures and hence of the *in situ* stress. As with local microseismic monitoring, *in situ* stress measurements may be justified only at higher-risk sites. However, measurement or estimation of stress orientations prior to well stimulation is critical for selecting a development well pattern and the design of hydraulic fractures for effective hydrocarbon recovery. Such measurements can be used to inform induced seismic hazard assessment for well stimulation activities within a field, and also for any nearby wastewater disposal operations.

4.7. Findings

The dramatic increase in the rate of earthquake occurrence that has accompanied the boom in unconventional oil and gas recovery in the central and eastern U.S. since 2009 has highlighted the fact that injecting fluids into the subsurface for well stimulation by hydraulic fracturing—and, in particular, for disposal of recovered fluids and produced wastewater—can cause induced seismicity. Induced seismicity can occur when fluid injection results in increased pore pressure within a fault. This reduces the force holding the two sides of the fault together, allowing the fault to slip.

Hydraulic fracture treatments inject relatively small volumes injected over short time periods. As a result, the subsurface volume affected by pressure perturbations is normally within hundreds of meters from the injection well, which, current experience suggests, limits the size of induced seismic events caused by well stimulation. To date, the largest event generally considered to have been caused by hydraulic fracturing is the 2011 M_L3.8 earthquake in the Horn River Basin in British Columbia (BC Oil and Gas Commission, 2012).

Injection of large volumes of wastewater over long time periods increases pressures over much larger distances than those resulting from hydraulic fracturing, which increases the likelihood of inducing larger seismic events. Therefore, injection of wastewater presents a much larger potential seismic hazard than hydraulic fracturing. The largest earthquake suspected of being related to wastewater disposal is the 2011 M_w5.7 Prague, Oklahoma event (Keranen et al., 2013; Sumy et al., 2014), but the causal mechanism of this event is still the subject of active research. The possibility that this was a naturally occurring tectonic earthquake cannot yet be confidently ruled out. The largest earthquake for which there is clear evidence for a causative link to stimulation-related wastewater injection is the 2011 M_w5.3 event in the Raton Basin, Colorado (Rubinstein et al., 2014).

The potential impacts from ground shaking caused by induced seismicity are structural damage—and possibly injuries and loss of life—and nuisance resulting from seismic events that are felt in nearby communities. While the vast majority of fluid injection-induced earthquakes are too small to be perceptible at the ground surface, some are strongly felt and on rare occasions can be large enough to cause damage (e.g., Kerenan et al., 2013; Rubinstein et al., 2014). The magnitude threshold for local structural damage is generally considered to be about M_w5, depending on the depth of the earthquake, surface site conditions, and the fragility of nearby structures. To date, the maximum magnitudes of earthquakes induced by hydraulic fracturing worldwide have been substantially below this threshold, which suggests that the likelihood of seismic damage resulting from hydraulic fracturing in general is very low.

The likelihood of damaging events resulting from wastewater disposal is much higher than that from hydraulic fracturing. Four earthquakes greater than M4.5 related to wastewater disposal have occurred in the U.S. since 2011 (see Appendix 4.C), of which the 2011 Prague and Raton Basin events mentioned above caused localized structural damage. However, given that induced seismicity has been associated with only a small fraction of the tens of thousands of injection wells currently or formerly active in the U.S., viewed in a global context the overall likelihood of a damaging event being induced by wastewater injection is low in absolute terms.

The magnitude threshold for felt events can be as low as M1.5–2.0 for the shallow depths of seismicity that are typically associated with fluid injection. There are only five documented cases of seismicity related to hydraulic fracturing worldwide that included felt events, but numerous cases related to wastewater injection. Because, in general, the rate of earthquake occurrence increases by about a factor of ten for every decrease of one magnitude unit, the overall likelihood of nuisance from wastewater injection-induced earthquakes is relatively high.

4.8. Conclusions

Although induced seismicity occurs at several geothermal fields in California, there have been no published reports of felt seismicity linked to either hydraulic fracturing or wastewater disposal in the state, apart from one highly anomalous event reported by Kanamori and Hauksson (1992). However, in many areas of California, discriminating induced events in the M2-4 range from frequently occurring natural events is difficult, and the systematic studies necessary have begun only recently.

The lack of reported felt seismicity related to hydraulic fracturing is consistent with injection into predominantly vertical wells at relatively shallow depths in California and the small injection volumes currently employed. Therefore, based on experience elsewhere, hydraulic fracturing as currently carried out in California is not considered to pose a high seismic risk.

The total volume of wastewater injected in California is much larger than the volume used for well stimulation, but current volumes are relatively small compared to the regions in the U.S. that have recently experienced large increases in induced seismic activity related to wastewater disposal. Although this might imply a lower current potential for induced seismicity than in the mid-continent, the relationship between seismicity and wastewater injection in California has not been fully evaluated. Therefore, the potential level of seismic hazard posed by wastewater disposal is at present uncertain. A comprehensive, in-depth study of spatial and temporal correlations, if any, between wastewater injection and seismicity will be required to provide a firm basis for assessment of seismic hazard related to induced seismicity.

As evidenced by the upswing in induced seismicity in the central and eastern U.S. since 2010, an increase in hydraulic fracturing activity and expanded production in California could increase the seismic hazard from wastewater disposal and perhaps also from hydraulic fracturing, particularly if they involve higher per-well injected volumes approaching those employed elsewhere in the U.S. and a shift to deeper stimulation. However, based on the data presented in Volume I of this study, such shifts in well stimulation in California are not expected in the near or mid term.

The closest wastewater disposal wells to the SAF are located in oilfields just over 10 km (6.2 mi) away in the southern San Joaquin Valley. It is unlikely that current wastewater injection in these wells would induce earthquakes on the fault. If future high-volume injection took place in or close to these existing oilfields, it is plausible that the likelihood of triggering earthquakes on the SAF could increase.

Even if the overall potential for induced seismicity in California proves to be low, some level of incremental seismic hazard and risk assessment to inform permitting and regulation of stimulation-related injection projects is justified. Initial low-level assessment during site screening could be used to place each site into one of a few risk categories, based on planned injection rate, cumulative volume and depth, distance from active or potentially active faults, estimated pressure changes on those faults, background seismicity, and proximity to population centers and critical facilities. An appropriate level of probabilistic seismic hazard and risk assessment would then be carried out for sites in higher risk categories, and the permit would specify bounds on injection parameters to achieve an acceptable level of risk. For these sites, monitoring requirements and operating procedures to manage and, if necessary, mitigate induced seismicity during injections would also be specified.

Injection projects that could possibly cause significant pressure changes on the most active major faults like the SAF should be subject to the most stringent risk assessment and regulatory requirements.

The mechanics of fluid-induced seismicity are fairly well understood, and, as such, it is theoretically possible to carry out full hazard assessments at higher-risk sites. However, much more detailed information on injection than is currently available in publicly available databases will be required, first to gain an understanding of the potential for induced seismicity in California oil-producing basins, and then to carry out hazard and risk assessments. Site-specific investigations will also require definition of local faults, the state of stress on those faults, characterization of rock, fault, and hydrological properties, measurement or modeling of the subsurface pressure perturbation based on injection rates, and characterization of the seismicity at the site and in the surrounding area.

Two aspects of the current DOGGR (2014a) database limit its usefulness in identifying past induced seismicity. First, injected volume rates and wellhead pressures are reported only as (presumed) monthly averages, whereas peak volumes, rates and pressures, and significant short-term variations are of relevance in detecting effects on seismicity. Secondly, depth information is not available for the majority of wastewater injection wells. Filling these gaps in the existing database would facilitate a much more comprehensive analysis of the correlations of injection with seismicity. More complete reporting in the future would enable hazard and risk assessment and mitigation of induced seismicity for new stimulation-related injection operations.

Adequate characterization of local seismicity requires recording of local microearthquakes as small as about M1 or less. Existing regional and local networks provide this detection capability in some areas of California, but the threshold for complete detection in some oil-producing basins is M2 or higher, which presents an obstacle to discriminating potential past induced seismicity in these areas. Moving forward, local microearthquake networks should ideally be installed to monitor seismicity at higher-risk sites located in areas currently having detection thresholds higher than about M1.

The current compilation of stress data for California provides only a sparse sampling of the regional *in situ* field in most areas within oil-producing basins. Therefore, detailed analysis of the potential for induced seismicity and seismic hazard assessment at higher-risk sites would ideally utilize site-specific stress measurements obtained from borehole data and other techniques. Similarly, detecting faults that are potential sources of felt or perhaps damaging induced earthquakes will require site-specific characterization to augment the existing active fault database and geologic maps. Advanced detection of faults on the 100 m to 1 km (328 to 3280 ft) scale that may be sources of small felt earthquakes presents a particular challenge, but these may be revealed by low-magnitude microseismicity during the initial stage of injection.

Inevitably, many of the parameters needed for induced seismicity hazard calculations will be poorly constrained. However, seismic hazard assessment in general is invariably subject to considerable uncertainty, and an important and mature part of the PSHA procedure is to properly characterize the uncertainties in the input parameters, and then propagate them through the calculation to provide rigorous uncertainty bounds on the final hazard estimates.

Induced seismicity that could potentially accompany an increase in well stimulation activity in California could likely be managed and mitigated by adopting a protocol similar to the one developed by the U.S. DOE for enhanced geothermal systems. In addition to hazard and risk assessment, one of the core recommendations in the U.S. DOE protocol is provision of a set of procedures to modify an injection operation in response observed changes in seismicity. These entail staged reduction in injection flow rate and pressure up to and including well shutdown. The procedures should be based on quantitative forecasts of the probability of inducing earthquakes of concern derived from observations of evolving seismicity and changes to the state of the reservoir, rather than the essentially ad hoc criteria that have been employed to date.

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Chapter Five

Potential Impacts of Well Stimulation on Wildlife and Vegetation

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5.1. Abstract

In this chapter, we examine the impact of well stimulation on California's wildlife and vegetation. Potential impacts to wildlife and vegetation from oil and gas operations using well stimulation considered in this chapter are: (1) habitat loss and fragmentation, (2) introduction of invasive species, (3) releases of harmful fluids to the environment, (4) diversion of water from waterways, (5) noise and light pollution, (6) vehicle collisions, and (7) ingestion of litter by wildlife.

In this chapter we focus on habitat loss and fragmentation, because it was the only impact for which we had sufficient data to quantify impacts, and because our analysis indicates that habitat loss and fragmentation caused by production enabled by hydraulic fracturing is large enough to be of concern for habitat conservation in Kern and Ventura counties.

The degree to which hydrocarbon production and natural habitat come into contact depends on two major factors: (i) the density of oil and gas production infrastructure, and (ii) other human land uses in the area. Areas dominated by near-continuous well pads are largely inhospitable to native wildlife and vegetation. In other places, oil and gas production, including operations that use well stimulation, is interspersed with agricultural and urban development that has already displaced native habitat. In contrast, large portions of some oil fields have little other development and a relatively low density of oil wells. Native species inhabit the areas in and around these oil fields.

In areas where there is natural habitat, new oil and gas development impacts native species via a variety of mechanisms, the most well-understood of which is habitat loss and fragmentation. New wells bring new well pads, new roads, more vehicle traffic, and

other human activities that alter open land in ways that can make it uninhabitable to most wildlife and vegetation. In California, most hydraulic-fracturing-enabled-development takes place in and around areas that were already producing oil and gas without the application of well stimulation. Well stimulation, in particular hydraulic fracturing, has enabled an increased density of oilfield development and alight increases in the footprint of developed areas. Our analysis of habitat types, vegetation cover, well density and well stimulation activity in California indicates that impacts of well stimulation to wildlife and vegetation are most pronounced in the southwest portion of the San Joaquin Basin and the transverse ranges in the Ventura basin.

Aside from habitat loss and fragmentation, we are unable to quantify the impacts of well stimulation on wildlife and vegetation in California using available data, and we restrict our discussion of them to general description and literature review.

We also discuss the relevant rules and regulations governing impacts to wildlife and vegetation from oil and gas activities. Although regulations exist to evaluate and mitigate site- or project-specific impacts when new oil and gas development is proposed, the agencies of jurisdiction have not routinely evaluated the incremental impacts of individual oil and gas development projects within the larger context of habitat loss and fragmentation at the regional level. We also discuss the most commonly implemented best practices and mitigation measures. We conclude with a discussion of important data gaps, particularly a lack of information to more precisely quantify impacts of well stimulation on population growth rates of species, a poor understanding of the degree to which abandoned oil and gas leases can be restored, and a lack of studies evaluating the efficacy of best practices and mitigation measures.

5.2. Introduction

There are a number of potential ways that well stimulation can affect wildlife and vegetation. In this chapter we discuss potential impacts due to: (1) loss and fragmentation of habitat, (2) introduction of invasive species, (3) contamination of the aquatic environment, (4) diversion of water from waterways, (5) noise and light pollution, (6) vehicle traffic, and (7) ingestion of litter. Most of these impacts are not directly caused by the process of well stimulation, but are common to any form of oil or gas production.

Many of the impacts to wildlife and vegetation require an intermediary such as water use or contamination, light and noise pollution, or increases in traffic that are discussed in other chapters in this volume: water use or contamination in Chapter 2, and noise, light and traffic in Chapter 6. This chapter examines these topics with an eye to their potential effect on wildlife and vegetation. We also explore the following potential impacts that are not discussed elsewhere in Volume II: habitat loss, introduction of invasive species, and ingestion of litter by wildlife. We focus most of our quantitative analysis on the impact of well-stimulation-enabled hydrocarbon production on habitat loss for three reasons. First, of the seven potential impacts listed in Table 5.2.1, habitat loss was the only impact with sufficient data available to conduct a statewide quantitative assessment. Second, habitat loss is a well-documented impact of oil and gas development in the terrestrial environment (Weller et al., 2002; Northrup, 2013). Last, habitat loss is generally regarded as the leading cause of biodiversity loss on the planet, followed by invasive species, pollution, and commercial exploitation (Moyle and Leidy, 1992; Wilcove et al., 1998). Closely related to habitat loss is fragmentation. The general principle behind habitat fragmentation is that the configuration as well as the quantity of habitat remaining affects the survival of species. Habitat fragmentation is not discussed in depth here, but is discussed in the San Joaquin Case Study in Volume III.

We note whether impacts are direct or indirect throughout the chapter. Direct impacts are uniquely associated with well stimulation and do not occur when oil and gas are produced without the aid of well stimulation. Examples of direct impacts of well stimulation include a spill of stimulation chemicals, or noise generated by equipment used in hydraulic fracturing. Indirect impacts stem from other aspects of the oil and gas production process apart from well stimulation. Examples of indirect impacts include the construction of a well pad and other infrastructure necessary for oil and gas production (resulting in habitat loss), and disposal of produced water (which can contaminate habitat). If these impacts are incurred by a well that is only economical to produce with the enabling technology of hydraulic fracturing, then they are indirect impacts. In other words, a proportion (but not all) of the indirect impacts to wildlife and vegetation caused by oil and gas production are enabled by hydraulic fracturing, since certain low-permeability reservoirs are not economical to produce without the technology. Matrix acdizing and hydraulic fracturing are not important drivers of increased production in California.

Habitat loss and fragmentation, introduction of invasive species, and litter are indirect impacts of hydraulic fracturing: they are not caused uniquely by hydraulic fracturing, but by expanded development and production allowed by hydraulic fracturing. Contamination of the aquatic environment, diversion of water from waterways, noise and light pollution, and vehicle traffic can be direct or indirect impacts, depending on context – for example, a spill of stimulation chemicals would be directly attributable to well stimulation, whereas a spill of produced water would be an indirect impact. The distinction between direct and indirect impacts is important because it has policy implications. Banning hydraulic fracturing would eliminate direct impacts. It would reduce indirect impacts, but not eliminate them, since indirect impacts are also caused by other forms of oil and gas production. For a more detailed discussion of direct and indirect impacts, please see the Summary Report.

Volume I of the report found that hydraulic fracturing is an important driver of expanded production in the state, whereas acid stimulations are not (Volume I, Chapter 1, Finding 5). Consequently, hydraulic fracturing is the only well-stimulation technology driving expanded hydrocarbon production in the state and thereby causing indirect impacts such as habitat loss and fragmentation. We discuss well stimulation as a whole, including acid stimulations, when addressing direct impacts, such as potential releases of stimulation fluids to the environment.

5.2.1. Overview of Chapter Contents

This chapter covers five major topics. In Section 5.2, the Introduction, we describe the ecology of Kern and Ventura counties, the two regions where we found major impacts from hydraulic fracturing-enabled production. We also describe land use patterns within the administrative boundaries of oil fields. In Section 5.3, "Assessment of Well Stimulation Impacts to Wildlife and Vegetation," we describe how well stimulation can impact wildlife and vegetation in California. Each potential impact is defined and relevant literature is reviewed. Whenever possible we discuss studies conducted in California, although most of the available work was not peer reviewed, and the majority focus on one region in the San Joaquin Valley. Because habitat loss and fragmentation is likely to have the greatest impact on wildlife and vegetation, we explore this topic in greater depth by quantifying habitat loss and fragmentation attributable to well-stimulation-enabled hydrocarbon production. We also summarize the potential future impacts to wildlife and vegetation. In Section 5.4, we describe how oil and gas production activities are regulated with respect to their impacts on wildlife and vegetation. In Section 5.5, we discuss measures to mitigate oil field impacts on terrestrial species and their habitats. In Section 5.6 we assess major data gaps and ways to remedy the gaps. In Sections 5.7 and 5.8, we summarize the major findings and conclusions of the chapter.

5.2.2. Regional Focus: Kern and Ventura Counties

In our analysis, we focused on the areas in the state where substantial amounts of well stimulation occurred in the context of undeveloped areas of natural habitat. We evaluated the ecological impacts of hydraulic-fracturing-enabled development with respect to the impact to loss of natural habitat, the rarity of that habitat statewide, and occurences of endangered species and designated critical habitat in the vicinity. Two regions emerged as locations where hydraulic-fracturing-enabled development was heavily impacting natural habitat. The first was southwest Kern County in the vicinity of Elk Hills, North and South Belridge, Buena Vista, and Lost Hills Fields. The second key region was along the southern perimeter of Los Padres National Forest in Ventura County, in the Ojai and Sespe Fields, within the Santa Barbara-Ventura Basin (referred to for brevity as the Ventura Basin). Matrix acidizing is much rarer and tends to be concentrated in southwestern Kern county. As a result, we focus our discussion primarily on Kern County, and secondarily on Ventura County, followed by other counties in the state.

5.2.2.1. Kern County: Ecology, Oil and Gas Development, and Well Stimulation

Kern County lies in the southern portion of the San Joaquin Valley, which was a region once dominated by lakes, wetlands, riparian corridors, valley saltbush scrub, and native grasslands. Most of the natural habitat has been converted to agricultural or urban use since the mid-19th century (Figure 5.2-1). Owing primarily to loss of habitat, there are approximately 143 federally-listed species, candidates and species of concern[1] with distributions wholly or partially in the San Joaquin Valley (Williams et al., 1998). For comparison, there were 568 state and federally listed and candidate species in California as of 2015 (Biogeographic Data Branch DFW, 2015a; b). The majority (76%) of California's remaining valley saltbush scrub habitat and its associated endangered species persists in southwestern Kern County. This area also has major petroleum resources. As a result, forty-two percent of California's remaining valley saltbush scrub habitat is within the boundaries of a Kern County oil field (Appendix 5.D, Table 5.D-1). The relationship is not entirely coincidental. The giant oil fields of the southwestern San Joaquin Valley such as Midway-Sunset, North and South Belridge, Elk Hills, Buena Vista and Lost Hills were discovered between 1894 and 1912 and were controlled by oil development interests before agriculture dominated the region. Within large portions of those oil fields, development is sparse enough that native habitat, principally valley saltbush scrub and non-native grassland, persists. Very little of the original aquatic and wetland habitats of the San Joaquin Valley remain, with more than 90% of open water, wetlands, and riparian habitat converted to farmland and cities (Kelly et al., 2005).

1. "Federally-listed" refers to species listed as endangered or threatened under the Endangered Species Act. "Candidate species" are organisms for which the U.S. Fish and Wildlife Service has sufficient information on their biological status and threats to propose them as endangered or threatened under the Endangered Species Act, but for which development of a proposed listing regulation is precluded by other higher priority listing activities. "Species of concern" are deemed to be potentially in decline, but are not presently candidates for listing.

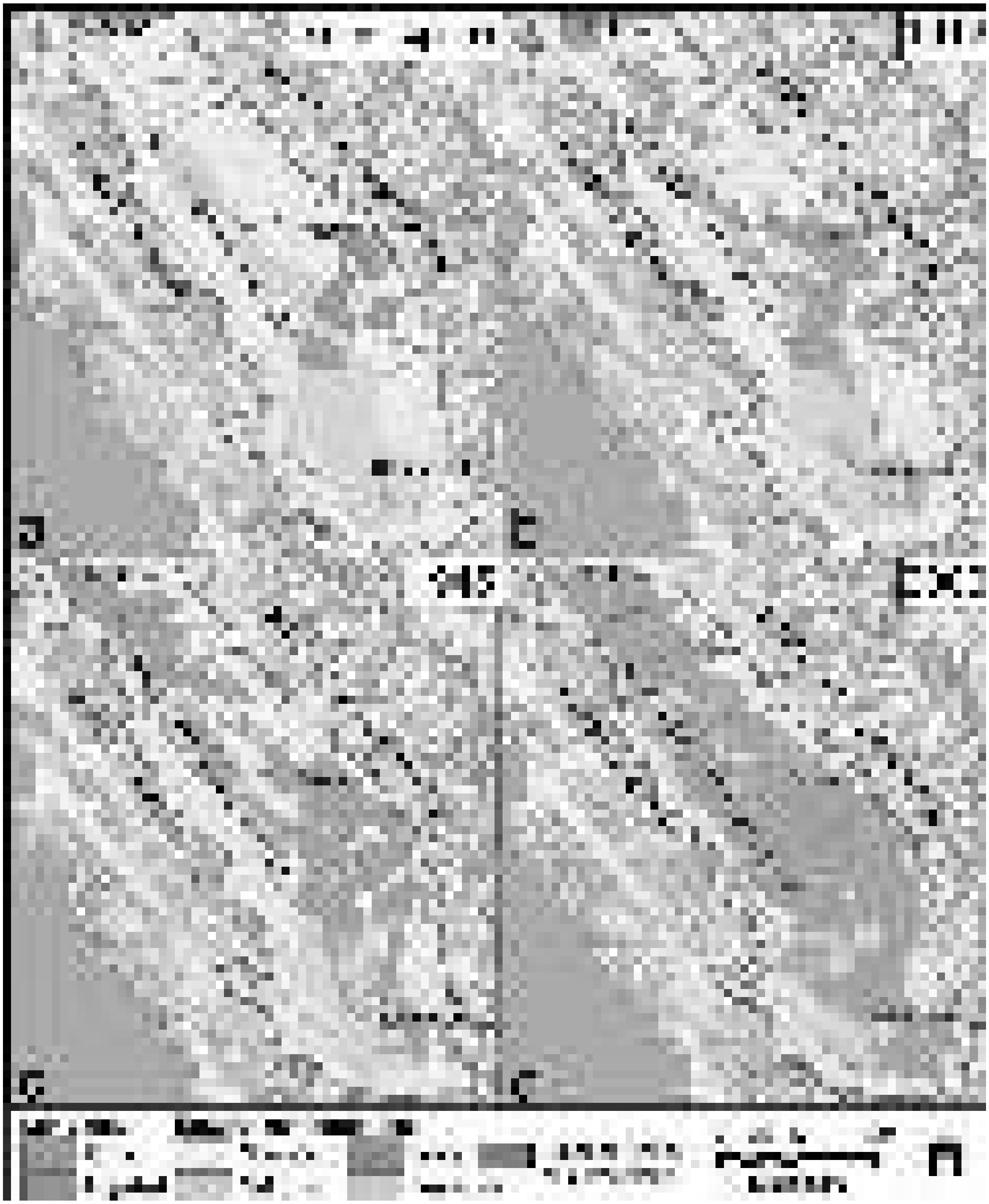

Figure 5.2.1. Maps of the San Joaquin Valley from pre-European settlement to the year 2000. The majority of natural habitat in the region has been converted to human use, principally agriculture, over the past century. The bulk of remaining valley saltbush scrub habitat is in the southwestern San Joaquin, where a combination of hillier terrain and ownership by oil developers prevented conversion to agriculture. Reprinted with permission from Kelly et al. (2005).

Kern County has the highest density of hydraulic fracturing and matrix acidizing in the state. More than 85% of hydraulic fracturing in the state occurs in six fields in southwestern Kern County: North and South Belridge, Elk Hills, Lost Hills, Buena Vista, and Midway-Sunset (Volume I Section 3.2.3.2, “Location).” More than 95% of matrix acidizing occurs in three fields in the same region: Elk Hills, Buena Vista, and Railroad Gap (Summary Report).

5.2.2.2. Ventura County: Ecology, Oil and Gas Development, and Well Stimulation

Ventura County is dominated by chaparral and Venturan coastal sage scrub with some dispersed riparian and annual grassland areas. The southern portion of the county has largely been converted to urban and agricultural use, while the northern half overlaps with Los Padres National Forest. Because much of southern California has been so heavily altered by human use, the national forest serves as an important refuge for species extirpated elsewhere in the region. It provides habitat for 468 permanent or transitory species of fish and wildlife, over 100 of which are listed as federally- or state-endangered, threatened, or sensitive[2] (CDFW, 2014a; 2014b; USFWS, 2014b). Listed species in the region include the vernal pool fairy shrimp, the Southern willow flycatcher, the California red legged frog, the California condor, southern steelhead, Least Bell’s Vireo, and the Santa Ana sucker. Typical habitat types are buck brush chaparral, chamise chaparral, and Venturan coastal sage scrub (UCSB Biogeography Lab, 1998).

While the total number of wells and hydraulic fracturing is much lower in Ventura than Kern County, a high proportion of the activity was enabled by hydraulic fracturing in eleven oil fields in the Ventura Basin (Volume I, Appendix N). Two fields, the Ojai and the Sespe, fall at least partially within the Los Padres National Forest and abut the Sespe Wilderness, home to the Sespe Condor Sanctuary. The Sespe Oil Field is also adjacent to the Hopper Mountain National Wildlife Refuge.

5.2.2.3. The Ecology of Kern and Ventura County Oil Fields

There is a common misperception that there is little or no natural habitat in areas developed for oil and gas production. In fact, oil and gas production, including operations that use well stimulation, is often interspersed with natural habitat (Fiehler and Cypher, 2011; Spiegel, 1996). As a result, native biota, including listed species, can be found in and around some areas developed for oil and gas, notably in Kern and Ventura Counties (USFWS, 2005; Fiehler and Cypher, 2011). However, other oil fields are dominated by human land uses to the exclusion of natural habitat.

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2. Sensitive plants include those plants listed as endangered, threatened or rare (Section 670.2, Title 14, California Code of Regulations; Section 1900, Fish and Game Code; ESA Section 17.11, Title 50, Code of Federal Regulations) or those meeting the definitions of rare or endangered provided in Section 15380 of the CEQA Guidelines.

The degree to which natural habitat persists on oil fields depends primarily on two factors: (i) the density of oil and gas production infrastructure, and (ii) other human land uses in the area. Areas dominated by near-continuous well pads, such as large expanses of the North and South Belridge, Lost Hills, and Ventura Oil Fields, are largely inhospitable to native wildlife and vegetation (Fiehler and Cypher, 2011 and **Figure 5.2.2**a). In other places, oil and gas production is interspersed with agriculture and urban development that by themselves displace the native habitat. Oil fields such as Rose and North Shafter are dominated by agriculture and urban development with scattered oil wells; there is virtually no intact natural habitat remaining in those regions, so oil development in those areas has little impact on wild animals and vegetation (**Figure 5.2.2**b).

In contrast, large portions of oil fields such as Elk Hills, Lost Hills and Buena Vista in Kern County and Ventura, Ojai and Sespe in Ventura are otherwise unimpacted by human development and have a relatively low density of oil wells (**Figure 5.2.2**c). Native species can survive on and around these oil fields. For example, outside of the Carrizo Plain Natural Area in San Luis Obispo County, the largest extant populations of the federally endangered/state threatened San Joaquin kit foxes are in the Elk Hills and Buena Vista oil fields in Kern County (USFWS, 2005). **Figure 5.2.3** and **Figure 5.2.4** depict areas of varying well density and land use in the southern San Joaquin Valley and Ventura County. Areas denoted as having medium or low well density that are not developed for human use are areas where habitat interacts with oil and gas production.

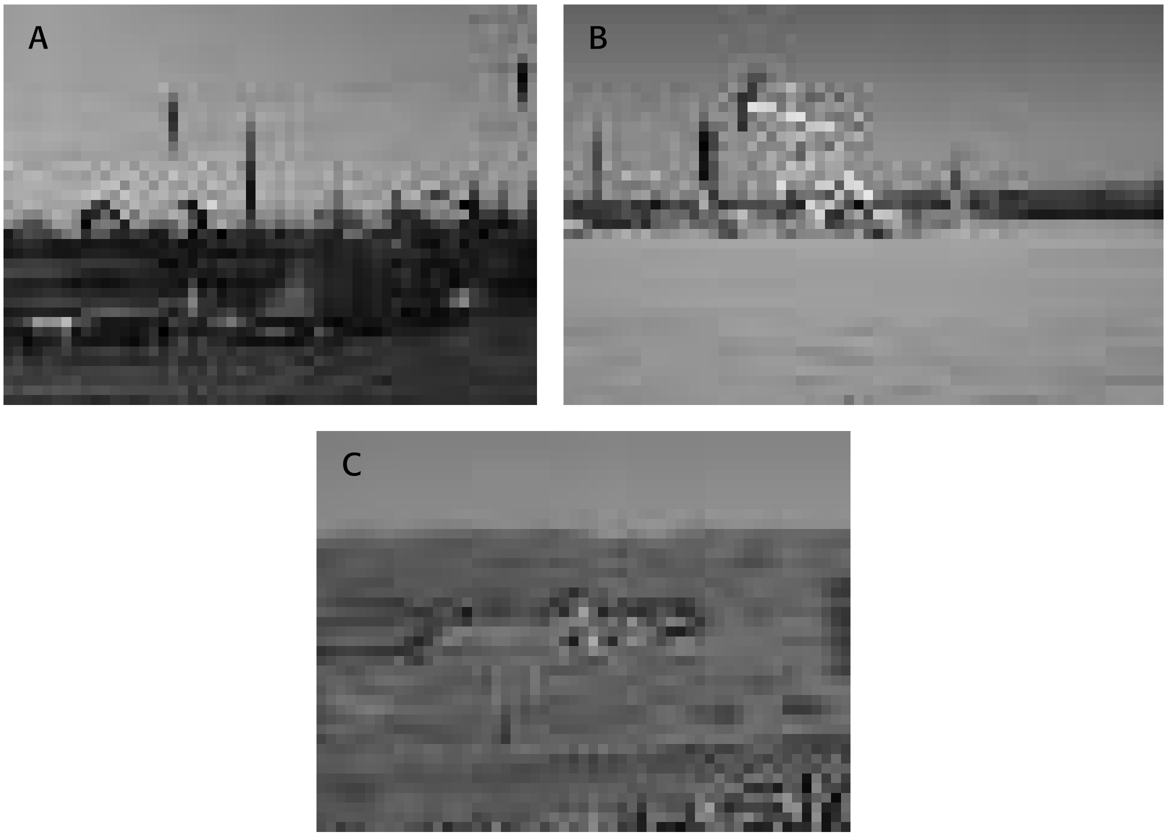
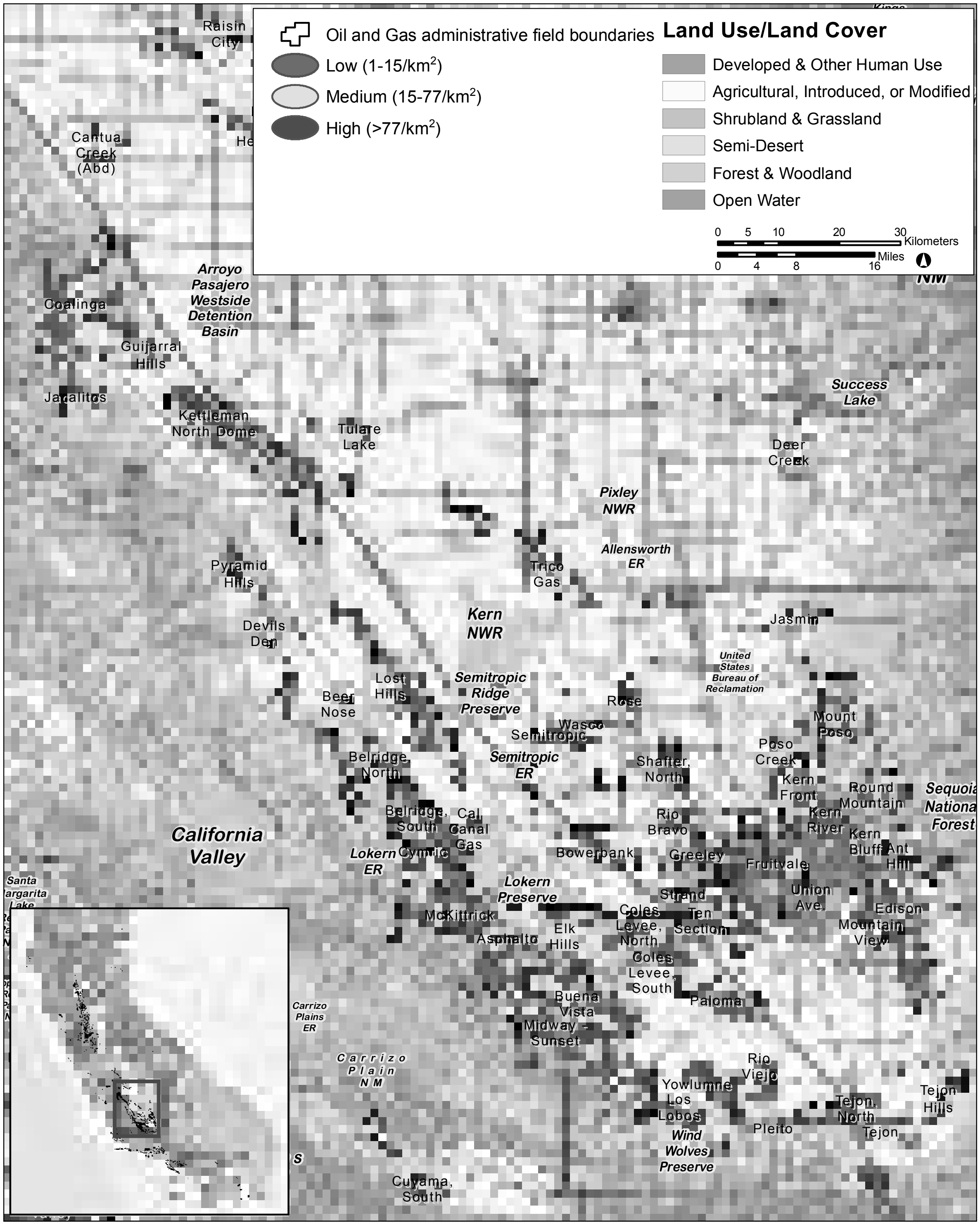


Figure 5.2.2. (a) An area of high well density at Lost Hills field is largely inhospitable to the native biota. (b) Pump jacks in the North Shafter field are surrounding by a fallow field and an orchard; there is little or no native habitat. (c) The Elk Hills Oil Field in Kern County has areas of low well density surrounded by large areas of intact valley saltbush scrub vegetation, habitat for a number of threatened and endangered native species. While well stimulation takes place in all three fields, activities in areas surrounded by native habitat are more likely to have ecological impacts. Photo credits: (a) C. Varadharajan, (b) L. Feinstein, (c) C. Varadharajan, 2014.



Oil and Gas administrative field boundaries
Low (1-15/km²)
Medium (15-77/km²)
High (>77/km²)
Land Use/Land Cover
Developed & Other Human Use
Agricultural, Introduced, or Modified
Shrubland & Grassland
Semi-Desert
Forest & Woodland
Open Water
0 5 10 20 30 Kilometers
0 4 8 16 Miles
Raisin City
Cantua Creek (Abd)
Coalinga
Arroyo Pasajero Westside Detention Basin
Guijarral Hills
Jacalitos
Kettleman North Dome
Tulare Lake
Success Lake
Deer Creek
Pixley NWR
Allensworth ER
Pyramid Hills
Trico Gas
Devils Den
Kern NWR
Jasmin
United States Bureau of Reclamation
Lost Hills
Beer Nose
Semitropic Ridge Preserve
Rose
Wasco
Semitropic
Mount Poso
Semitropic ER
Belridge North
Shafter, North
Poso Creek
Kern Front
Round Mountain
Sequoia National Forest
Belridge, South
Cal Canal Gas
Rio Bravo
Kern River
Kern Bluff
Ant Hill
California Valley
Lokern ER
Cymric
Bowerbank
Greeley
Fruitvale
Lokern Preserve
Union Ave.
Strand
Santa Margarita Lake
McKittrick
Coles Levee, North
Ten Section
Edison
Asphalto
Elk Hills
Mountain View
Coles Levee, South
Buena Vista
Paloma
Carrizo Plains ER
Midway - Sunset
Carrizo Plain NM
Rio Viejo
Yowlumne
Los Lobos
Tejon Hills
Tejon, North
Pleito
Tejon
Wind Wolves Preserve
Cuyama, South

Figure 5.2.3. Well density in the southern San Joaquin (and Cuyama) basins. Opaque blue, yellow and red indicate the density of wells, both stimulated and unstimulated; all wells that had recorded activity recorded activity from January 1977 through September 2014 are shown. Background shading indicates land use and cover categories. Larger versions of these maps, and maps of other basins, can be found in Appendix 5.B. Data from California Division of Oil, Gas and Geothermal Resources (DOGGR), 2014a; 2014b; 2014c; UCSB Biogeography Lab, 1998; California DOC, 2012.

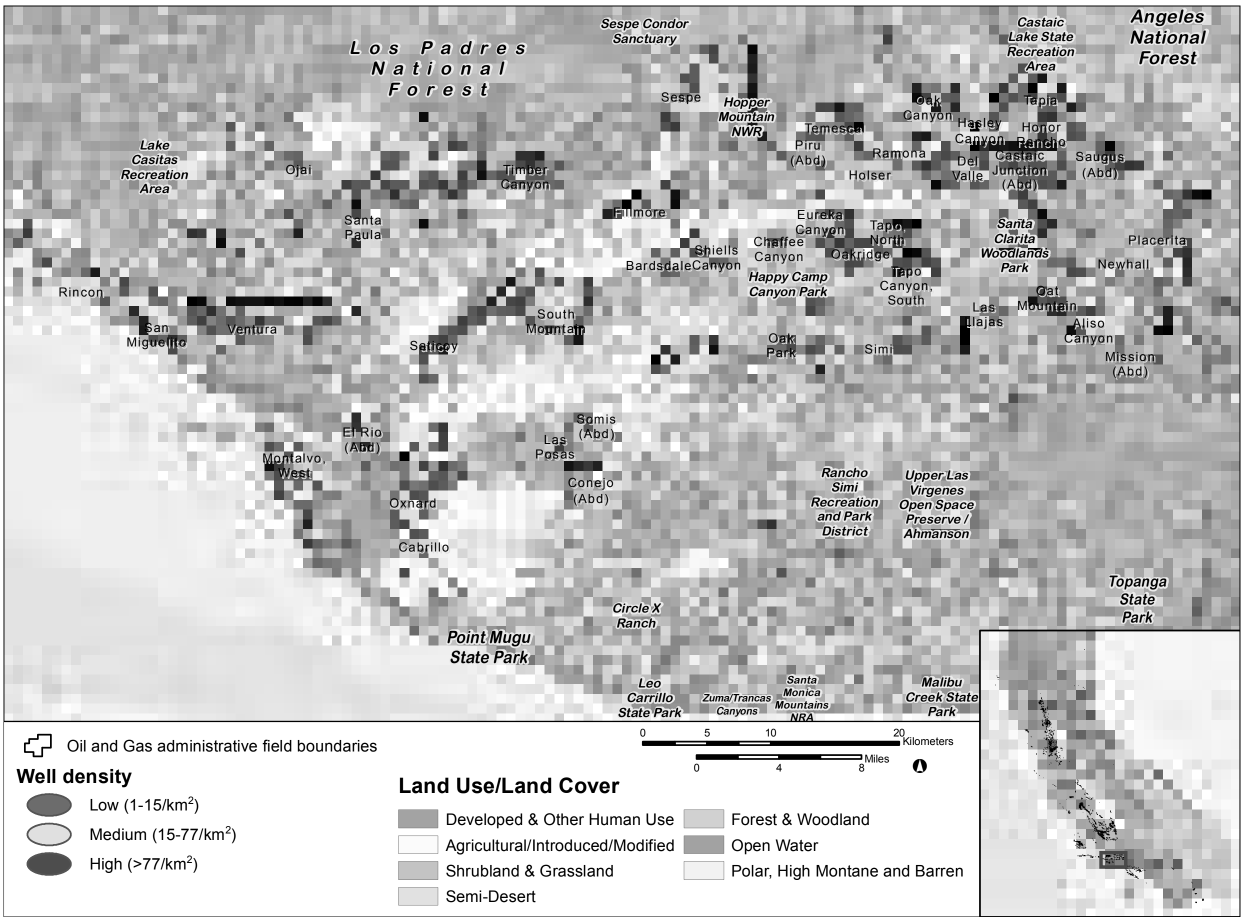


Figure 5.2.4. Well density in Ventura Basin. Opaque blue, yellow and red indicate the density of wells, both stimulated and unstimulated; all wells that had recorded activity recorded activity from January 1977 through September 2014 are shown. Background shading indicates land use and cover categories. Larger versions of these maps, and maps of other basins, can be found in Appendix 5.B. Data from DOGGR, 2014a; 2014b; 2014c; UCSB Biogeography Lab, 1998; California DOC, 2012.

5.3. Assessment of Well Stimulation Impacts to Wildlife and Vegetation

In this section we describe the following ways that well stimulation can impact wildlife and vegetation: habitat loss and fragmentation, facilitating invasive species, discharging potentially harmful fluids, use of water, noise and light pollution, traffic, and litter. Because we expect habitat loss and fragmentation to have the greatest effect on wildlife and vegetation, and adequate data was available, we conduct an original quantitative analysis on the topic, in which we identify the areas where well stimulation has had the greatest impact, how much of various habitat types were affected, and describe in detail the special-status species that occur in the vicinity.

5.3.1. Land Disturbance Causes Habitat Loss and Fragmentation

5.3.1.1. Overview and Literature Review of Habitat Loss and Fragmentation

Oil and gas production contribute to habitat loss and fragmentation through the construction of well pads and support infrastructure and related land disturbance, not directly by hydraulic fracturing itself (Jones and Pejchar, 2013). Expanding production of unconventional resources in new areas, often in areas of open habitat relatively unaffected by people, is resulting in habitat loss and fragmentation in areas such as Canada (Council of Canadian Academies, 2014), Wyoming (Thomson et al., 2005), Colorado (Jones and Pejchar, 2013), and Pennsylvania (Johnson et al., 2010). Unlike California, in regions where the only hydrocarbons produced are from source rock, all oil and gas production is indirectly attributable to hydraulic fracturing. For example, Pennsylvania's Marcellus Shale is only producible with hydraulic fracturing, and it underlies valuable forest and freshwater habitat. In regions outside of California, there are a number of locations where hydraulic fracturing enables production in areas never before developed for oil and gas. When these areas happen to underlie areas of relatively pristine habitat, the oil and gas production enabled by hydraulic fracturing causes habitat loss and fragmentation (Slonecker et al., 2013; Roig-Silva et al., 2013; Johnson et al., 2010).

In California, it is difficult to isolate the impact of hydraulic fracturing on habitat from the impacts of oil and gas production in general. This is because most hydraulic fracturing is occurring on lands that would be used for oil and gas production regardless of hydraulic fracturing. This is because hydraulic fracturing is necessary for production from certain types of low-permeability reservoirs. In many places in California, these low-permeability reservoirs are stacked vertically with reservoirs that do not require hydraulic fracturing. As a result, at the land's surface, wells that are hydraulically fractured are interspersed with wells that are not, because they are tapping different vertical layers of rock with different geologic properties.

Roughly half of the wells installed in California in the past decade were hydraulically fractured, and about one in fifteen were acidized; 85% of this activity is the North Belridge, South Belridge, Elk Hills and Lost Hills fields. These fields were discovered more than a century ago (Volume I, Executive Summary; California Division of Oil, Gas and Geothermal Resources (DOGGR), 1998). We found that hydraulic-fracturing-enabled oil production is occurring within regions with a wide spectrum of existing habitat, including: (1) relatively intact habitat, (2) areas already disturbed by other oil and gas production, and (3) locations dominated by human uses such as agriculture or urban development. We attempted to isolate the impact of hydraulic-fracturing-enabled production on natural habitat by analyzing hydraulic-fracturing-enabled production in the context of the underlying land use.

Over the last century, habitat loss has been the largest documented impact to wildlife and vegetation stemming from oil and gas production activities in California. The extent of the impact was dependent upon the amount and the location of disturbances. Fiehler and Cypher (2011) found that valley saltbush scrub specialists such as San Joaquin antelope squirrels, short-nosed kangaroo rats and San Joaquin kit foxes disappeared from high density oil development, but persisted in areas with less than 70% disturbance. Construction activities that destroyed active den or burrow sites had significant impacts on San Joaquin kit fox populations (O'Farrell and Kato, 1987; Kato and O'Farrell, 1986; O'Farrell et al., 1986). On the other hand, nightly movements (Zoellick et al., 1987), den use patterns (Koopman et al., 1998), and reproductive and survival parameters of the San Joaquin kit fox did not differ between an undeveloped area and an intensely developed area of an oil field (Spiegel and Tom, 1996; Spiegel and Disney, 1996; Cypher et al., 2000).

Smaller species such as blunt-nosed leopard lizards and giant kangaroo rats were minimally impacted by oil and gas production because most of the activities were outside the core habitat areas for both species (O'Farrell and Kato, 1987). In areas where high-quality habitat and activities overlapped, the intensity of development and amount of habitat disturbed determined the carrying capacity[3] (Kato and O'Farrell, 1986). It has been documented that abandoned oil and gas fields undergoing revegetation can be recolonized by blunt-nose leopard lizards as long as densities of shrubs and ground cover do not become excessive (O'Farrell and Kato, 1980).

The studies we surveyed for impacts of oil and gas production to habitat loss and fragmentation within California were all conducted at the Elk Hills oil field, therefore it is difficult to assess the generality of the results to the rest of the state. There also were some limitations to the study designs, principally that the non-developed areas used for comparisons were not equivalent in habitat quality when compared to the developed areas, even prior to any activity.

3. The carrying capacity is the number of individuals of a species that an area can support.

5.3.1.2. Quantitative Analysis Of Hydraulic Fracturing-Enabled Production On Habitat Loss

Our analysis addressed three major questions:

1. How has hydraulic-fracturing-enabled oil production altered well density in California?

2. How are the areas with increased well density distributed across counties, land uses, and habitat types in California?

3. What special-status species occurred in the vicinity of oil fields highly impacted by well stimulation?

5.3.1.2.1. Methods

Here we briefly summarize our methods for the quantitative analysis of the impact of hydraulic fracturing on habitat loss; more information is given in Appendix 5-C, "Detailed Methods for Quantitative Analysis of Hydraulic Fracturing-Enabled Production On Habitat Loss."

For our analysis, we looked at well density as a proxy for habitat loss. As well density increases, the amount of intact habitat tends to decrease; see Figure 5.3.1. for an illustration of how plant cover is affected by increasing well density. We examined 506 plots at least 10 hectares (ha) in size for well density and bare (unvegetated) ground and found that well density predicted 95% of the variation in presence of bare ground. We concluded that well density is an accurate indicator of habitat loss.[4] For this analysis we did not look at how well density correlated with habitat fragmentation; we will look more closely at the issue of fragmentation in the San Joaquin case study in Volume III of this report.

In order to assess the impact of hydraulic-fracturing-enabled oil production on habitat, we set out to quantify the density of hydraulically fractured wells in the state. This was challenging given that reporting of hydraulic fracturing was not required until 2013, so records of the activity are likely incomplete. We used a compilation of well records, voluntary reporting to FracFocus, and recent mandatory reporting to estimate the proportion of hydraulically fractured wells tapping each pool (also called reservoirs). We then generated two alternate scenarios: actual well density, and a "without hydraulic fracturing" well density. Actual well density is the true density of wells in California

4. We performed a linear regression of proportion of bare ground as predicted by well density for 506 plots at least 10 hectares in size. The relationship was highly significant; $F(1,504) = 9107$, $p=<2.48x10^{-7}$, adjusted $r^2 = 0.95$. See Appendix 5.C for further details.

as of September 2013. Background well density represents a hypothetical scenario representing the well density of California as of September 2014 if every well that had been hydraulically fractured vanished. The difference between the two is the marginal impact of hydraulic fracturing-enabled production on well density and, by proxy, habitat loss and fragmentation.

An important point to understand about this analysis is that hydraulic fracturing compared to background well density does not represent a change over time. That is, well density was *not* at the background level at some point in time, then hydraulic fracturing increased the density from that time forward. Hydraulically fractured and unstimulated wells continue to be drilled and produced simultaneously. The main reason why wells that are hydraulically fractured are geographically interspersed with other wells in California is because low-permeability reservoirs that require hydraulic fracturing are often stacked above and below reservoirs that do not require hydraulic fracturing. For example, in the South Belridge field, the Tulare pool is above the Diatomite pool. 91% of well records in the Diatomite report hydraulic fracturing, as compared to only 1% in the Tulare. This creates a patchwork of wells at the surface that are and are not hydraulically fractured. Even if all hydraulically fractured wells disappeared from South Belridge, the well density in much of the field would still be high, and there would be little usable habitat for native organisms.

We split well density into four categories comparable to those used in Fiehler and Cypher (2011): Control – less than one well/km^2; Low – 1-15 wells/km^2; Medium - 15-77 wells/km^2; High - more than 77 wells/km^2. We chose to use the same categories because Fiehler and Cypher (2011) conducted the only previous work we could find systematically associating land disturbance from oil and gas activities with the decline of natural communities in California[5]. We then calculated the number of hectares that either were unchanged or increased in density category because of hydraulic fracturing-enabled production. We refer to areas that did not change categories as "not noticeably impacted," areas that moved from the control group to a higher category as "newly impacted," and areas that shifted from the low and medium categories to a higher category as experiencing "increased intensity" of production. We refer to the newly impacted and increased intensity areas collectively as "altered" areas. Table 5.3.1 summarizes how we categorize changes in well density.

5. Our categories differ from Fiehler and Cypher (2011) in two respects. First, Fiehler and Cypher had a gap between the medium and high categories: the medium category ended at 77 wells/km^2 and high began at 150 wells km^2; we reassigned the lower end of the high category as 77 wells/km^2 to eliminate the gap. Second, Fiehler and Cypher counted wells in study areas of around 0.648 km^2 in size while we estimated the number of wells/km^2 in a moving window of comparable size.

Table 5.3.1. Description of well density categories used in this study. We divided the effect of hydraulic-fracturing-enabled production on well density into three major categories: newly developed, increased intensity, and not noticeably impacted areas. The three categories are defined in terms of the types of shifts between density classes. We use blue, yellow, red and gray consistently to color-code the three categories throughout this chapter. For simplicity, we refer collectively to areas that were newly developed or increased in intensity as showing an increase in hydraulic fracturing, with the caveat that our results do not factor in areas that increased in well density due to hydraulic-fracturing-enabled-production, but not enough to move up a category.

	Category	**Change between density classes**
Altered	Newly developed	Control -> Low, Med, High
	Increased intensity	Low -> Med, High
		Med ->High
Unaltered	Either no change in well density, or no noticeable change in well density (that is, not enough to shift the density to a higher class).	Control-> Control Low->Low Med->Med High->High

Figure 5.3.1. Aerial photos of each well-density category. The off-white areas are well pads, roads, and other unvegetated, highly disturbed areas. The gray, blotchy regions are vegetated areas that represent a natural habitat type. As well density increases, the amount of unvegetated land increases. (A) Control – less than one well per / km^2. (B) Low – 1-15 wells / km^2). (C) Medium - 15-77 wells / km^2 (D) High - more than 77 wells / km^2.

We classified areas first by *land use* (developed, agricultural, or natural areas); for natural areas, we looked more closely at broad *land cover* types, which refer to functional types of vegetation: shrubland and grassland, forest and woodland, open water, and so forth (UCSB Biogeography Lab, 1998). We further subdivided land cover types into *natural communities*, which subdivides the state into common plant associations such as valley saltbush scrub, non-native grassland, and so forth (Holland 1986). There are more than

200 natural community categories; as a result, we focused on the four with more than 1,000 hectares of altered area plus two aquatic habitat types, and grouped the remainder under “other natural communities.” Table 5.3.2. gives the categories and classifications we used in our assessment.

Table 5.3.2. Categories of land use, land cover, and natural communities used in this assessment.

Category	**Classifications**	**Data Source**
Land Use	1. Developed and other human use 2. Agricultural, introduced, or modified vegetation 3. Natural habitat, subdivided by the classifications given in Land Cover and Habitat Type	California DOC (2012)
Land Cover	1. Shrubland and grassland 2. Semi-desert 3. Forest and woodland 4. Open water 5. Polar, high montane, and barren	UCSB Biogeography Lab (1998)
Natural Community*	1. Valley saltbush scrub 2. Non-native grassland 3. Venturan coastal sage scrub 4. Buck brush chaparral 5. Water 6. Riparian and wetland 7. Other natural communities	Holland (1986)

** Some of our “Natural Community” groups are equivalent to the natural communities described in Holland (1986), while others (water, and riparian and wetland) group a number of Holland natural communities under one header.*

5.3.1.2.2. Results and Discussion of Quantitative Analysis of Well Stimulation Impacts to Habitat Loss and Fragmentation

We estimated that 33,000 hectares shifted to a higher well density category with hydraulic-fracturing-enabled oil production; of this, about 21,000 hectares (60%) was natural habitat. About 1% of California’s land is developed for oil and gas production (with a well density greater than $1/km^2$), compared to 5% for urban development and 14% for agriculture. About 3.5% of the habitat loss due to oil and gas production as a whole is attributable to hydraulic-fracturing-enabled activity.

The impacts of oil and gas production in general, and well stimulation in particular, are concentrated in a few areas of the state. Of the 33,000 hectares statewide that shifted to a higher well density category with hydraulic-fracturing enabled production, about 27,000 hectares (81%) were in Kern and Ventura Counties. About 8% of Kern and 4% of all lands in Ventura Counties are developed for oil and gas production (with a well density greater than $1/km^2$).

The main habitat types disturbed by hydraulic fracturing-enabled production are valley saltbush scrub, non-native grassland, Venturan coastal sage scrub, and buck brush chaparral. These habitat types are mainly found in Kern and Ventura Counties. Twenty-four federally and/or state-listed threatened and endangered species have documented occurrences in oil fields where at least 200 hectares have reached a higher well-density class with hydraulic-fracturing-enabled production.

Question 1*: How has hydraulic fracturing-enabled production altered well density in California?*

Well density has increased in California due to hydraulic-fracturing-enabled production (Table 5.3.3). We estimate that about 33,000 hectares of land in the state have shifted into a higher-density category due to hydraulic-fracturing-enabled production (Table 5.3.3, red, yellow, and blue cells). 15,196 hectares were newly impacted by oil and gas development because of hydraulic-fracturing-enabled development (Table 5.3.3, blue cells). About 18,999 hectares already had wells present, but hydraulic fracturing enabled an increase in density (Table 5.3.3, yellow and red cells).

Table 5.3.3. The effect of hydraulic-fracturing-enabled production on well density in California oil and gas fields. The table shows the number of hectares in the state in a given category of well density without hydraulic-fracturing-enabled-production along the rows, and with hydraulic-fracturing-enabled-production along the columns. For example, 13,075 hectares in California had a control well density without hydraulically fractured wells, and a low well density with hydraulically fractured wells. Blue backgrounds indicate the area that was newly impacted by oil and gas production because of hydraulic-fracturing-enabled production. Yellow and red backgrounds show areas that were more intensively developed for oil and gas with hydraulic-fracturing enabled production. Gray backgrounds show the area where well density was not noticeably affected by hydraulic-fracturing-enabled production. The sum of blue, yellow, and red cells equals the total area altered by hydraulic-fracturing-enabled production.

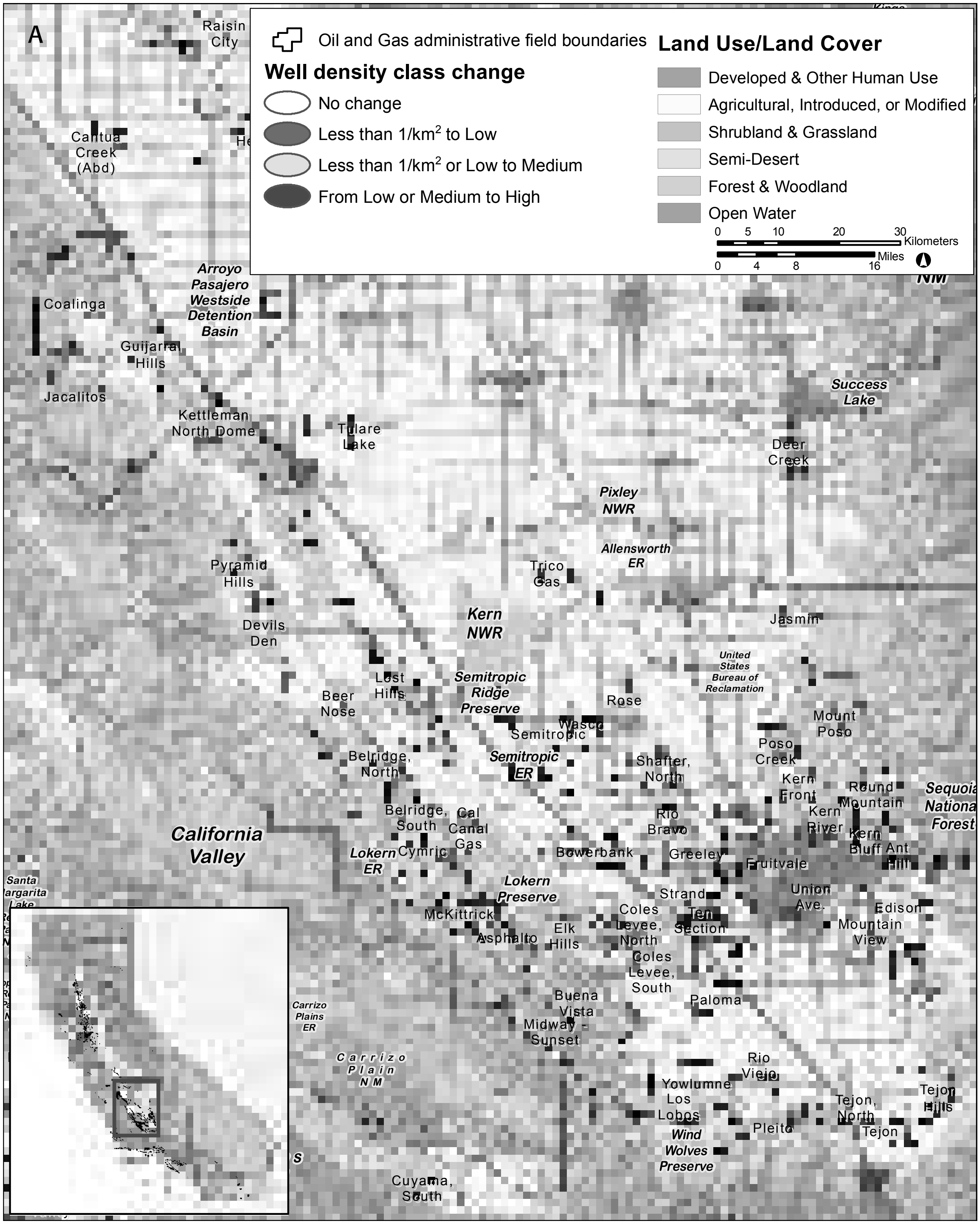
		Well Density With Hydraulic-Fracturing-Enabled Production (ha)			
		Control	Low	Medium	High
Background Well Density (ha)	Control	41,958,038	13,075	2,114	7
	Low		301,709	11,773	772
	Medium			70,044	5,308
	High				31,799

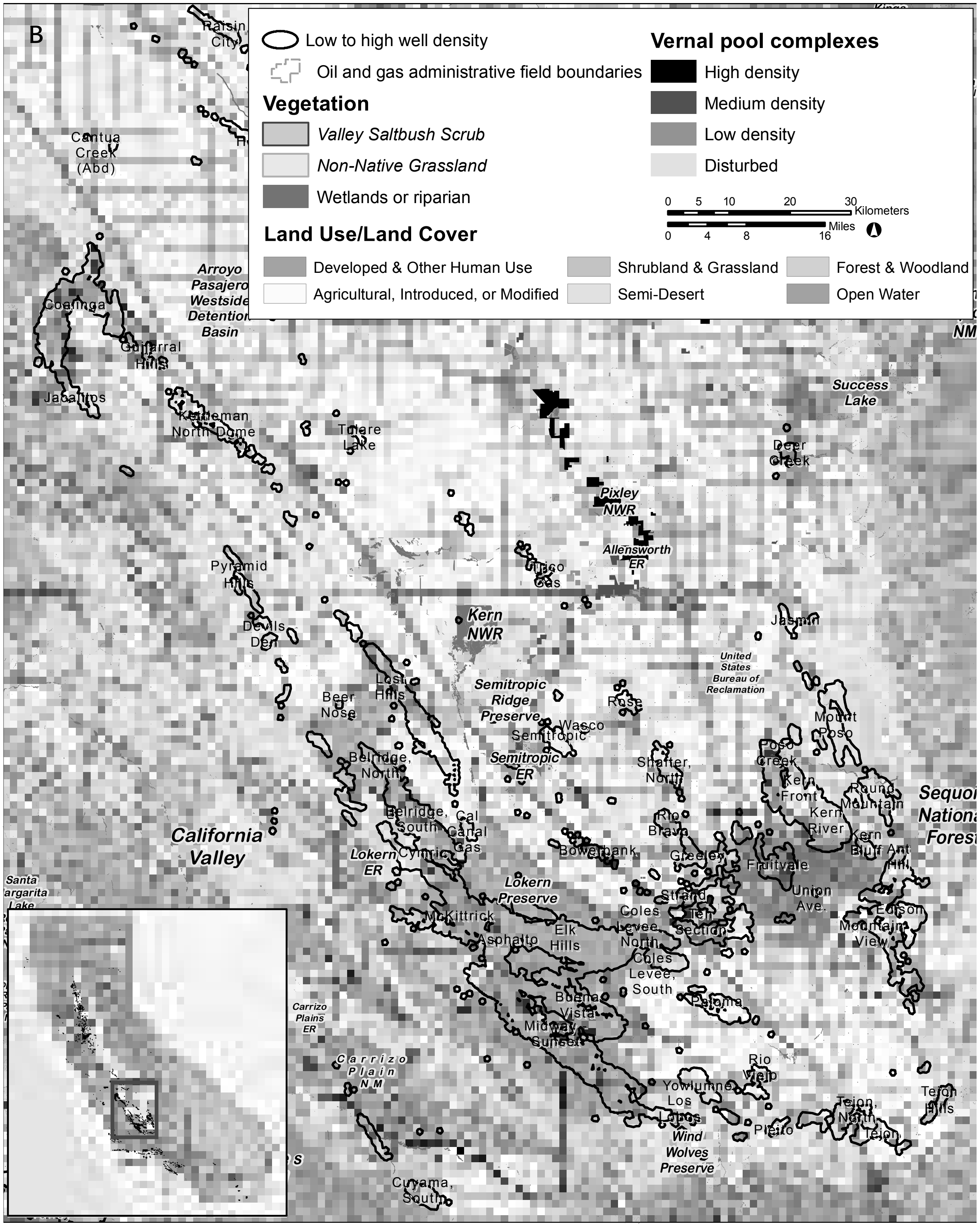
The majority of altered area in the San Joaquin Valley occurred around the southern perimeter of the valley in fields dominated by shrubland and grassland such as Elk Hills, Buena Vista, Midway-Sunset, Lost Hills, Mt. Poso and Round Mountain. Figure 5.3.2 (a) and Figure 5.3.3 (a). There are smaller amounts of altered habitat in the central portion of the valley where agriculture is the dominant land use in oil fields such as North Shafter and Rose.

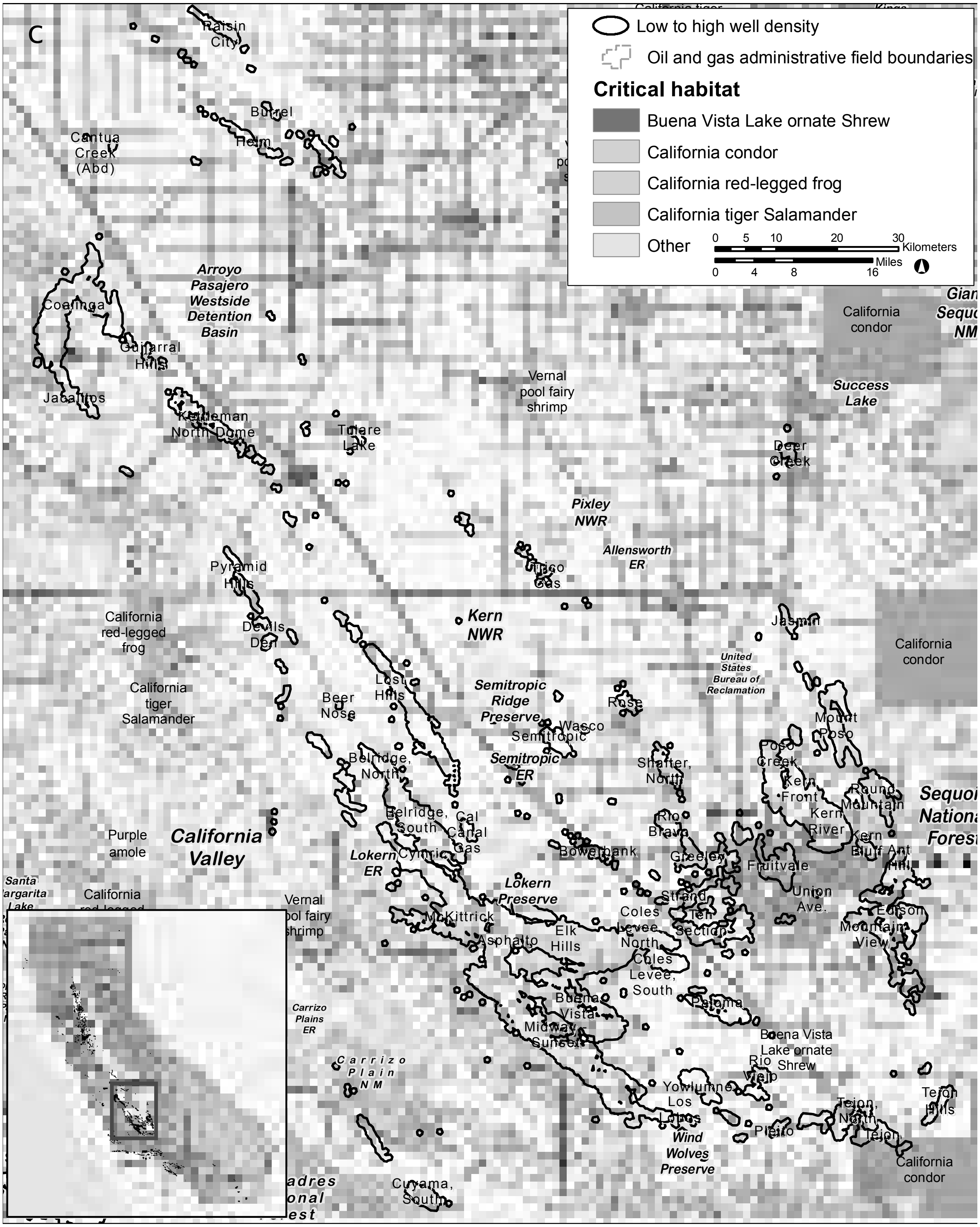
The inner core of fields such as Lost Hills and North and South Belridge Fields, where production of diatomite pools requires hydraulic fracturing, were considered unaltered (for the purposes of habitat quality) by well stimulation because they were already high-density regardless of hydraulic-fracturing-enabled-development. Lost Hills, North and South Belridge collectively represent 79% of reported hydraulic fracturing in the state (Volume I, Chapter 3, Table 3-1). Because these fields are also the location of intensively developed pools that do not require hydraulic fracturing, much of this area is already largely inhospitable to most native wildlife and vegetation, regardless of the added well density attributable to hydraulic fracturing. Thus, the additional impact of hydraulic fracturing to habitat degradation in these areas is probably minimal.

In Ventura County, the majority of altered area occurred in a string of three fields along the transverse mountain range: the Sespe, Ojai, and Ventura fields. Although the total well densities of the Ojai and Sespe are not very high, nearly all of the development is enabled by hydraulic fracturing. The Ventura field is a bit different as it already had a moderate level of development and hydraulic-fracturing-enabled-development increased the intensity. The portions of the Ojai and Sespe altered by hydraulic-fracturing-enabled-development overlap mostly with natural habitat; in the Ventura Field, the altered areas were mostly in urban and built-up land.

Appendix 5.B, Maps of Well Density in California, shows larger versions of these maps for the major hydrocarbon-producing basins of California.


A
Oil and Gas administrative field boundaries
Well density class change
No change
Less than 1/km^2 to Low
Less than 1/km^2 or Low to Medium
From Low or Medium to High
Land Use/Land Cover
Developed & Other Human Use
Agricultural, Introduced, or Modified
Shrubland & Grassland
Semi-Desert
Forest & Woodland
Open Water
0 5 10 20 30 Kilometers
0 4 8 16 Miles
Raisin City
Cantua Creek (Abd)
Coalinga
Arroyo Pasajero Westside Detention Basin
Guijarral Hills
Jacalitos
Kettleman North Dome
Tulare Lake
Success Lake
Deer Creek
Pixley NWR
Allensworth ER
Pyramid Hills
Trico Gas
Kern NWR
Devils Den
Jasmin
United States Bureau of Reclamation
Lost Hills
Semitropic Ridge Preserve
Beer Nose
Rose
Wasco
Semitropic
Mount Poso
Poso Creek
Belridge, North
Semitropic ER
Shafter, North
Kern Front
Round Mountain
Sequoia National Forest
Belridge, South
Cal Canal Gas
Rio Bravo
Kern River
Kern Bluff
Ant Hill
California Valley
Lokern ER
Cymric
Bowerbank
Greeley
Fruitvale
Lokern Preserve
Strand
Union Ave.
Edison
McKittrick
Coles Levee, North
Ten Section
Mountain View
Asphalto
Elk Hills
Coles Levee, South
Carrizo Plains ER
Buena Vista
Paloma
Midway - Sunset
Carrizo Plain NM
Rio Viejo
Yowlumne
Los Lobos
Tejon, North
Tejon Hills
Wind Wolves Preserve
Pleito
Tejon
Cuyama, South


B
Low to high well density
Oil and gas administrative field boundaries
Vegetation
Valley Saltbush Scrub
Non-Native Grassland
Wetlands or riparian
Vernal pool complexes
High density
Medium density
Low density
Disturbed
Land Use/Land Cover
Developed & Other Human Use
Agricultural, Introduced, or Modified
Shrubland & Grassland
Semi-Desert
Forest & Woodland
Open Water
0 5 10 20 30 Kilometers
0 4 8 16 Miles
Cantua Creek (Abd)
Coalinga
Jacalitos
Kettleman North Dome
Tulare Lake
Arroyo Pasajero Westside Detention Basin
Pixley NWR
Allensworth ER
Success Lake
Deer Creek
Pyramid Hills
Devils Den
Kern NWR
Jasmin
Lost Hills
Beer Nose
Semitropic Ridge Preserve
Semitropic
Semitropic ER
Wasco
Rose
Mount Poso
Poso Creek
Kern Front
Round Mountain
Kern River
Kern Bluff
Ant Hill
Belridge, North
Belridge, South
Cal Canal Gas
Cymric
Lokern ER
Lokern Preserve
California Valley
McKittrick
Asphalto
Elk Hills
Bowerbank
Shafter, North
Rio Bravo
Greeley
Fruitvale
Strand
Ten Section
Union Ave.
Edison
Mountain View
Coles Levee, North
Coles Levee, South
Buena Vista
Midway Sunset
Paloma
Carrizo Plains ER
Carrizo Plain NM
Yowlumne
Los Lobos
Rio Viejo
Wind Wolves Preserve
Pleito
Tejon, North
Tejon
Tejon Hills
Cuyama, South
United States Bureau of Reclamation
Santa Margarita Lake
NM


C
Low to high well density
Oil and gas administrative field boundaries
Critical habitat
Buena Vista Lake ornate Shrew
California condor
California red-legged frog
California tiger Salamander
Other
0 5 10 20 30 Kilometers
0 4 8 16 Miles
Raisin City
Burrel
Helm
Cantua Creek (Abd)
Arroyo Pasajero Westside Detention Basin
Coalinga
Guijarral Hills
Jacalitos
Kettleman North Dome
Tulare Lake
Vernal pool fairy shrimp
California condor
Success Lake
Deer Creek
Pixley NWR
Allensworth ER
Pyramid Hills
Trico Gas
Kern NWR
Jasmin
California condor
California red-legged frog
Devils Den
United States Bureau of Reclamation
California tiger Salamander
Beer Nose
Lost Hills
Semitropic Ridge Preserve
Wasco
Rose
Semitropic
Semitropic ER
Mount Poso
Poso Creek
Shafter, North
Belridge, North
Kern Front
Round Mountain
Sequoia National Forest
Belridge, South
Cal Canal Gas
Rio Bravo
Kern River
Kern Bluff
Ant Hill
Purple amole
California Valley
Lokern ER
Cymric
Bowerbank
Greeley
Fruitvale
Lokern Preserve
Strand
Union Ave.
Edison
California
Vernal pool fairy shrimp
McKittrick
Coles Levee, North
Ten Section
Mountain View
Asphalto
Elk Hills
Coles Levee, South
Carrizo Plains ER
Buena Vista
Paloma
Midway-Sunset
Buena Vista Lake ornate Shrew
Rio Viejo
Carrizo Plain NM
Yowlumne
Los Lobos
Tejon Hills
Tejon, North
Wind Wolves Preserve
Pleito
Tejon
California condor
Cuyama, South

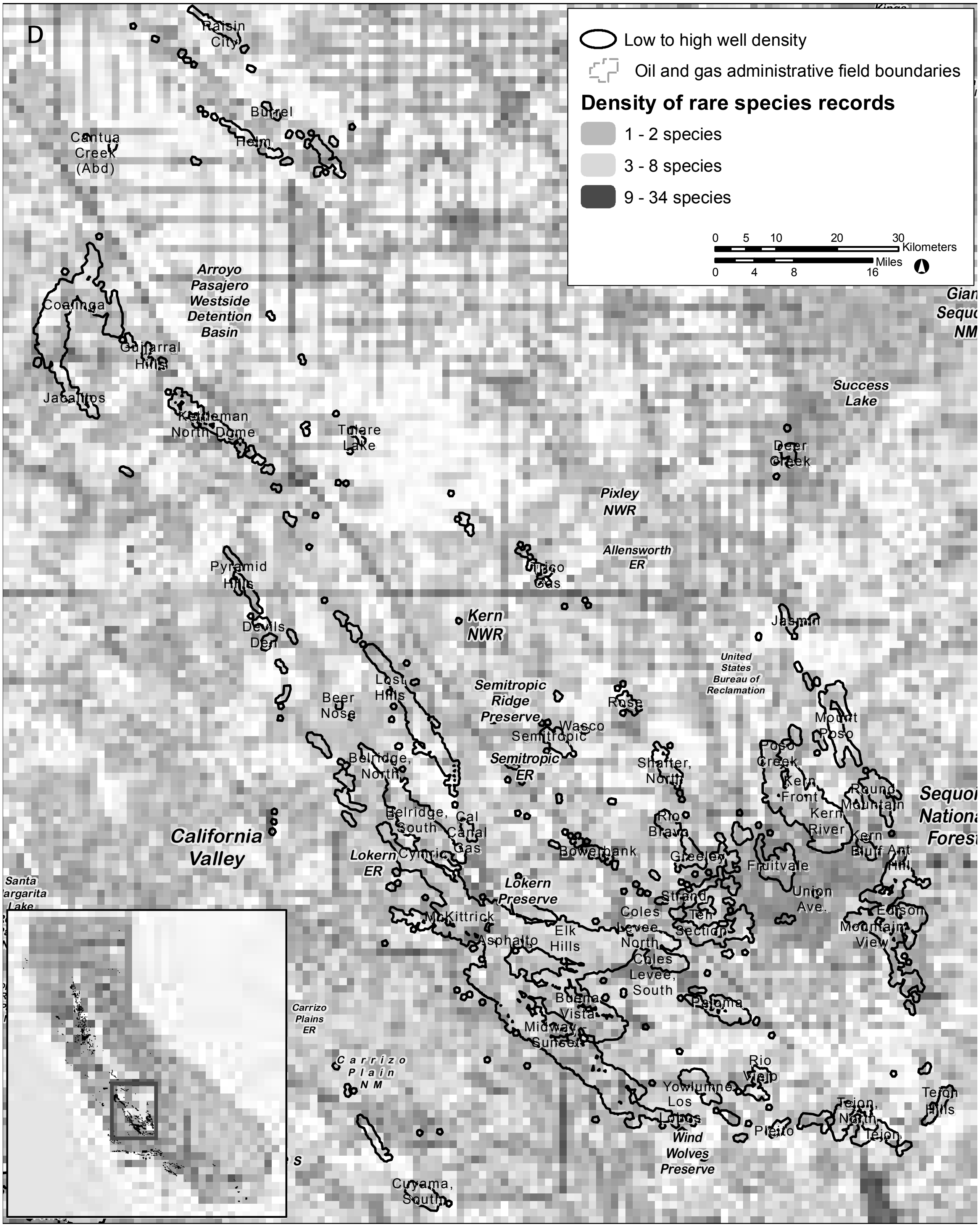
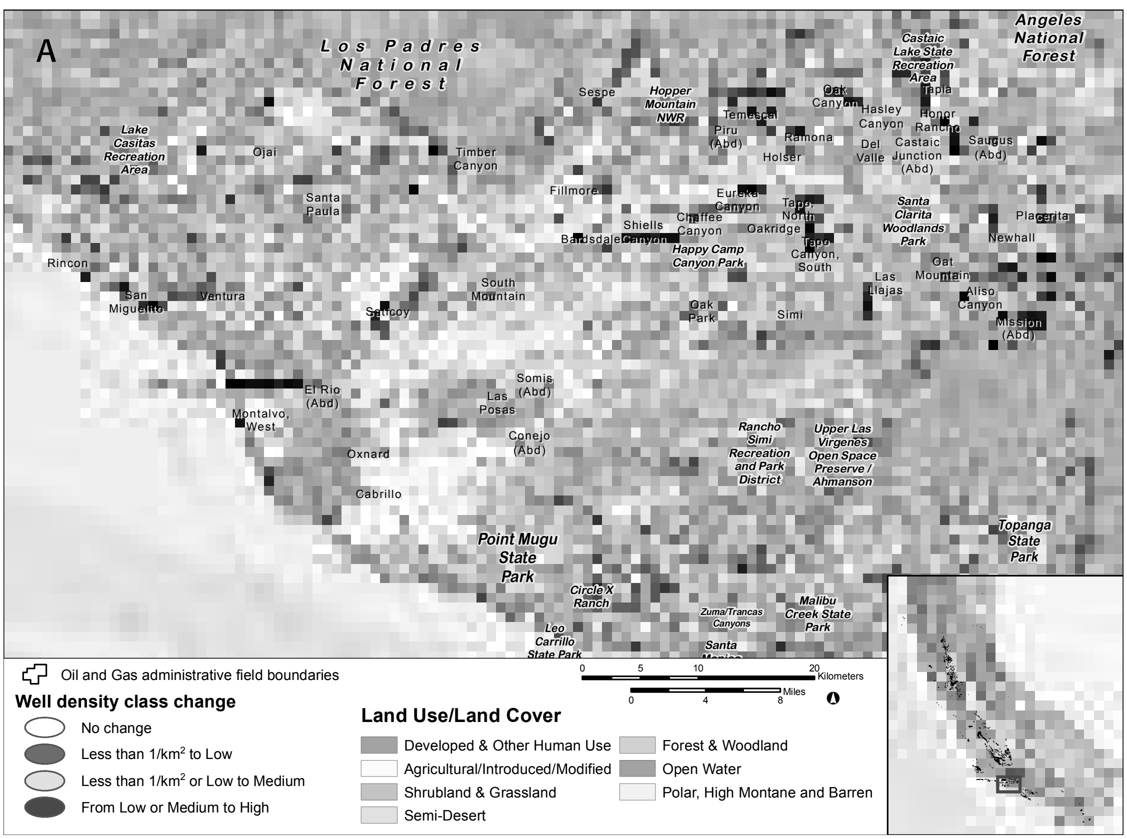
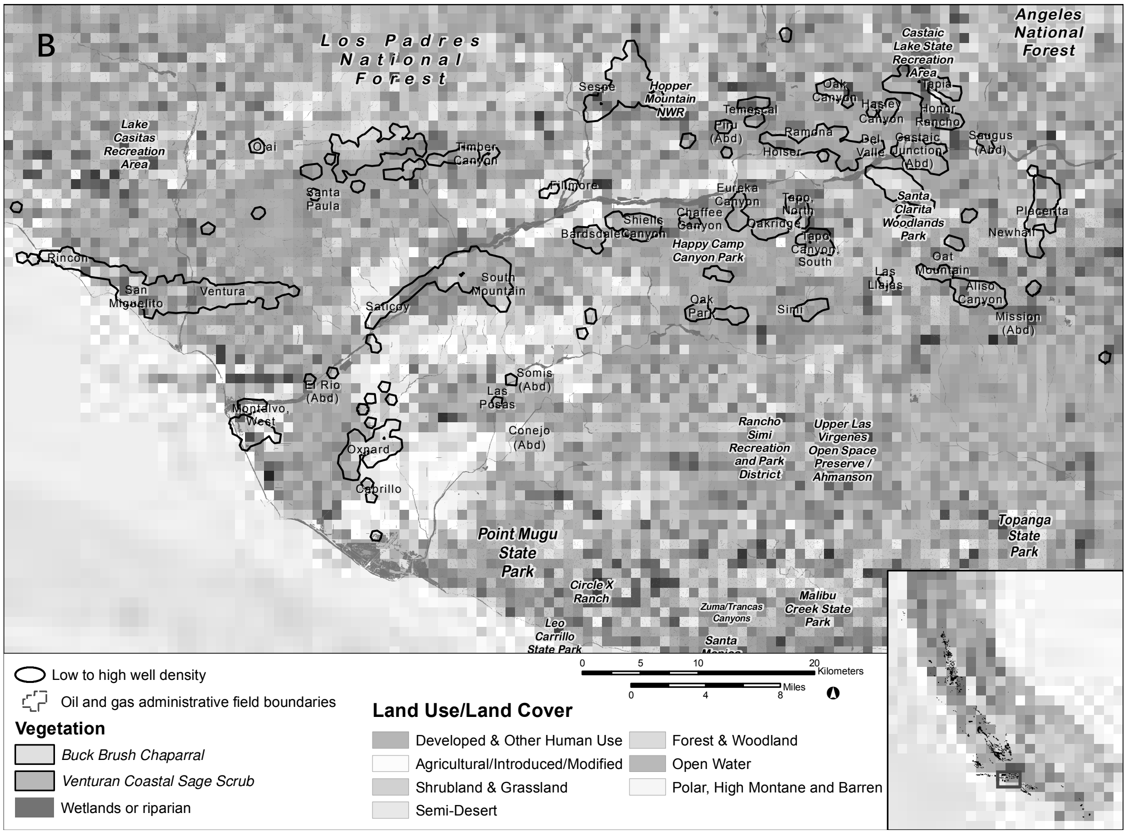
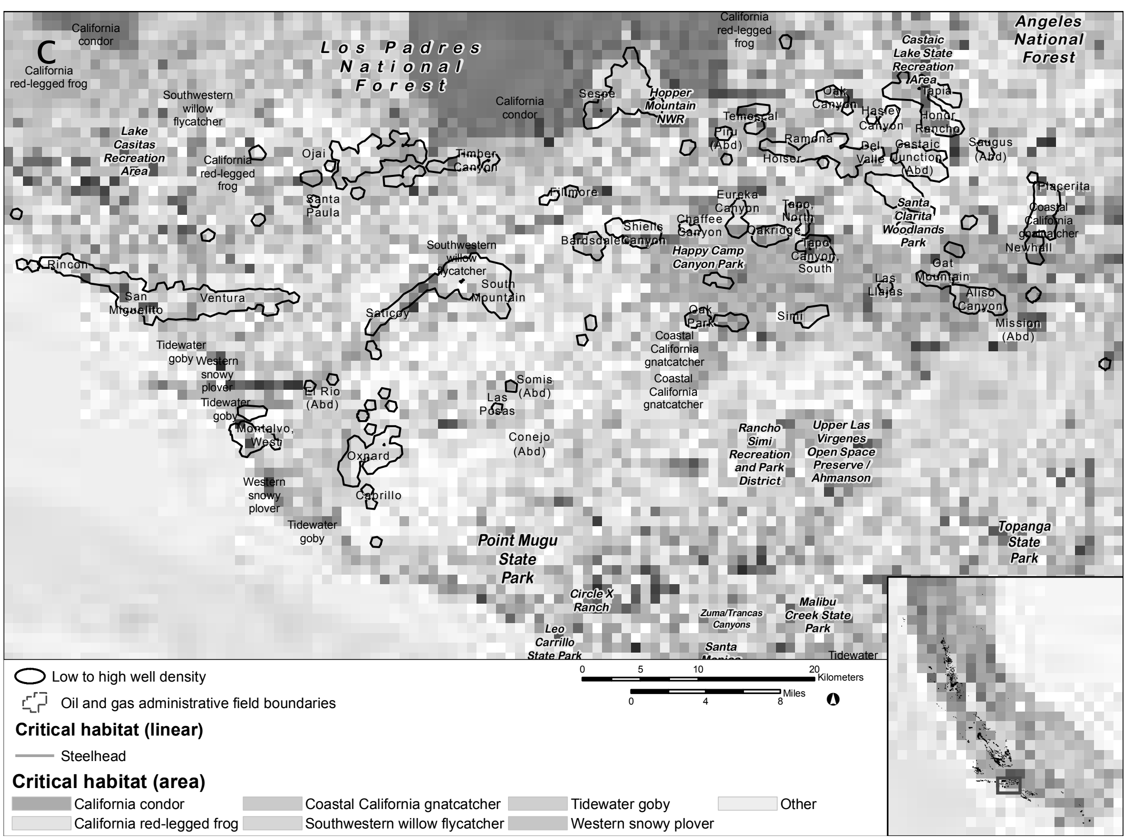

D
Low to high well density
Oil and gas administrative field boundaries
Density of rare species records
1 - 2 species
3 - 8 species
9 - 34 species
0 5 10 20 30 Kilometers
0 4 8 16 Miles
Raisin City
Burrel
Helm
Cantua Creek (Abd)
Arroyo Pasajero Westside Detention Basin
Coalinga
Guijarral Hills
Jacalitos
Kettleman North Dome
Tulare Lake
Success Lake
Deer Creek
Pixley NWR
Allensworth ER
Pyramid Hills
Tulare Lake
Trico Gas
Kern NWR
Jasmin
Devils Den
United States Bureau of Reclamation
Lost Hills
Beer Nose
Semitropic Ridge Preserve
Rose
Wasco
Semitropic
Mount Poso
Poso Creek
Belridge, North
Semitropic ER
Shafter, North
Kern Front
Round Mountain
Sequoia National Forest
Belridge, South
Cal Canal Gas
Rio Bravo
Kern River
California Valley
Lokern ER
Cymric
Bowerbank
Kern Bluff
Ant Hill
Greeley
Fruitvale
Santa Margarita Lake
Lokern Preserve
Strand
Union Ave.
Edison
McKittrick
Coles Levee, North
Ten Section
Mountain View
Asphalto
Elk Hills
Coles Levee, South
Buena Vista
Paloma
Carrizo Plains ER
Midway-Sunset
Carrizo Plain NM
Rio Viejo
Yowlumne
Los Lobos
Tejon, North
Tejon Hills
Pleito
Tejon
Wind Wolves Preserve
Cuyama, South

Figure 5.3.2. Maps of the San Joaquin Basin showing the increase in well density attributable to hydraulic fracturing-enabled development and key ecological features (a) Change in well density due to hydraulic fracturing-enabled production. Colors show areas that increased in well density due to fracturing-enabled production. Blue indicates areas that increased to low density with the addition of hydraulically fractured wells, yellow shows areas that increased to medium, and red indicates areas that increased to high well density. (b) Selected habitat types for the San Joaquin Basin, including dominant types (non-native grassland and valley saltbush scrub), wetland and riparian habitat, and vernal pools complexes are indicated. Black outlines indicate areas developed for oil and gas production (with at least 1 well per km²). (c) Critical habitat in the region, shown as colored polygons. Despite a high concentration of threatened and endangered species, little critical habitat has been designated in the San Joaquin Valley. Critical habitat for the Buena Vista Lake ornate shrew is south of Bakersfield, it is labeled to but too small to be visible. Black outlines indicate areas developed for oil and gas production. (d) Density of rare species records recorded in the CNDDB. Black outlines indicate areas developed for oil and gas production. Data sources: DOGGR, 2014a; 2014b; 2014c; UCSB Biogeography Lab, 1998; California DOC, 2012; USFWS, 2014b, Biogeographic Data Branch DFW, 2014.




B
Los Padres National Forest
Angeles National Forest
Castaic Lake State Recreation Area
Lake Casitas Recreation Area
Ojai
Timber Canyon
Santa Paula
Sespe
Hopper Mountain NWR
Temescal
Piru (Abd)
Ramona
Holser
Oak Canyon
Hasley Canyon
Tapia
Honor Rancho
Del Valle
Castaic Junction (Abd)
Saugus (Abd)
Fillmore
Eureka Canyon
Tapo North
Chaffee Canyon
Shiells Canyon
Bardsdale
Oakridge
Happy Camp Canyon Park
Tapo Canyon, South
Santa Clarita Woodlands Park
Placerita
Newhall
Oat Mountain
Las Llajas
Aliso Canyon
Mission (Abd)
Rincon
San Miguelito
Ventura
Saticoy
South Mountain
Oak Park
Simi
Somis (Abd)
Las Posas
El Rio (Abd)
Montalvo, West
Oxnard
Conejo (Abd)
Cabrillo
Rancho Simi Recreation and Park District
Upper Las Virgenes Open Space Preserve / Ahmanson
Topanga State Park
Point Mugu State Park
Circle X Ranch
Leo Carrillo State Park
Zuma/Trancas Canyons
Malibu Creek State Park
Santa Monica
Low to high well density
Oil and gas administrative field boundaries
Vegetation
Buck Brush Chaparral
Venturan Coastal Sage Scrub
Wetlands or riparian
Land Use/Land Cover
Developed & Other Human Use
Agricultural/Introduced/Modified
Shrubland & Grassland
Semi-Desert
Forest & Woodland
Open Water
Polar, High Montane and Barren
0 5 10 20 Kilometers
0 4 8 Miles


C
California condor
California red-legged frog
Southwestern willow flycatcher
Los Padres National Forest
Angeles National Forest
Castaic Lake State Recreation Area
Lake Casitas Recreation Area
Ojai
Timber Canyon
Santa Paula
Sespe
Hopper Mountain NWR
Temescal
Piru (Abd)
Ramona
Holser
Oak Canyon
Hasley Canyon
Tapia
Honor Rancho
Del Valle
Castaic Junction (Abd)
Saugus (Abd)
Fillmore
Eureka Canyon
Tapo North
Chaffee Canyon
Shiells Canyon
Bardsdale
Oakridge
Happy Camp Canyon Park
Tapo Canyon, South
Santa Clarita Woodlands Park
Placerita
Coastal California gnatcatcher
Newhall
Oat Mountain
Las Llajas
Aliso Canyon
Mission (Abd)
Rincon
San Miguelito
Ventura
Saticoy
South Mountain
Oak Park
Simi
Tidewater goby
Western snowy plover
Somis (Abd)
Las Posas
El Rio (Abd)
Montalvo, West
Oxnard
Conejo (Abd)
Cabrillo
Rancho Simi Recreation and Park District
Upper Las Virgenes Open Space Preserve / Ahmanson
Topanga State Park
Point Mugu State Park
Circle X Ranch
Leo Carrillo State Park
Zuma/Trancas Canyons
Malibu Creek State Park
Santa Monica
Low to high well density
Oil and gas administrative field boundaries
Critical habitat (linear)
Steelhead
Critical habitat (area)
California condor
California red-legged frog
Coastal California gnatcatcher
Southwestern willow flycatcher
Tidewater goby
Western snowy plover
Other
0 5 10 20 Kilometers
0 4 8 Miles

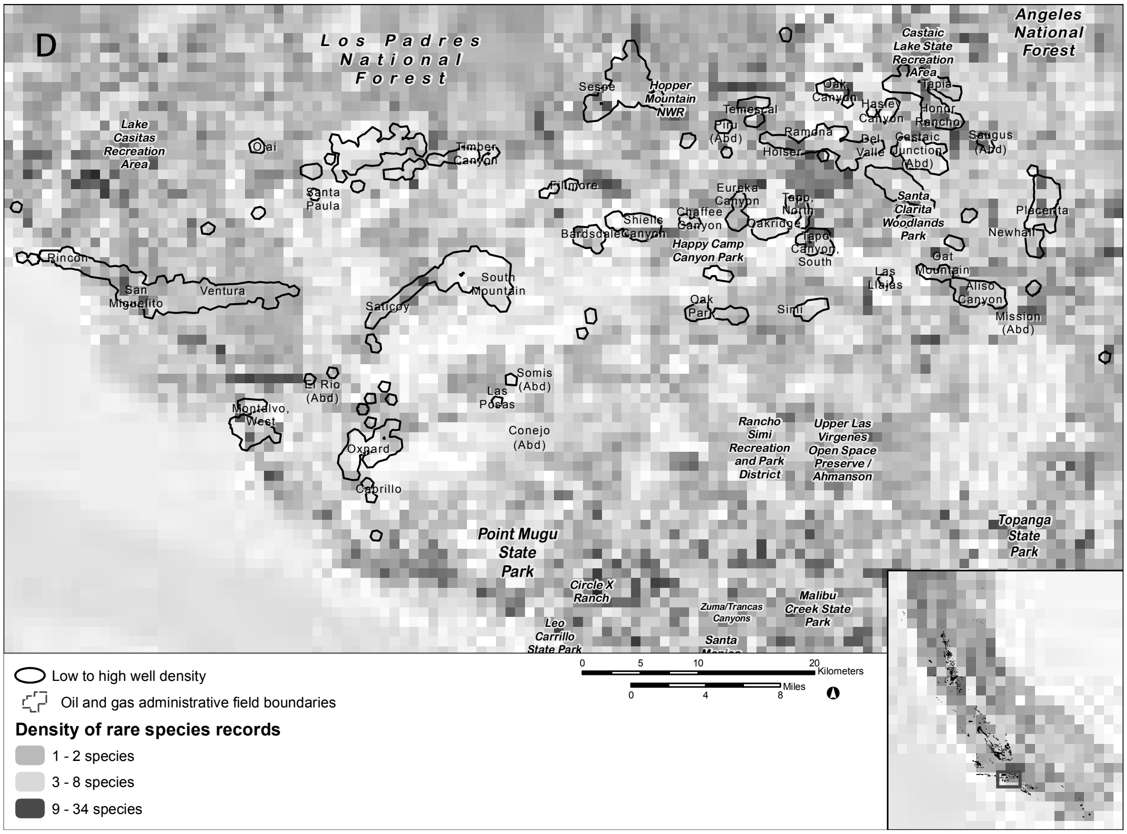


Figure 5.3.3. Maps of the Ventura Basin showing the increase in well density attributable to hydraulic fracturing-enabled development and key ecological features. (a) Change in well density due to hydraulic fracturing-enabled production. Blue indicates an area changed from control to low or medium density with the addition of hydraulically fractured wells. Yellow shows areas that changed from low to medium or high. Red indicates areas that changed from medium to high. Shrub and grassland were the land cover types most impacted by fracturing-enabled production. (b) Vegetation in the Ventura Basin. Dominant habitat types (buck brush chaparral and Venturan coastal sage scrub), wetland and riparian habitat are indicated. Black outlines indicate areas developed for oil and gas production. (c) Designated critical habitat shown as colored polygons. Critical habitat for California condor and steelhead salmon overlap with impacted areas. Black outlines indicate areas developed for oil and gas production. (d) Density of rare species records recorded in the CNDDB. Black outlines indicate areas developed for oil and gas production. Data sources: DOGGR, 2014a; 2014b; 2014c; UCSB Biogeography Lab, 1998; California DOC, 2012; USFWS, 2014b, Biogeographic Data Branch DFW, 2014.

Question 2*: How are the areas with increased well density distributed across habitat types and counties in California?*

Of the 33,000 hectares in the state affected by hydraulic-fracturing-enabled production, 60% was natural habitat, 32% was agricultural, and 8% was urban, built-up, or barren. Nearly 90% of natural habitat impacted by hydraulic fracturing was in Kern and Ventura Counties, 64% in Kern and 24% in Ventura (see Table 5.3.4). This finding motivated us to focus principally on Kern and Ventura Counties for the remainder of our assessment of the effect of hydraulic fracturing on habitat loss and fragmentation.

Table 5.3.4. Hectares by county and all of California for areas developed for oil and gas production (with a well density of at least 1 well per km^2), altered area (areas that shifted up in well density category with hydraulic fracturing-enabled production), and altered natural habitat (areas classified as natural habitat that shifted up in well density category with hydraulic fracturing-enabled production). All numbers rounded to the hundreds place; some numbers may not sum due to rounding.

	Developed Area		**Altered Area**		**Altered Natural Habitat**	
	Hectares	% of Column	Hectares	% of Column	Hectares	% of Column
Kern	163,100	37%	20,100	61%	13,400	64%
Ventura	23,200	5%	6,900	21%	5,000	24%
All Other Counties	250,300	57%	6,000	18%	2,500	12%
State Total	436,600	100%	33,000	100%	20,900	100%

The habitat types that were most impacted were those that occur in oil fields of Kern and Ventura Counties where a large proportion of wells are stimulated: valley saltbush scrub, non-native grassland, Venturan coastal sage scrub, and buck brush chaparral all had over 1,000 hectares increase in well density. The maps in Figure 5.3.2 (b) and Figure 5.3.3(b) show the locations of the key altered communities in the southern San Joaquin and Ventura Counties. Figure 5.3.4 shows impacts to land use and habitat types broken out by county.

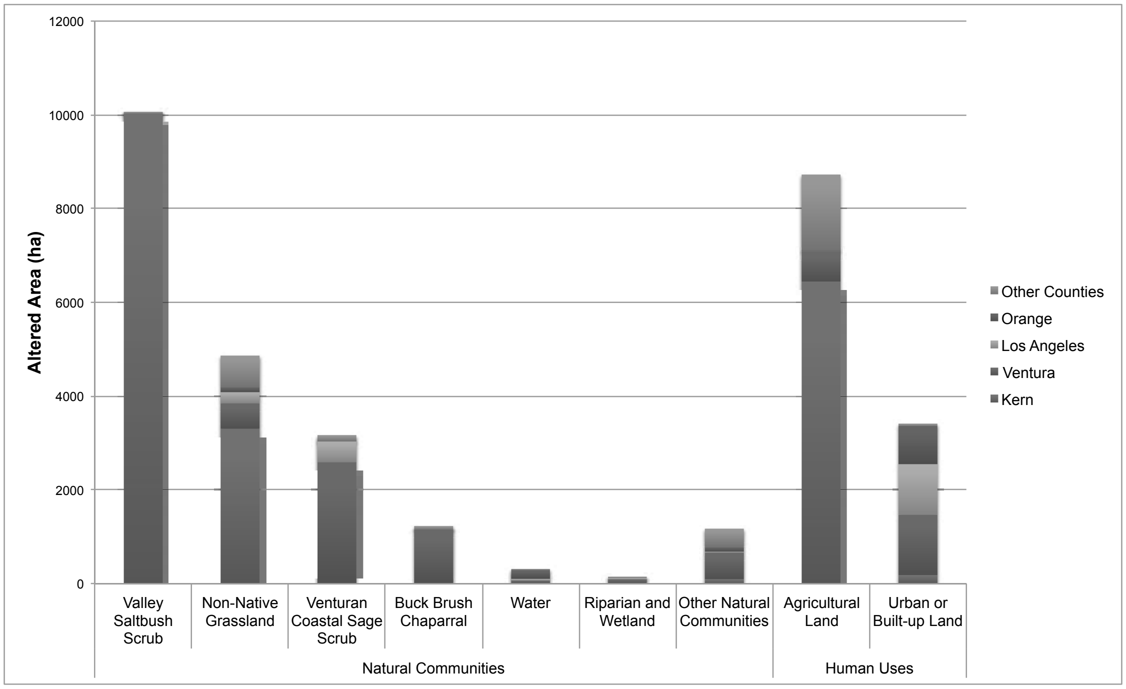


Figure 5.3.4. Land use and habitat types impacted by hydraulic-fracturing-enabled production in California. A large amount of the area that increased in well density due to hydraulic fracturing is agricultural or urban land already highly disturbed by humans and generally unsuitable as habitat for native wildlife and vegetation. Areas designated as natural communities are important habitat for wildlife and vegetation. The counties that had the greatest amount of impacted area are color-coded. The data used to generate this figure are in Appendix 5.D, Table 5.D.2.

The rate of natural habitat areas newly impacted by hydraulic-fracturing-enabled production is a larger proportion of recent activity (from Oct 1, 2012 – Sep. 30, 2014). Of the 1,400 hectares that were newly developed for oil and gas production during the period from Oct. 1, 2012 to Sep. 30, 2014, about 300 hectares (18%) could be attributed to hydraulic fracturing.

Habitat loss caused by hydraulic-fracturing-enabled-production is highly localized and has disproportionate effects in a few areas and for a few habitat types. For valley saltbush scrub, 6% of its statewide extent was impacted by hydraulic-fracturing-enabled-production, and 2% for Venturan coastal sage scrub (Appendix 5.D, Table 5.D.1). In proportion to the total amount of habitat in the state, the amount of habitat impacted by hydraulic-fracturing-enabled-production is small: on the order of less than one-tenth of one percent.

The area of altered aquatic habitat was quite small. Statewide, there were about 300 hectares of altered open water habitat and 140 of riparian and wetland habitat. While the impacts to aquatic habitats was small in terms of total area affected by hydraulic-fracturing-enabled-production, even small impacts to aquatic areas merit consideration because they are generally considered high-value habitats and are accorded special protections under the Federal Clean Water and Coastal Zone Management Acts, as well as the State Lake and Streambed Alteration, Porter-Cologne Water Quality Act, and California Coastal Acts. Most of the altered riparian and wetland habitat was in Ventura County, followed by Los Angeles County (Appendix 5.D, **Table 5.D.2**(a)). For open water, altered areas were concentrated in Orange County, followed by Ventura County. Despite the high intensity of hydraulic fracturing activity in the San Joaquin Valley, there is little impact in terms of increased well density in aquatic habitat because the two do not overlap geographically. Potential impacts to aquatic habitats are discussed further in the chapter in the sections on fluid discharges and water use associated with well stimulation, in Sections 5.3.3 and 5.3.4, below.

Our results should be interpreted with caution, as the resolution of the data on natural communities is coarse relative to the size of a well pad. The natural community data is given on a scale of tens to 400 hectares (from one-tenth to four square kilometers). Well pads for a single well are typically smaller than a tenth of a square kilometer (SHIP, 2014). Therefore, when we find that well density increased in an area of a given habitat type, this may mean that the wells were in the vicinity of these habitat types, but not directly in them.

Question 3*: What special-status species occurred in the vicinity of oil fields highly impacted by well stimulation?*

Under the Federal and California Endangered Species Acts (ESA and CESA), threatened and endangered species, referred to collectively as “listed” species, are entitled to special legal protections. Species are listed as endangered because they are at risk of extinction; they are listed as threatened because they are likely to become endangered. In Table 5.3.5 we identify threatened and endangered species with occurrences recorded in the California Natural Diversity Database (CNDDB) on or within 2 km of oil and gas fields with at least 200 hectares impacted by hydraulic fracturing.

Table 5.3.5. Number of occurrences of listed species within 2 km of a field with at least 200 hectares of altered habitat. Table based on detections of rare species submitted to the California Natural Diversity Database (Biogeographic Data Branch DFW, 2014).

San Joaquin kit fox (*Vulpes macrotis mutica*)	234
Nelson's antelope squirrel (*Ammospermophilus nelsoni*)	189
blunt-nosed leopard lizard (*Gambelia sila*)	78
giant kangaroo rat (*Dipodomys ingens*)	68
Kern mallow (*Eremalche kernensis*)	32
Tipton kangaroo rat (*Dipodomys nitratoides nitratoides*)	15
least Bell's vireo (*Vireo bellii pusillus*)	13
coastal California gnatcatcher (*Polioptila californica californica*)	11
California jewelflower (*Caulanthus californicus*)	4
Bakersfield cactus (*Opuntia basilaris var. treleasei*)	3
California red-legged frog (*Rana draytonii*)	3
giant garter snake (*Thamnophis gigas*)	3
San Joaquin woollythreads (*Monolopia congdonii*)	3
Santa Ana sucker (*Catostomus santaanae*)	3
southern steelhead - southern Calif. DPS (*Oncorhynchus mykiss irideus*)	3
Swainson's hawk (*Buteo swainsoni)*	3
Buena Vista Lake ornate shrew (*Sorex ornatus relictus*)	2
California condor (*Gymnogyps californianus*)	2
Ventura Marsh milk-vetch (*Astragalus pycnostachyus* var. *lanosissimus*)	2
California Orcutt grass (*Orcuttia californica*)	1
slender-horned spineflower (*Dodecahema leptoceras*)	1
southwestern willow flycatcher (*Empidonax traillii extimus*)	1
tidewater goby (*Eucyclogobius newberryi*)	1
unarmored threespine stickleback (*Gasterosteus aculeatus williamsoni*)	1
Total	**676**

An important indicator of valuable habitat is whether it has been designated as critical habitat for the recovery of a federally listed species. Critical habitat should be taken as a conservative indicator of valuable habitat; that is, there are likely to be habitats necessary for the survival of endangered species that have not been designated as critical habitat due to the legal and administrative difficulties in finalizing the process. The United States Fish and Wildlife Service (USFWS) has designated critical habitat for only 44% of all listed species in the U.S.

The only designated critical habitat in the southern San Joaquin Valley is for the Buena Vista Lake ornate shrew. Four small patches on the scale of a few square kilometers each are scattered through the southern portion of the valley in the vicinity of Coles Levee North and South, Buttonwillow Gas, Semitropic, and Semitropic Gas fields. Little to no well stimulation occurs in these fields; the only reported hydraulic fracturing events in these five fields were two in the Semitropic field in 2012 (Volume I, Appendix M) and four notices of planned jobs at Coles Levee North (DOGGR, 2015).

Critical habitat has been designated for a number of species in Ventura County. Areas where substantial amounts of hydraulic-fracturing-enabled production has taken place in the Ojai and Sespe fields overlap with critical habitat for the California condor (*Gymnogyps californianus*) and steelhead salmon (*Oncorhynchus mykiss irideus)* (Figure 5.3.3c).

5.3.2. Human Disturbance Can Facilitate Colonization by Invasive Species

Hydraulic-fracturing-enabled production, like any other oil and gas production, can facilitate the introduction of invasive species, including non-native species (Hobbs and Huenneke, 1992). This occurs because human disturbances such as clearing and levelling land tend to open new niches, and humans and their vehicles can act as vectors for colonizers (Didham et al. 2005). Colonization by invasive species would largely be an indirect impact of well stimulation, given that most of the surface disturbance and vehicle traffic not directly in the service of well stimulation, but there would be some truck traffic that would be directly related to transporting materials and workers to implement a stimulation operation.

Invasive species are defined as non-native organism that reproduce and spread rapidly. They are typically habitat generalists and they frequently displace native species (Rejmánek and Richardson, 1996; Belnap, 2003; Coffin, 2007; Jones et al., 2014). Among plants, these species usually are typical of early successional stages in vegetation communities. Thus, any soil disturbances such as grading, disking, earthmoving, or vegetation clearing result in conditions that favor invasive species (Tyser and Worley, 1992; Gelbard and Belnap, 2003). In oilfields, such activities also can create novel micro-habitats such as borrow areas that collect moisture, berms along roads and around tank settings, and so forth, that provide colonization opportunities for species not native to an area. In the Elk Hills oilfield, the diversity of grasses and forbs (both non-native and native) increased on higher intensity oilfield plots, probably due to the increase in micro-habitats (Fiehler and Cypher, 2011). Also, seeds of species not native to an area are commonly transported in on equipment, vehicles, and boots, further increasing the opportunities for colonization.

Non-native animals also are able to colonize areas disturbed by humans. Rodents such as rats and house mice are common around developments. In western Kern County, Spiegel and Small (1996) found that house mice were extremely abundant in highly developed oilfields, but did not occur in nearby undisturbed habitat. Fiehler and Cypher (2011) found that bird abundance and species richness increased with level of oilfield development. They attributed this to increased contact between areas of intact habitat and human-disturbed areas, increased structural diversity resulting from the presence of facilities such as buildings, facilities, power lines, and pump jacks, and also to increased vegetation diversity both from colonization by non-native plants and landscape plantings. They also found that non-native bird species were more abundant in highly developed areas whereas certain sensitive native species were much less abundant. In the San Joaquin Valley, another potential concern is colonization by non-native red foxes. This species has been increasing in this region, particularly in human-altered areas where its natural predator, the coyote, is less abundant (B. Cypher, CSU-Stanislaus, pers. observ.). Red foxes can compete with and even occasionally kill endangered San Joaquin kit foxes (Ralls and White, 1995; Cypher et al., 2001; Clark et al., 2005).

Occasionally, anthropogenic disturbances can benefit native species, including rare or sensitive species. In western Kern County, a federally threatened plant, Hoover's wooly-star (*Eriastrum hooveri*) quickly colonized disturbed sites and was commonly found on abandoned roads and well pads (Hinshaw et al., 1998; Holmstead and Anderson, 1998). Also in western Kern County, endangered blunt-nosed leopard lizards (*Gambelia sila*) commonly used dirt roads for foraging and movements in areas where dense ground cover impeded such activities (Warrick et al., 1998).

5.3.3. Discharges of Wastewater and Stimulation Fluids Can Affect Wildlife and Vegetation

The discussion in this chapter on discharges of fluids summarizes information presented in Chapter 2, with an expanded discussion of the literature relevant to assessing potential impacts to wildlife and vegetation. We review the potential pathways for release of fluids to the environment, the ecotoxicology of well stimulation fluids and wastewater, and consider the potential impacts of fluid releases to terrestrial, freshwater and marine ecosystems. Discharges of fluids can be a direct or indirect impact of well stimulation. The brines and hydrocarbons produced from the formation are part of any oil and gas production and are considered an indirect impact, while the a release to the environment of a stimulation fluid is a direct impact of well stimulation.

5.3.3.1. Potential Pathways for Release of Fluids to the Environment

Discharges of fluids related to well stimulation can occur intentionally through discharges of waste products to the surface, or by accidental spills and leaks. Chapter 2, Figure 2.6.1 shows surface (and near-surface) contaminant release mechanisms of concern in California related to stimulation, production, and wastewater management and disposal activities. The additives for stimulation fluids and proppant are typically transported by truck to a stimulation site (see Chapter 2, 2.4.3, "Evaluation of the Use of Additives in Stimulation Fluids," for more detail). They are diluted with water and injected into the stimulated well. Some portion of the stimulation fluids returns to the surface, mixed with hydrocarbons, formation water and possibly well clean-out fluids (see Chapter 2 Section 2.5.2, "Description of Wastewaters Generated by Well-Stimulation Operations").

The fluid produced from a well that remains after the marketable hydrocarbons are separated out is referred to as wastewater. For the purposes of this report, we are interested in any release of stimulation fluids to the environment as a direct impact of well stimulation. We are also interested in discharge of wastewater from stimulated wells to the environment, whether or not it contains stimulation fluids, as an indirect effect of well-stimulation enabled production.

Stimulation fluids and wastewater can potentially come into contact with wildlife and vegetation in a number of ways. Accidental releases can occur at any stage of the process, from transport of chemicals to the site, at the site during a stimulation operation, through an underground pathway, or once the fluids return to the surface after well completion. Wastewater can also be legally discharged to the terrestrial or freshwater environment under certain conditions to unlined surface pits, used for groundwater discharge, or applied to agricultural land for irrigation. In federal waters, treated wastewater can legally be discharged to the ocean.

5.3.3.1.1. Exposure to Stimulation Fluids and Wastewater in Land and Freshwater Ecosystems

Potential routes of environmental exposure to hydraulic fracturing chemicals include accidental spills and intentional discharges to surface storage ponds. Outside of California, Bamberger and Oswald (2012) documented a number of observations of harm to livestock, domestic animals, and wildlife that correlated with surface spills or intentional surface applications of wastewater from hydraulically fractured wells; however, these case studies were analyzed retrospectively through interviews, veterinary reports and other sources, and did not distinguish hydraulic fracturing flowback from produced water, so they cannot be taken as definitive evidence of direct harm from hydraulic fracturing operations.

Wildlife can suffer negative effects or mortality by drinking from or immersing themselves in wastewater storage or disposal ponds (Ramirez, 2010; Timoney and Ronconi, 2010). In the limited studies available of ecological impacts of oil field activity in California, there are a few documented cases of giant kangaroo rats, blunt-nosed leopard lizards and San Joaquin kit foxes drowning in accidental spills of oil and oil-laden wastewater (Kato and O'Farrell, 1986; O'Farrell and Kato, 1987). Suter et al. (1992) examined the elemental content of fur samples from San Joaquin Kit Foxes inhabiting two oil fields (one active, one inactive), and two control areas. They found that foxes on the developed sites had elevated levels of a number of elements which may be attributable to oil field materials. However, their results must be interpreted with caution because of flaws the authors themselves acknowledge in sampling design and statistical methods.

As described in Chapter 2, Section 2.5, discharge of wastewater to percolation pits, also called evaporation-percolation ponds, is the most commonly reported disposal method for stimulated wells in California. Percolation pits are primarily regulated by the state's nine

Regional Water Quality Control Boards. Much of the state's well stimulation takes place within the jurisdiction of the Central Valley Regional Water Quality Control Board. Within its jurisdiction, wastewater can legally be disposed of in percolation pits with a permit from the regional water board. However, it was recently found that an estimated 36% of sumps have been operating without the necessary permits (Holcomb, 2015). The Central Valley Regional Water Board requires that the fluid in the pits meet certain water quality standards for salinity (measured as electrical conductivity), chlorides, and boron. Oil field wastewater that exceeds the salinity thresholds may be discharged in percolation pits, or to local streams or ponds "if the discharger successfully demonstrates to the Regional Water Board in a public hearing that the proposed discharge will not substantially affect water quality nor cause a violation of water quality objectives." There is no testing required, or thresholds specified, for other contaminants. However, oil field wastewater typically contains other chemicals such as volatile organic compounds (VOCs), benzene, and naturally occurring radioactive material (NORM) that are of concern for human and environmental health.

Based on information obtained from the Central Valley Regional Water Quality Control Board and the State Water Resources Control Board, there are 950 known evaporation-percolation ponds in eight California counties, listed in Table 5.3.5 (Borkovich 2015a and b, CVRWQCB 2015). In Kern County, there were 484 active pits, 221 inactive, and 138 of unknown status, for a total of 843. There were no sump locations in Ventura County in the datasets we obtained. However, these datasets must be treated with caution as likely representing a minimum, but not necessarily a comprehensive list of percolation pit locations in the state. Chapter 2 discusses the caveats for these datasets.

Table 5.3.5. Reported sump locations in California. Locations were coded by status: active indicates that the location contained produced water, inactive sumps were empty, and the rest are unknown status. Data from CVRWQCB 2015 and Borkovich 2015a and 2015b (Appendix Chapter 2, 2.G).

	Status			
County	**Active**	**Inactive**	**Unknown**	**Total**
Kern	484	221	138	843
Fresno	31	16		47
Tulare	30			30
Santa Barbara	9	4		13
Kings	9			9
San Benito	1	3	1	5
Monterey	1	1		2
San Luis Obispo	1			1
Grand Total	**566**	**245**	**139**	**950**

To reduce access to sumps by animals, California regulations require that any pond containing oil or a mixture of oil and water must be covered with a net with no more than a two-inch mesh (California Code of Regulations Title 14 § 1770 on Oilfield Sumps). Ponds not containing oil are not subject to such a requirement. We used the reported locations of percolation pits gathered by the Central Valley Regional Water Quality Control Board to plot the locations of sumps in Google Earth and survey the pits for nets. We randomly selected 200 sumps to survey. Of these, 114 contained fluid at the time the aerial photograph for Google Earth was taken. Twenty-seven of the 114 pits in use (24%) were covered with nets. We could not determine whether unnetted pits had trace oil in the water or whether they all met the legal requirements to be unnetted. Nonetheless, other constituents besides oil could impact the health of organisms that come in contact with the sumps, particularly if the produced water contains traces of stimulation chemicals.

While there are at least 950 known sumps in eight counties, not all of these have necessarily received produced water from stimulated wells. As discussed in detail in section 2.5.3.3 of this volume, “Management of Produced Water,” and 2.6.2.1, “Use of Unlined Pits for Produced Water Disposal,” reports of disposal of wastewater specifically from stimulated wells to unlined pits was limited to Kern County and was associated with the Elk Hills, South Belridge, North Belridge, Lost Hills, and Buena Vista fields. Very few operators are discharging wastewater from stimulated wells to creeks or streams, with two stimulated wells reported to be discharging a total of 2,060 m^3 (2 acre-feet) of wastewater into surface water bodies during the first full month following stimulation.

As described in depth in section 2.6.2.9 of this volume, “Spills and Leaks,” there are two databases maintained by the state on spills of oil and produced water, one by DOGGR and one by California Governor’s Office of Emergency Services (OES). The OES database also documents chemical spills on oil fields. Neither dataset provides information, such as an American Petroleum Institute (API) number, that would allow a spill to be associated with a stimulated well. The databases also do not give precise identification nor concentrations of the chemical constituents of spilled substances, giving very general descriptions such as “produced water” or “acid” that do not allow evaluation of the ecological impacts. Between January 2009 and December 2014, a total of 575 produced water spills were reported to OES, or an average of about 99 spills annually. The majority (55%) of these spills occurred in Kern County, followed by Los Angeles (16%), Santa Barbara (13%), Ventura (6%), Orange (3%), Monterey (2%), and San Luis Obispo (1%), and Sutter (1%) counties. Nearly 18% of these spills impacted waterways. Chemical spills were also reported in California oil fields, including spills of chemicals typically used in well stimulation fluids, e.g., hydrochloric, hydrofluoric, and sulfuric acids. Between January 2009 and December 2014, a total of 31 chemical spills were reported to OES. Forty-two percent of these spills were in Kern County, followed by Los Angeles (16%), Sonoma (16%), and Lake (3%) counties. Chemical spills represent about 2% of all reported spills attributed to oil and gas development during that period. 10% of the chemical spills were reported to enter a waterway.

At present there is insufficient data available to determine the concentration and volume of the chemical constituents in wastewater intentionally and accidentally released to the environment. The impact to the environment will depend on a multitude of unknown factors including the volume and chemical content of the wastewater, how it is treated, where it is released, and transformations in the environment.

5.3.3.1.2. Discharges to the Ocean

Although ocean discharge from platforms in State waters (within 3 nautical miles of the coast) is prohibited, platforms operating in federal waters off California's coast are legally allowed to discharge treated produced water which may contain flowback containing stimulation chemicals to the ocean. Chapter 2 Section 2.5.3.3.2, "Wastewater from Offshore Oil and Gas Operations," describes the scope of the discharge and the regulations on its volume, composition, and monitoring. Accidental discharge of fluids to the ocean is also possible, although we are not aware of any data indicating that the rate of accidental spills, such as blowouts, differs for stimulated and unstimulated wells. As such, the main difference between a spill from a stimulated versus unstimulated well would be the potential presence of stimulation fluids. The potential impacts to the marine ecosystem of intentional and accidental discharge will be examined in-depth in the Volume III Offshore Case Study.

5.3.3.2. Ecotoxicology of Well Stimulation Fluids and Wastewater

Adverse impacts on wildlife and vegetation can result from exposure to chemicals in stimulation fluids and wastewater from stimulated wells. The data on the chemical content of these substances is discussed in depth in Vol II Chapter 2 Sections 2.4, "Characterization of Well Stimulation Fluids," and 2.5.4, "Wastewater Characteristics." In that chapter, environmental hazards of well stimulation additives and wastewater were evaluated in detail with respect to acute and chronic toxicity, bioaccumulation, and environmental persistence. In this section, we briefly revisit the topic with a focus on potential impacts to wildlife and vegetation if organisms are exposed to these fluids. However, our understanding of the long-term impacts of low-level exposure to these chemicals is limited, because much of the information on toxicity to organisms is collected in the laboratory using relatively high concentrations of individual chemicals. Impacts to organisms from a release of well stimulation and/or wastewater to the environment will depend on the actual concentration of chemicals and the reactions they undergo in the environment. In addition, standard toxicity tests are conducted on a limited suite of organisms that may not reflect the biology of California's native biota (see Vol II Chapter 2 Section 2.4.4.4, "Characterization by environmental toxicity," for more detail).

5.3.3.2.1. Stimulation Fluids

Exposure to chemicals used in well stimulation has been shown to adversely affect mammals, fish, invertebrates and algae in acute toxicity tests. Environmental toxicity of stimulation fluids is discussed in depth in Volume II Chapter 2 Section 2.4.4.4, “Characterization by Environmental Toxicity” and Section 2.4.7, “Other Environmental Hazards of Well Stimulation Fluid Additives.”

5.3.3.2.2. Inorganics in Wastewater

Wastewater from stimulated wells is made up of a mixture of stimulation fluids, formation fluids, and well clean-out fluids (see Chapter 2 Section 2.5.2, “Description of Wastewaters Generated by Well-Stimulation Operations).” Some inorganic chemicals in underlying rock formations that are brought to the surface through oil and gas production can be hazardous to wildlife and vegetation. Some geologic formations associated with well stimulation activity in California contain relatively high levels of trace elements and radionuclides (Piper et al., 1995; Presser et al., 2004). Inorganics mobilized by well stimulation may pose a risk to California wildlife and vegetation. Selenium enrichment is particularly problematic in the western San Joaquin Valley, including Kern County (Presser and Ohlendorf, 1987). Selenium exposure can cause developmental toxicity in birds and fish at environmentally relevant levels (Presser and Barnes, 1985). Several other trace elements (e.g., Cd, Cu, Ni, V) that are enriched in well stimulation areas are known to cause adverse effects in wildlife and vegetation at environmentally relevant levels (e.g., Eisler 1998; Larison et al., 2000; Rattner et al., 2006; Shahid et al., 2014). Formation water is also typically high in salt content; many plants and aquatic organisms in particular are highly sensitive to salt concentrations (Allen et al., 1975; Pezeshki et al., 1989; Ruso et al., 2007). Among the metals copper, selenium, titanium and vanadium are the most likely to accumulate (Love et al., 2013). Persistence, biodegradation, and bioaccumulation are discussed in more detail in Chapter 2 Section 2.4.7.1, “Environmental Persistence.”

A major gap in knowledge of the ecotoxicology of stimulation fluids and associated wastewater is how the number of toxic and/or persistent compounds already used in well stimulation fluids might alter the toxicity and persistence of the chemical compounds in produced waters. The literature on possible additivity and synergistic interactions of persistent/toxic compounds is scarce and a proper risk assessment of chemical mixtures is currently hampered by the lack of data (Martins et al., 2009; Shen et al., 2006; Stelzer & Chan, 1999; Pellacani et al., 2012).

5.3.3.2.3. Hydrocarbons in Wastewater

Produced water generally contains a number of soluble hydrocarbons, along with metals and other compounds used in well treatment (Benko & Drewes, 2008; Clark & Veil, 2009). In California most information on produced water in the marine environment is from oil production facilities in the Santa Barbara Channel. Most of the toxicity of produced water is attributed to the water-soluble fractions of the hydrocarbons (Garman, et al., 1994). At a well blowout site in Kern County, Kaplan et al. (2009) found evidence that Heermann’s kangaroo rats (*Dipodomys heermanni*) incorporated into their livers a set of chemicals, polycyclic hydrocarbons, that originated from crude oil.

5.3.3.3. Summary of Impacts of Discharges of Stimulation Fluids and Wastewater to Wildlife and Vegetation

When handled without accident, wastewater can be either reused or disposed of. One type of reuse involves re-injecting produced water into the formation to enhance oil recovery and counteract subsidence. Occasionally wastewater is used for irrigation or industrial purposes. Alternatively, wastewater may be disposed of in pits or injection wells, referred to as Class II wells in the USEPA's Underground Injection Control Program. A very small amount is disposed of by discharging it directly into the ocean. No matter how wastewater is reused or disposed of, there is the potential for spills and environmental releases of chemicals used in the well stimulation process. Laws and regulations seek to minimize the occurrence and consequences of environmental releases of inadequately treated fluids, however releases of chemicals to the environment can and do occur. Chapter 2 of this volume analyzed the potential effects of these releases by considering the toxicity of the most commonly used chemicals for well stimulation, and the chemicals used in the greatest mass. The evaluation considered toxicity of relatively high concentrations of the chemical, and therefore represents a worst possible scenario. In practice, the chemicals are often diluted or removed by treatment practices before fluids are released to the environment.

Our understanding of the impacts of discharges of stimulation fluids and wastewater to wildlife and vegetation is hampered by lack of data on multiple levels. Based on ecotoxicology data on stimulation fluids and wastewater, we can state that the discharge of stimulation fluids and wastewater from stimulated wells has the potential to harm wildlife and vegetation, but the actual magnitude of the impacts will depend on the frequency, location, volume, and chemical concentrations of discharges. We lack substantive data on the frequency of releases, the volumes and concentrations of discharges, and the long-term impacts on wildlife and vegetation once the fluids enter the environment. More is known about the potential indirect impacts of inorganics and hydrocarbons in formation waters and production fluids than the direct effect of stimulation fluids. Mammalian wildlife can be more susceptible to adverse effects of inorganics and hydrocarbons due to higher exposure levels than the human population. Increased data collection on potential releases of stimulation fluids and wastewater to the environment and refinement of the ecotoxicological analysis would lead to a better understanding of this risk.

5.3.4. Use of Water Can Harm Freshwater Ecosystems

Water is the main constituent of stimulation fluids, and water use to make stimulation fluids is a direct impact of well stimulation. Well stimulation can also in some situations enable production from reservoirs that also require enhanced oil recovery for effective production (EOR). Common forms EOR such as water flooding, steam flooding, and cyclic steaming require. Use of water for EOR enabled by hydraulic fracturing is an indirect impact of well stimulation. Competition for water with human uses is a major cause in the alteration and decline of the state's aquatic ecosystems (Moyle and Leidy, 1992). Water use for well stimulation is discussed in detail in Chapter 2 Section 2.3, "Water Use for Well Stimulation in California." Water for well stimulation is a small fraction of freshwater used in the state. Chapter 2 reports that well stimulation in the state uses 850,000 to 1,200,000 m^3 (690–980 acre-feet) annually; this is about 0.01% (one ten-thousandth) of California's annual human water use. Even factoring in EOR enabled by well stimulation, the proportion of water use for both well stimulation and well stimulation – enabled EOR is 0.03% (three ten-thousandths) of annual human water use in the state. However, well stimulation is a highly geographically clustered activity, so it is important to consider water use in a regional context. Chapter 2 looks at water use for well stimulation and EOR enabled by well stimulation within planning areas. There are 56 planning areas in the state, ranging in size from 320 to 7,500 square miles, with an average size of 2,600 square miles. The planning area with the largest proportion of its water used by well stimulation and EOR enabled by well stimulation is the Semitropic Planning Area in the western portion of Kern county. In the Semitropic, .19% of the annual water use, or 2,900,000 m^3, is for well stimulation and EOR enabled by well stimulation. Thus, even in the region where most of the well stimulation in the state occurs, it represents a small proportion of total water use.

The statistics on water use for well stimulation on a state-wide and regional scale indicate that well stimulation represents a small percentage of water diverted from large sources. Of the 495 well stimulation completion reports filed with DOGGR between January 1 and December 10, 2014, all but two were for operations in Kern County. Most of the Kern County operations (397, or 83%) used water from the Belridge Water Storage District, which sources water from the State Water Project. The State Water Project delivers about 470 million m^3 (2.3 million acre-feet) in average years, which dwarfs the amount of water used for well stimulation; as a result, a very small proportion of the impact to ecosystems by the State Water System can be attributed to withdrawals for well stimulation.

The available data on water use for stimulation does not allow us to do is to determine whether water diversions for well stimulation cause very small-scale, local impacts on surface waterways. The main pathway for water use to impact the health of an ecosystem is if water use is a large proportion of streamflow for a surface waterway, or if groundwater is drawn down locally so that it substantially decreases baseflow to a stream. While water use for well stimulation is of a small enough volume that it is unlikely to have a substantial impact on large bodies of water, it is conceivable that an operator could divert a large proportion of a small waterway or locally draw down groundwater enough to affect small bodies of surface water. In order to understand very local impacts such as these, data on the source of well stimulation water would need to be reported on a finer spatial scale than it is at present. In well stimulation disclosures, operators report the source of water by category such as irrigation districts (68%), produced water (13%), operators' own wells (13%), a nearby municipal water supplier (4%), or a private landowner (1%). This level of reporting does not allow us to establish if, for example, a proportionately large amount of water is being withdrawn from the groundwater by private wells in one small area, or diverted from a small surface waterway.

5.3.5. Noise and Light Pollution Can Alter Animal Behavior

Oil and gas operations are sources of anthropogenic noise caused by equipment and night-time lighting. Some noise is generated by the equipment used specifically for well stimulation, chiefly the hydraulic fracturing pumps, and would be considered an indirect impact. Noise is also generated at other stages of process such as site preparation, drilling, and production and would be considered an indirect result of well stimulation. Night-time lighting for production enabled by stimulation would be an indirect impact. Well stimulation operations typically last on the order of hours (King, 2012), so the duration of noise and light directly caused by well stimulation is brief compared to the months to years of noise and light associated with ensuing production.

Noise and artificial night lighting have been shown to effect the communication, foraging, competition, and reproduction of organisms. Sound is an important sensory tool for animals and noise pollution from oil and gas production has been shown to alter their behavior, distribution, and reproductive rates (Blickley et al., 2012a and b; Francis et al., 2012). Noise is generated at all stages of the oil and gas production process, from construction of the well, stimulation, and production, until the well is abandoned. We could find only one reported measurement of noise specifically during hydraulic fracturing in California. Noise levels of 68.9 and 68.4 decibels (dBA) were measured 1.8 m (5 ft) above the ground 33m (100 ft) and 66 m (200 ft) away from a high-volume hydraulic fracturing operation in the Inglewood Field (Cardno ENTRIX, 2012). These levels are substantially lower than those found to disturb wildlife and ecosystem processes in Blickley et al. (2012a and b) and Francis et al. (2012). Observational data collected in the Elk Hills region of western Kern County between 1980 and 2000 suggested that the San Joaquin kit fox and other wildlife appeared to have habituated and acclimated to the regimen of noise, ground vibrations, and human disturbances associated with an active oil field (O'Farrell et al., 1986).

Ecological light pollution is a specific term describing chronically increased illumination and temporary unexpected fluctuations in lighting (Longcore and Rich, 2004). Sources of ecological light pollution include lighted buildings, streetlights, security lights, vehicle lights, flares on off-shore oil platforms, and lights on well pads. Light pollution has been shown to extend diurnal or crepuscular foraging behaviors (Hill, 1990; Schwartz and Henderson, 1991), reduced nocturnal foraging in desert rodents (Kotler, 1984), disorient organisms who hatch at night such as sea turtle hatchlings (Salmon, 2003; Witherington, 1997) and disorient nocturnal animals such as birds (Ogden, 1996) and frogs (Buchanan, 1993) leading to mortality or predation. Many studies have also noted changes of breeding and migration behaviors (Rydell, 1992; Eisenbeis, 2006; Stone et al., 2009; Titulaer et al., 2012; Bergen and Abs, 1997). Ecological light pollution can also disrupt plant by distorting their natural day-night cycle (Montevecchi et al., 2006). It is considered an important force behind the loss of light-sensitive species and the decline of nocturnal pollinators such as moths and bats (Potts et al., 2010) and can change the composition of whole communities (Davies et al., 2012).

There are no specific studies on the effect of artificial lighting on wildlife on or around well pads, however, some states like Maryland have implemented best management practices for oil and gas development to mitigate any potential effects. These include using only night lighting when necessary, directed all light downward, and using low pressure sodium light sources when possible (Maryland Department of the Environment and Maryland Department of Natural Resources, 2014).

5.3.6. Vehicle Traffic Can Cause Plant and Animal Mortality

Vehicles impact natural habitats by striking and killing animals; vehicles traveling off-road can cause plant mortality and compact the soil. The proppant, and occasionally water, required for well stimulation is transported via trucks; vehicles are also an integral piece of equipment in all other stages of oil and gas production. Road mortality is noted as a major factor affecting the conservation status of two state and federally listed species in California known to occur on the oil fields of the San Joaquin Valley: the San Joaquin kit fox and the blunt-nosed leopard lizard (Williams et al., 1998). Vehicle traffic is inherent in most stages of the oil and gas production process, including, but not limited to stimulation; therefore it is both a direct and indirect impact.

Road mortality on oil fields has specifically been studied in the San Joaquin kit fox. In one study at the Elk Hills Oil Field, the proportion of San Joaquin kit fox deaths due to road accidents was four times greater in developed areas versus in undisturbed areas (O'Farrell et al., 1986). A later study at the same field found vehicle-related mortality rates for endangered San Joaquin kit foxes were approximately double in oil-developed areas versus non-developed areas, although overall rates were considered low (20 of 225 deaths during 1980-1995; (Cypher et al., 2000). Similarly, (Spiegel and Disney, 1996) found that none of 29 foxes found dead during 1989-1993 in the highly developed Midway-Sunset and McKittrick-Cymric oilfields had been killed by vehicles. Restrictions on speed limits and off-road driving that are imposed in many oil fields as a measure to mitigate vehicle strikes may explain the low mortality rates.

5.3.7. Ingestion of Litter Can Cause Condor Mortality

As with many sites of human activity, oil and gas pads can become deposits for litter. While there may be marginally more litter as a result of the process of preparing a site for production taking slightly longer and requiring more staff when stimulation is involved, litter is presumably mainly an indirect impact that is associated with all stages of the hydrocarbon production process, not just well stimulation.

Critical habitat for the California Condor overlaps with the Sespe Oil Field in the Los Padres National Forest, and the Sespe Condor Sanctuary is adjacent to the oil field of the same name. U.S. Forest Service guidelines that well pads be maintained free of debris. Nonetheless, oil operations are nonetheless potential sources of microtrash that can cause mortality in condors (Mee et al., 2007a and b; USFWS, 2005). Microtrash consists of any man-made item that is sufficiently small to be ingested by a condor, up to about 4 cm in diameter. Items found in condors have included nuts, bolts, washers, copper wire, plastic, bottle caps, glass, and ammunition cartridges (Mee et al., 2007a and b; Walters et al., 2010). For reasons that are unclear, adults will collect such items and feed them to nestlings (Mee et al., 2007a and b; Rideout et al., 2012). Of 18 nestlings for which cause of death could be determined, 8 (44%) deaths were attributable to microtrash ingestion (USFWS, 2013). The national forest, the U.S. Bureau of Land Management (BLM) (which administers the mineral rights in the forest), and the USFWS all have imposed measures to minimize or eliminate the presence of microtrash (USFWS, 2005).

5.3.8. Potential Future Impacts to Wildlife and Vegetation

In this report we predict that the main focus for hydraulic fracturing in the state will continue to be in and around the areas where it is already used, principally the southwestern San Joaquin Basin (Volume I Chapter 4). The possibility of a sudden development of new areas with hydraulic fracturing-enabled production hinges largely on the possibility of developing Monterey source rock, which is a highly uncertain possibility at this stage. Here we briefly summarize what we know and the data gaps about potential future well stimulation impacts to wildlife and vegetation and refer the readers to the relevant sections of other volumes for more detail.

- Hydraulic fracturing will likely continue to be an important part of oil and gas production in California. In this report we predict that it will continue in and around the fields where it is already routinely used, principally in the San Joaquin Valley (Volume I Chapter 4, Volume II Chapter 5). However, we cannot predict the future location and density of hydraulically fractured wells. As a result we refrain from making detailed forecasts about future habitat loss and fragmentation caused by hydraulic fracturing.

- The degree to which new development will affect habitat loss and fragmentation will depend on whether future development is “infill” (an increased density of already-developed areas) or expansion (growth in undeveloped areas), and the degree to which wells and other infrastructure are clustered or evenly distributed across the landscape. Volume III Chapter 5 examines production as a function of well density in one pool of the Lost Hills oil field and concludes that production increases linearly with well density, suggesting that operators will continue to drill new wells in already-developed areas to increase total yields. The lease with the highest yield in the Cahn pool has a well density of approximately 200 wells per km^2; we would predict that, as long as the activity remains profitable, the remainder of this pool will reach similar densities. A study in another San Joaquin oil field found that native species disappeared at well densities of about 100 wells per km^2 (Fieler and Cypher, 2011). We do not know if all hydraulically fractured pools show a similar linear relationship between yield and well density, but the study of the Cahn pool suggests a possible way to examine this question on a pool-by-pool basis in future research.

- We do not know the limit of the surface footprint of pools requiring hydraulic fracturing. Volume III, Chapter 5 examines two pools in detail, the Cahn Pool at Lost Hills field and the Pyramid Hill-Vedder pool in Mount Poso field, and notes that there is a mix of curved and linear borders of wells producing from these pools. The linear borders suggest that development was limited by a legal boundary (such as a lease) and that the geological resource extends further. This suggests that there are untapped resources just beyond the reach of existing wells that can be developed in the future with the application of hydraulic fracturing.

- While we identify potential pathways for impacts of well stimulation to wildlife and vegetation besides habitat loss and fragmentation in this chapter, the available information is insufficient to quantify past or future impacts to populations. For example, while we know that the release of stimulation chemicals is a possible impact, we do not know to what degree it occurs nor whether it causes declines in population sizes. Without adequate information on past and present impacts, we cannot hope to predict the future impacts.

- It is possible that hydraulic fracturing could open large new areas for development if operators learned how to effectively develop Monterey source rock, although these areas would still be in the general vicinity (within 20 kilometers) of existing oil fields in the six largest oil-producing basins in the state (Volume III Chapter 3). At present there is no reliable resource assessment of Monterey source rock. Based on the documented challenges in developing Monterey source rock, economic production of Monterey source rock appears to be a remote possibility at present, and one which would require technological innovations that may change the profile of impacts from oil and gas production (such as greater reliance on clustered, horizontal wells). Because of these many uncertainties, we did not perform a detailed prediction of future well density in the Monterey source rock footprint, although we did examine the biological resources present in the area to consider the environmental context in which the development could occur. The footprint of Monterey source rock is in the San Joaquin, Ventura, Los Angeles, Salinas, Santa Maria, and Cuyama basins. Within the footprint, about 60% of the area is used intensively by people (i.e. for cities, agriculture, or industry), and about 40% is open space (grass and shrublands, forest, and open water). The footprint of potential Monterey source rock underlies the area of the southwestern San Joaquin identified as highly sensitive in this chapter.

5.4. Laws and Regulations Governing Impacts to Wildlife and Vegetation from Oil and Gas Production

While the preceding has outlined the major potential hazards to wildlife and vegetation, the degree to which these hazards actually impact wildlife and vegetation is mitigated to some extent by the numerous federal and state laws governing how human activities such as well stimulation must be carried out to minimize impacts on wildlife and vegetation.

For example, the National Environmental Policy Act (NEPA), the Federal Endangered Species Act (ESA), the Migratory Bird Treaty Act, the California Endangered Species Act (CESA), California Fully Protected Designations, and the California Environmental Quality Act (CEQA) are directed at protecting the natural environment. In this section, we briefly review regulations applicable in California in order to describe the regulatory system as it pertains to impacts to wildlife and vegetation of oil and gas production and well-stimulation-enabled oil and gas production.

A detailed description of the regulatory setting for biological resources in California is given in the SB4 Draft Environmental Impact Report (Aspen Environmental Group, 2015a and 2015b). However, the pertinent laws do not consistently establish practices that all California oil and gas producers must enact to reduce their impacts on wildlife and vegetation. The relevant laws are brought to bear differently depending on which agencies have jurisdiction over the project and site-specific circumstances. This results in a patchwork of agreements that are not necessarily consistent with one another on a statewide or even regional scale, and that are not compiled in one central repository that is publicly available, but rather exist in the records of a multitude of federal, state, and local agencies, and the private entities who entered into the agreements. For example, Occidental Petroleum and the California Department of Fish and Game[6] entered into a memorandum of understanding and take authorization governing activities at Elk Hills oil field (California Department of Fish and Game, 1997). This document does not apply to any of the other fields in the state.

The process by which environmental regulations are applied to minimize impacts to wildlife and vegetation varies depending upon the landowner and the mineral rights owner at a given location. Not uncommonly, a “split estate” situation exists whereby the owner (s) of the land and the owner (s) of the mineral rights beneath that land are different. If the land or mineral rights are federally owned, then the process is more consistent. In these situations, the federal agency that owns the surface and/or mineral estate must authorize any oil and gas development projects and grant permits. These actions necessitate formal review of the proposed project under NEPA. The federal action agency, often with a project description and site-specific information provided by the project proponent, prepares an Environmental Assessment or Environmental Impact Statement under NEPA to analyze the effects of the project. Appropriate terms and conditions are attached to the federal authorization to avoid or mitigate project effects on natural resources.

Ideally, this document describes how the project will comply with all applicable environmental laws. Also, the federal agency is responsible for ensuring that the project proponent complies with all applicable laws and regulations (see Aspen Environmental Group 2015a and b for a list of applicable laws and regulations).

6. Now known as the California Department of Fish and Wildlife.

If the land and mineral rights are privately owned, then the process depends upon the nature of the proposed project. If the project is to drill a new well, the well must be permitted by DOGGR. Before DOGGR can issue a permit, the project is required to be subjected to review under CEQA. The project proponent prepares an Environmental Impact Report, and this is the document that is subject to review. Ideally, this document describes how the project will comply with all applicable environmental laws. If the project is something other than a new well (e.g., construction of infrastructure such as pipelines, facilities, etc.), then the responsible agency usually is a county or local municipality. The requirements and process are then very variable with some agencies providing little to no requirements or oversight with regards to environmental regulations, and others imposing rigorous requirements and oversight. Even when agency oversight is minimal or non-existent, project proponents still are required to comply with all laws and regulations, but such compliance tends to be variable.

Given the patchwork of regulatory agreements pertaining to oil and gas activities throughout the state and the lack of any centralized collection for such agreements, it is not possible for us to fully evaluate the regulations that the various oil and gas operators may or may not be operating under, nor evaluate the degree to which these agreements are consistent or complementary with one another. We emphasize that the lack of consistency in the application of regulatory requirements is in no way unique to oil and gas operations, but instead is common to all activities evaluated under the acts listed at the beginning of this section. The requirements tend to vary among habitats, species, agency staff conducting the evaluations, and precedents established among offices within agencies. Finally, requirements for a given oil and gas project may vary depending upon whether the project was initiated before or after a given regulatory act was passed and implemented.

5.5. Measures to Mitigate Oil Field Impacts on Terrestrial Species and Their Habitats

The potential hazards to wildlife and vegetation posed by well stimulation and the production it enables can be reduced through application of the appropriate mitigation measures. A variety of measures are frequently required in oil fields in California to avoid or mitigate impacts to terrestrial species and their habitats resulting from oil and gas extraction activities. To our knowledge, no mitigation measures for the protection of terrestrial species and their habitats are specific to well-stimulation activities, but apply to oil and gas production activities that can be enabled by well stimulation such as construction of well pads, roads, facilities, and pipelines; maintenance and operations; and seismic surveys.

The list of measures presented in this section is largely derived from examples in the San Joaquin Valley, where oil field activity is extensive and where sensitive biological resources are abundant (see Introduction, Section 5.2, for synopsis of San Joaquin Valley biological values). Measures implemented in other regions probably are similar with nuances specific to the species and habitats in those regions.

Below, we list and describe commonly implemented mitigation measures in oil fields. This list was compiled from documents that addressed oil and gas production in large oil fields or over large regions. The primary documents were U.S. BLM, 2010; U.S. DOE, 1991; 2001; US DOI, 2012; USFWS, 2001. The documents used in this compilation addressed large, extensive oil and gas production operations conducted over multiple years. All of the information presented below was distilled from the sources above unless otherwise cited. The measures are grouped into broad categories based on their intended purpose. Here we focus principally on impacts to the terrestrial environment; the alternative and best practices given in Volume II, Chapter 2 focus on strategies for reducing risks to water supply and quantity that can impact the aquatic environment.

5.5.1. Habitat Disturbance Mitigation

5.5.1.1. Compensatory habitat

In an effort to compensate for habitat destruction resulting from oil field activities, project proponents commonly are required to permanently conserve undisturbed habitat elsewhere. Such habitat is referred to as “compensatory habitat.” This requirement can be satisfied by project proponents in various ways including using lands they already own, purchasing lands, and purchasing credits in an approved habitat mitigation bank. For lands owned or purchased, the project proponent can retain and manage the lands, or transfer them to a natural resources agency (e.g., CDFW) or an approved conservation organization (e.g., Center for Natural Lands Management). The lands must be protected in perpetuity and managed appropriately. Agency-approved management plans typically are required for lands retained by project proponents, and endowment funds for management must be provided along with lands transferred to another agency or organization.

This approach to mitigation uses what are generally referred to as “environmental offsets,” and has become a common form of environmental regulation in the United States and Europe. The goal of offsets is to counteract the impact of development to achieve a net neutral or beneficial outcome. For example, beginning in the 1970s, most states adopted a “no net loss” policy for wetlands. Rather than banning all development in wetland areas, developers were given the option of compensating for wetland loss by creating new wetlands elsewhere on an acre-for-acre basis. The mitigation approach is not without its detractors, however; see e.g. McKenney (2005), Race and Fonseca (1996).

For California oil and gas projects, the ratio of compensatory land to altered land is variable. In the San Joaquin Valley, a common ratio is 3:1, meaning three units of compensatory habitat for every one unit of habitat disturbed. For “temporary” habitat disturbances (usually defined as disturbances lasting less than two years), the ratio is 1.1:1. Examples of temporary disturbances include the installation of buried pipelines and equipment staging areas. In such situations, the disturbed area is allowed to revegetate through natural or active habitat restoration, and then is again available for use by species. Other ratios have been required, including 4:1 in cases where protected lands are disturbed (USFWS, 2001). (Many lands in the San Joaquin Valley are “split estates” in which one party owns the surface of the land and another party owns the mineral rights underlying the land. In such situations, access to the minerals must be granted. Thus, mineral extraction activities are not uncommon on protected lands.) A ratio of 6:1 was required for any projects that disturbed habitat for federally endangered Kern Mallow (*Eremalke kernensis*; USFWS, 2001). In the case of an oil field waste-processing facility constructed in highly sensitive habitat used by multiple listed species in Kern County, the required ratio was 19:1 (D. Mitchell, Diane Mitchell Environmental Consulting, personal communication).

Compensatory habitat is typically "in kind;" that is, the habitat must be of equal or higher value than the habitat that was disturbed. Furthermore, listed species present on the disturbed habitat also must be present on the compensatory habitat.

5.5.1.2. Disturbance minimization

Measures commonly are implemented to reduce the amount of habitat disturbed by oil-field activities. Some of the measures are implemented in the planning phase of a project (e.g., planning to drill multiple wells from a single pad). Other measures constitute best management practices implemented during the construction or operations phases.

- Use existing roads to the extent possible.
- Use previously disturbed areas to the extent possible.
- Try to aggregate facilities to the extent possible.
- Drill multiple wells from a single pad by using directional and horizontal drilling.
- Route pipelines along existing roads whenever possible.
- Elevate pipelines to minimize surface disturbance and allow animals to freely move under the pipeline.
- If off-road travel is necessary and permitted (e.g., seismic surveys), use all-terrain vehicles (ATVs) instead of full-sized vehicles when possible for cross-country travel, as ATVs are smaller and lighter and therefore cause less damage when driven across habitat.

In some situations, the total habitat disturbance permitted in a given area is restricted. Lands administered by the U.S. BLM in the southern San Joaquin Valley have been categorized based on the suitability of the lands for listed species. In "Red Zones," which are within identified reserve areas, surface disturbance from oil and gas extraction activities may not exceed 10%. In "Green Zones," which are identified as dispersal corridors between reserve areas, surface disturbance cannot exceed 25% (USFWS, 2001). This policy takes into account cumulative impacts from all projects on BLM land in the region.

5.5.1.3. Habitat degradation mitigation

Measures commonly are implemented to reduce habitat degradation. These measures are different from disturbance minimization measures in that they are intended to avoid or mitigate transient or accidental impacts that can degrade habitat quality.

- Prohibit off-road travel. Vehicles are restricted to use of existing roads.

- Contain and remediate fluid spills. Various types of fluids are used or produced in oil fields. Many of these fluids are highly toxic, but even clean water in inappropriate situations can cause flooding of burrows, drowning of individuals, and soil erosion. Control strategies can include building berms around facilities that hold fluids. If spills do occur in habitat, then clean up, removal of contaminated soils, and restoration may be required.

- Prevent and suppress fires. Fires can significantly degrade habitat quality, particularly in regions like the San Joaquin Valley where vegetation communities are not fire-adapted. Thus, oil field operators may implement a variety of measures to prevent fires, including use of spark arrestors on equipment, prohibiting open flames, restricting smoking at field sites, equipping all vehicles with fire extinguishers, and staging fire suppression equipment at field work sites.

- Prohibiting or restricting public access. Access to oil fields by the general public may be prohibited or at least limited. Access by the public can potentially result in environmental impacts, such as off-road vehicle use, shooting of animals, trampling of sensitive plant populations, wild fires, and trash dumping.

5.5.2. Avoidance of Direct Take

Measures commonly are implemented in oil fields to avoid the “taking” of listed species. According to the ESA and CESA, “taking” can include direct mortality, injury, harassment, or other actions that may adversely affect individuals of a listed species. This list was compiled from documents that addressed oil and gas production in large oil fields or over large regions. The primary documents were U.S. BLM, 2010; U.S. DOE, 1991; 2001; US DOI, 2012; USFWS, 2001.

- Conduct surveys to determine whether listed or sensitive species are present on or near sites where habitat will be impacted or where activities potentially put individuals at risk.

- Avoid to the extent practicable any sensitive habitat areas or biological features important to listed or sensitive species. Sensitive habitat areas can include vernal pools, riparian areas, wetlands, and rare plant locations. Important biological features can include dens, burrows, and roosting sites. Avoidance commonly is achieved through the establishment of exclusion zones that are closed to entry by humans and vehicles.

- Exclude, remove, or relocate individuals that cannot be avoided. If individuals or features cannot be avoided, then measures are usually required to remove them to avoid injury or death of individuals.

- Use signage to protect sensitive areas. Permanent signage sometimes is used to indicate sensitive habitat areas or important biological features and exclude entry by humans.

- Use fencing to exclude animal entry into dangerous areas. Fencing is sometimes used around project sites to exclude entry by rare animals. Typically, this strategy is applied to relatively small sites (e.g., well pads) that can be effectively fenced and that are not so extensive (e.g., long, linear projects) that the fencing would severely inhibit animal movements through the area. Occasionally, more extensive (e.g., long, linear projects) are fenced in segments so as to permit animal movements through an area. Examples of species commonly excluded with fencing include blunt-nosed leopard lizards (*Gambelia sila*), kangaroo rats (*Dipodomys spp.*), and California tiger salamanders (*Ambystoma californiense*).

- Install fencing and netting around and over sumps to exclude entry by animals. Sumps are commonly constructed to contain fluids produced in oil fields, in particular produced water that is pumped from wells along with oil and gas. Such water can include a variety of chemicals potentially harmful to animals. Animals can be attracted to sumps filled with produced water mistaking them for a source of drinking water or wetland habitat. Fencing and netting is placed around and over these sumps to prevent animals from accessing the water in which they could drown, or if ingested or absorbed, could cause injury or death.

- Cap all pipes to prevent entry by animals. Pipes are used in abundance in oil fields for drilling wells, constructing pipelines, and other purposes. Animals occasionally seek shelter in pipes, and then can be harmed or killed if they become entrapped in the pipe or the pipe is moved. Capping the ends of pipes prevents use by animals.

- Prevent animal entrapment in open trenches and pits. Trenches and pits are commonly dug in oil fields for a variety of purposes. Strategies to prevent animal entrapment include (1) covering them when work is not being performed, (2) monitoring, usually at the beginning and end of the work day, and removal of any animals, (3) reducing side slopes to 45 degrees or less, and (4) building ramps to allow any trapped animals to escape.

- Limit vehicle speeds. To reduce the potential for animals to be struck by vehicles, speed limits are commonly imposed in oil fields. In areas with listed or sensitive species, limits are typically no more than 25 mph and sometimes as low as 5 mph. Lower speed limits may be required at night when animals are active.

- Remove all trash and food that might attract animals to work sites. Typically at the end of the work day, all trash and food is removed from the site so as not to attract animals.

- Prohibit dogs or other pets. Domestic animals, particularly dogs, potentially could pursue, capture, and kill wildlife species. Even just the presence of dogs potentially could alter wildlife behavior in a detrimental manner. Domestic animals also could carry and introduce diseases into local wildlife populations.

- Prohibit firearms. This restriction is imposed to prevent the shooting of wildlife.

- Restrict pesticide use. Use of pesticides (e.g., rodenticides, insecticides, herbicides, etc.) may be prohibited or strictly regulated to avoid poisoning of wildlife and plants.

- Mitigation measures for rare plants. In areas where rare plant populations are known to occur, mitigation measures specifically for plants may be required. These measures include (1) complete avoidance of oil field activities, where possible, (2) limiting activities in plant populations to the period between seed set and germination, (3) collecting seeds and redistributing them in nearby undisturbed areas, (4) collecting and storing top soil, and then redistributing it in disturbed areas or back on the original site if the disturbance is temporary, and (5) prohibiting the use of herbicides in or near plant populations.

- Use of biological monitors. Biological monitors may be required to be present when work is being conducted. This is a common requirement in areas where listed species are known to be present. Biological monitors must be qualified biologists (i.e., trained to recognize species of interest and knowledgeable of applicable laws and regulations as well as appropriate responses to the appearance of species on work sites or non-compliance by workers). Monitors ensure that exclusion zones are avoided by workers, monitor activity by sensitive animals, monitor worker compliance, participate in worker education and awareness programs, and prepare compliance reports. Monitors commonly have the authority to halt work in situations such as (1) the appearance of a listed species on site, (2) death or injury of a listed species, or (3) non-compliance by workers.

5.5.3. Environmental Restoration

Restoration involves environmental remediation and recovery of ecological functions on sites where habitat has been disturbed. DOGGR provides some guidance and requirements (California Code of Regulations Title 14 § 1776 on Well Site and Lease Restoration). In essence, upon abandonment, wells must be plugged and all structures and materials on the surface must be removed. Any toxic or hazardous materials must be cleaned up. Any excavations must be filled and compacted, and any unstable slopes must be mitigated. Finally, the site should be "returned to as near a natural state as practicable."

Otherwise, requirements for restoration are inconsistent and range widely from none to extensive. On U.S. BLM lands in the southern San Joaquin Valley, intensive restoration is required and detailed protocols and procedures are provided to project proponents (USFWS, 2001). In other instances, project proponents are asked to prepare a restoration plan and submit it for agency approval (Padre Associates, 2014). The purpose of restoration efforts is to try to reestablish sufficient ecological function on previously disturbed lands such that they can again be used by local native species. Restoration usually is conducted whenever a disturbed area (e.g., road, well pad, facility site, pipeline) is no longer needed for oil and gas production activities.

Elements of restoration could include the following:

- Removal of all anthropogenic materials.
- Removal of any contaminated soil.
- Ripping/disking the site to reduce soil compaction.
- Earthwork to restore natural contours of a site.
- Seeding with native plants (seed mixes vary immensely but usually include one or more shrub species).
- Application of sterile straw or other cover material to inhibit erosion.
- Monitoring restoration success. A typical performance measure is to restore vegetative cover on a disturbed site such that it is equal to at least 70% of the cover on nearby undisturbed sites.

5.5.4. Employee Training

A common requirement for oil and gas production operations is to provide environmental training for employees. Such training generally is required of any individual that works on a given project, even if employee responsibilities do not include field work. Employee education and awareness programs commonly include information on:

- How to recognize listed and sensitive species.
- How to recognize sensitive habitats.
- Mandatory mitigation measures and their implementation.
- Applicable laws and regulations, and consequences that could result from non-compliance.

5.5.5. Regional Species-Specific Measures

Most of the measures described above are relatively general and therefore widely applied. In addition to these general measures, there may be measures required that are specific to local listed or sensitive species. Appendix 5.A gives specific measures that have been required in oil fields occurring within the range of California condors (*Gymnogyps californianus*), Arroyo toads (*Bufo californicus*), red-legged frogs (*Rana aurora draytonii*), and fairy shrimp (Castle Peak Resources, 2011; USFWS, 2009; 2005).

5.5.6. Efficacy of Mitigation Measures

As detailed above, numerous measures have been implemented in oil fields to mitigate impacts to terrestrial species and their habitats from oil and gas production activities. However, rarely has the efficacy of any of the measures been assessed. In general, most of the measures have not been subject to systematic studies quantifying the contribution of the measures to the conservation of biological resources. However, a small number of assessments have been conducted, and these are summarized below.

5.5.6.1. Use of Barriers to Exclude Blunt-Nosed Leopard Lizards

Germano et al. (1993) evaluated the use of barriers to exclude endangered blunt-nosed leopard lizards from a 2-km pipeline trench and associated right-of-way. Prior to erecting barriers, lizards were getting trapped in the trench and were observed along the right-of-way used by construction vehicles. They used strips of aluminum flashing and plastic erosion cloth, and both materials effectively excluded lizards from the construction area, although the flashing was cheaper and less likely to collapse.

5.5.6.2. Use of Topsoil Salvage to Conserve Hoover's Wooly-Star

Hinshaw et al. (1998) investigated the salvage of topsoil to establish threatened Hoover's wooly-star (*Eriastrum hooveri*) on disturbed sites. Topsoil laden with Hoover's wooly-star seeds was collected from within population areas and redistributed on disturbed sites in areas with and without the species. Within populations, reestablishment rates were similar between plot that received topsoil and control plots. In areas where the species was not present, Hoover's wooly-star was successfully established in low densities.

5.5.6.3. Habitat Restoration for San Joaquin Valley Listed Species

Hinshaw et al. (2000) assessed sites on Naval Petroleum Reserve No. 1 (Elk Hills Oil Field) on which habitat reclamation had been conducted. Reclamation methods had included site preparation and seeding with annual plants and shrubs. They examined 996 sites five years and 10 years post-reclamation. After five years, 47.2% of the sites met the success criterion of vegetative cover equal to or exceeding 70% of the cover on reference or adjacent undisturbed sites. After 10 years, 77.4% of the sites met the criterion. However, they cited unpublished data from a study in which a subset of the sites had been compared to sites on which no reclamation was conducted but instead were allowed to revegetate naturally. Revegetation occurred at least as rapidly on non-reclaimed sites as on reclaimed sites. Furthermore, reclaimed sites commonly had shrub densities exceeding those on reference sites, and these dense shrubs provided optimal cover for predators of endangered San Joaquin kit foxes, possibly to the detriment of the kit fox. Reclamation costs averaged $11,827 per successfully revegetated hectare. The authors concluded that at least in the southern San Joaquin Valley, habitat restoration could be achieved by simply preventing additional disturbance of sites and allowing them to revegetate naturally, and any conservation funding might be better spent on acquiring additional undisturbed habitat versus reclaiming disturbed habitat.

5.6. Assessment of Data Quality and Data Gaps

- For all the potential impacts of well stimulation to wildlife and vegetation identified in, there are major data gaps in understanding the actual extent of the impacts. Of all the impacts, the most data were available to quantify habitat loss caused by hydraulic-fracturing-enabled-production; even here we were hampered by the lack of comprehensive historical data on the frequency and location of hydraulic fracturing. For all other impacts the data gaps were even larger. For introduction of invasive species, releases of harmful fluids to the environment, water use, litter, noise, light and traffic, there are insufficient data on how well stimulation alters the environment and if and how wildlife and vegetation in California are actually affected.

- While we have data that allows us to make a reasonable estimate of habitat loss caused by hydraulic fracturing enabled production, we have very little information on other important pathways of impacts of well stimulation to wildlife and vegetation such as the kinds and quantities of hydraulic fracturing chemicals that enter the environment; the degree to which local streams could be impacted by water withdrawals for stimulation; the noise caused by well stimulation; litter, traffic, noise and light generated at well stimulation sites.

- While we know that an increasing density of wells causes loss and fragmentation of habitat, we have a very limited understanding of how this in turn affects the local organisms that inhabit the area. How does the increasing density of oil wells affect local population sizes, behavior, habitat selection, and migratory patterns of organisms? What are the mechanisms of any impacts to wildlife and vegetation – loss of habitat, water use, water contamination, noise, light, traffic, litter, or other causes?

- Most of the literature on ecological impacts of oil and gas production in California was conducted in order to comply with regulatory requirements and thus tends to focus on threatened and endangered species protected under the United States and California Endangered Species Acts. There has been relatively little work on species that are not listed as endangered or threatened, or on more general ecosystem properties such as biodiversity.

- To date, there has been little evaluation of the effectiveness of mitigation measures. Rigorous evaluation of the various, commonly prescribed mitigation measures would allow regulators to identify and require only those methods with proven value. The contribution of mitigation measures to overall conservation efforts is unknown. Even assuming that all mitigation measures are effective in achieving their intended purpose (e.g., avoiding take, preventing additional habitat disturbance, restoring habitat), there has been no assessment of whether such measures contribute significantly to the conservation of species.

- Habitat restoration of abandoned oil and gas well sites can be an important tool for conservation, but the very limited studies available in the San Joaquin Valley found that neither passive revegetation nor active restoration efforts restored sites to their pre-disturbance value for native species. More experimentation in this arena would tell us if restoration is possible, and if so, what approaches are effective.

- Cumulative effects analyses, which look at the additive impacts of multiple projects over regional scales and time scales of years or longer, are inadequate. Environmental impact reviews are conducted for most oil and gas production activities and these reviews typically include a cumulative effects analysis, but most are conducted on a project-specific or site-specific basis with little consideration of the larger regional landscape. No comprehensive analysis has been conducted on cumulative environmental effects. Such analyses are critical, particularly in regions like the San Joaquin Valley where profound habitat loss from a variety of sources including oil and gas production may have already precluded the recovery of some listed species.

5.7. Findings

- While some portions of oil and gas fields are dedicated nearly exclusively to hydrocarbon production, in other areas oil and gas production is interspersed with human development, agriculture, and natural habitat.

- There are a number of places in the state where valuable natural habitat is interspersed or adjacent to well-stimulation-enabled production. In those areas where hydraulic fracturing-enabled production occurs in a landscape of natural habitat, the additional production causes habitat loss and fragmentation. The counties with the greatest amount of habitat loss and fragmentation attributable to well-stimulation enabled production were (with hectares of altered habitat in parenthesis): Kern (13,400), and Ventura (5,000).

- Compared to the total area of natural habitat in the state, the amount altered by hydraulic-fracturing-enabled-production is modest, less than one-tenth of a percent of the total area of natural habitat. However, the effects are highly localized and have disproportionate effects in a few areas and for a few habitat types. For valley saltbush scrub, 6% of its statewide extent was impacted by hydraulic-fracturing-enabled-production, and 2% for Venturan coastal sage scrub.

- The natural communities most disturbed by well-stimulation-enabled production were valley saltbush scrub and non-native grassland (mainly in Kern County), and Venturan coastal sage scrub and buck brush chaparral (largely in Ventura County).

- We found recorded instances of 24 listed species on or within 2 km of oil fields with at least 200 hectares altered by hydraulic-fracturing enabled production. Threatened and endangered species occurring in the vicinity of areas highly altered by hydraulic-fracturing-enabled-production are the San Joaquin Valley upland species such as the San Joaquin kit fox, Nelson's antelope squirrel, blunt-nosed leopard lizard, and the giant kangaroo rat, and the California Condor in the Ventura Basin.

- Little data are available to assess the potential impacts of well stimulation on wildlife and vegetation by pathways other than habitat conversion. Factors such as introduction of invasive species, pollution from fluid discharges, water use, noise and light pollution, and vehicle traffic are known to affect wildlife and vegetation, but the extent to which well stimulation affects wildlife and vegetation by those pathways is unknown.

5.8. Conclusions

- With respect to habitat loss and fragmentation, the impact of stimulated wells is not inherently different from that of unstimulated wells. The construction of wells and their support infrastructure disturbs habitat regardless of whether a well is stimulated. Other potential impacts to wildlife and vegetation, such as pollution, could differ between stimulated and unstimulated wells, but we have insufficient data to quantify the effects.

- During the period of 1977 – September 2014, hydraulic fracturing enabled a modest proportion (about 3.5%) of the production that impacts natural habitat in California because most of it occurred in areas that are already highly altered by human activities such as other forms of oil and gas production, agriculture, or urbanization. In turn, oil and gas production as a whole has a much smaller footprint in the state than cities and cultivated land.

- Hydraulic fracturing is becoming an increasingly important driver for enabling oil and gas production in the state. During the period of October 2012 – September 2014, 20% of the land area that was newly developed for oil and gas production could be attributed to hydraulic fracturing.

- Hydraulic-fracturing-enabled activity can be locally important in certain regions, chiefly the southwestern San Joaquin Valley, where frequently stimulated fields overlap with high-quality habitat for rare species, and in Ventura County, where regularly stimulated fields overlap with critical habitat for the California condor and steelhead salmon.

5.9. References

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Chapter Six

Potential Impacts of Well Stimulation on Human Health in California

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6.1. Abstract

This chapter addresses environmental public health and occupational health hazards that are directly attributable to well stimulation or indirectly associated with oil and gas development facilitated by well stimulation in California. Hazards that are directly attributable to well stimulation primarily consist of human exposures to well stimulation chemicals through inadvertent or intentional release to water, air, or soil followed by environmental fate and transport processes. Hazards that are indirectly associated with well-stimulation-enabled oil and gas development also include chemicals and environmental releases. Such hazards may not be directly related to well stimulation, but rather could result from expanded development that is enabled by well stimulation.

The risk factors directly attributable to well stimulation stem largely from the use of a very large number and quantity of stimulation chemicals. The number and toxicity of chemicals used in well stimulation fluids make it impossible to quantify risk to the environment and to human health. To gain insight on the potential of chemicals used in stimulation to harm human health, we used a ranking scheme that is based on toxic hazards of chemicals and reported quantities used in well stimulation operations. The ranking includes both acute and chronic toxicity. (Note that these same chemicals were ranked for aquatic toxicity in Volume II Chapter 2.)

Important pathways for human exposure to well stimulation chemicals and emissions include both water and air pathways. For water, possible pathways leading to exposure in California were identified in Volume II Chapter 2. These pathways include (1) the possibility of shallow hydraulic fractures intersecting protected groundwater, (2) the possibility of hydraulic fracturing intersecting other wells that could provide leakage paths, (3) the potential for spills and leaks of stimulation fluids, (4) injection of produced water, which could contain stimulation chemicals, into protected aquifers, (5) use of produced water that may contain stimulation chemicals in agriculture, (6) disposal of produced water that may contain stimulation chemicals in unlined sumps, and (7) the impact of strong acid use in recovered fluids and produced water. Wastewater generated from stimulated wells in California includes "recovered fluids" (flowback fluids collected into tanks following stimulation, but before the start of production) and "produced water" (water extracted with oil and gas during production). Air pathways that could result in human exposure to chemicals used in well stimulation include atmospheric dispersion of air pollutant emissions to communities near production sites. Studies have found human health risks attributable to emissions of petroleum-related compounds associated with oil and gas development in general. However, public health impacts associated with proximity to oil and gas production have not been measured in California. As such, detailed studies of the relationship between health risks and distance from oil and gas development sites are warranted. In the interim, increased application and enforcement of emission control technologies to limit air pollutant emissions and science-based minimum surface setbacks between oil and gas development and human populations could help to reduce these risks.

Our assessment of the scientific literature for community and occupational exposures and health outcomes indicates that there are a number of potential human health hazards associated with well-stimulation-enabled oil and gas development, but that California-specific peer-reviewed studies are critically scarce, and that air, water, and human health monitoring data have not been adequately collected, analyzed, verified, or reported.

6.2. Introduction

This chapter addresses environmental public health and occupational health hazards that are directly attributable to well stimulation or indirectly associated with oil and gas development facilitated by well stimulation in California.

Hazards that are directly attributable to well stimulation primarily consist of human exposures to well stimulation chemicals through inadvertent or intentional release to water, air, or soil followed by environmental fate and transport processes. Hazards that are indirectly associated with well-stimulation-enabled oil and gas development also include chemicals and environmental releases. Such hazards may not be directly related to well stimulation, but rather result from expanded development that is enabled by well stimulation. A number of potential contaminant release mechanisms and transport pathways have been described in Volume II, Chapters 2 and 3. In this chapter, we extend the previous discussion of environmental release and environmental transport mechanisms to include potential human exposure pathways, and summarize the hazards in the context of community and occupational health.

Hydraulic fracturing enables some oil and gas development that would not occur without this technology, but any oil and gas development presents hazards to human health through exposure to chemicals. Thus, to the extent that stimulation increases oil and gas development, hazards associated with development will also be increased. For example, additional emissions of toxic air contaminants (TACs) that are directly or indirectly attributable to well stimulation might be small relative to other regional sources (see Volume II, Chapter 3), but might have a higher local health impact near to the point of release. In addition, air pollution associated with the entire operation of oil and gas production can create significant human exposures. Therefore, we extend the discussion of indirect air pollution and emissions from Chapter 3 to consider potential human exposure pathways, and summarize the indirect hazards in the context of community and occupation health.

California-specific data on the impacts of well-stimulation-enabled oil and gas development is insufficient to provide a conclusive understanding of potential hazards and risks associated with well stimulation. Studies conducted outside of California consider health impacts near oil and gas development that are enabled by hydraulic fracturing, but do not differentiate the association of observed health risks between hydraulic fracturing stimulation and oil and gas development in general. Thus, the same health impacts that have been found near oil development enabled by hydraulic fracturing may exist in any oil and gas development.

The approach we take to assess human health hazards follows the general recommendations of the National Research Council (NRC, 1983; 1994; 1996; 2009) to compile, analyze, and communicate the state of the science on the human health hazards associated with well stimulation.

We begin with a summary of all hazards that have been described in earlier chapters of this volume, with an emphasis on human health aspects and risk factors. This provides a single comprehensive list of human health risk factors and hazards for well stimulation activities in California, with reference to the specific locations in the report where each hazard is discussed. We then carry out a detailed assessment of human-health-relevant hazards from chemicals, and from water and air pollution.

Because it is extremely difficult to identify specific causal relationships for a given hazard and health outcome, we employ two alternative approaches to explore hazards associated with a given activity, a bottom-up and top-down approach. The bottom-up approach follows the standard risk assessment framework. In this approach, we characterize the composition of well stimulation fluids and toxic air contaminants associated with well stimulation activities, and then identify chemical-specific human-health-relevant toxicity data, where available, and rank the chemical hazards based on a combined hazard metric that includes frequency of use, mass used, and toxicity. Our second approach, the top-down assessment, evaluates chemical and physical hazards associated with well stimulation activity by starting with population health outcomes and working backwards to evaluate potential associations between health outcomes and well stimulation activity (or oil and gas development activity, more broadly). To apply the top down approach, we draw from the peer-reviewed literature, where individual outcomes and potential hazards are studied, and findings provide evidence of possible associations between public health hazards and risks. We conclude with a review of occupational-health-relevant regulations and studies and a discussion of noise- and light-pollution health hazards. We identify potential mitigation strategies that, if properly deployed and enforced, may reduce occupational and community health impacts. Finally, we discuss well-stimulation information gaps related to environment protection in California.

As explained in Volume II, Chapter 1, there are both direct and indirect impacts of well-stimulation-enabled oil and gas development that influence public health risks. Based on available evidence, public health risks associated with direct impacts (which are the incremental impacts of oil and gas development attributable to the stimulation process itself and activities directly supporting the stimulation) appear to be small relative to the indirect impacts. To say it another way, the majority of public health risks associated with well stimulation are likely to be indirect, in that they arise from the additional oil and gas development that is enabled by well stimulation. All forms of oil and gas development, not just that enabled by well stimulation, may cause similar public health risks.

As an example, Volume II, Chapter 3 (air) found that benzene and formaldehyde emissions from oil and gas development is a significant fraction of stationary source emissions and may result in elevated atmospheric concentrations in places where people live, work, play, and learn. The current scientific literature has established that benzene is emitted from nearly all oil and gas development (Pétron et al., 2012; Pétron et al., 2014; Helmig et al., 2014). Studies show elevated health risks near hydraulic-fracturing-enabled oil and gas development attributable to benzene (McKenzie et al., 2012). Benzene and formaldehyde are not intentionally added to hydraulic fracturing or other well stimulation fluids, but may be a component of some of the petroleum-based mixtures used in hydraulic fracturing fluids. Overall, the health risks associated with benzene and formaldehyde occur because oil and gas is co-produced—and co-emitted—with these compounds. If public health investigations of benzene exposure were to be conducted only for those exposures near *stimulated* wells, then such investigations would result in a very poor understanding of both the extent of these risks and potentially effective mitigation measures that could protect public health. Concern about the health effects from benzene, formaldehyde, and many other health risks associated with oil and gas development should be approached through studies of oil and gas development from all types of reservoirs, not just those that are stimulated.

6.2.1. Framing the Hazard and Risk Assessment Process

The terms *hazard*, *risk*, and *impact* are often used interchangeably in everyday conversation, whereas in a regulatory context they represent distinctly different concepts with regard to the formal practice of risk assessment. A hazard is defined as any biological, chemical, mechanical, environmental, or physical stressor that is reasonably likely to cause harm or damage to humans, other organisms, or the environment in the absence of its control (Sperber, 2001). Risk is the *probability* that a given hazard will cause a particular harm, loss, or damage as a result of exposure (NRC, 2009). Impact is the particular harm, loss, or damage that is experienced if the risk occurs. Hazard can be considered an intrinsic property of a stressor that can be assessed through some biological or chemical assay. For example, a pH meter can measure acidity, disintegration counters can detect ionizing radiation, cell or whole animal assays, etc. can detect biological disease potency. These types of tests allow us to declare that a substance is acidic, radioactive, a mutagen, a carcinogen, or other hazard. However, defining the probability of harm requires a receptor (e.g., human population) to be exposed to the hazard, and often depends on the vulnerability of the population based on age, gender, and other factors. As a result, risk is extrinsic and requires detailed knowledge about how a stressor agent (hazard) is handled, released, and transported to the receptor populations.

In its widely cited 1983 report, the National Research Council (NRC) first laid out the now-standard risk framework consisting of research, risk assessment, and risk management as illustrated in Figure 6.2-1 (NRC, 1983). The NRC proposed this framework to organize and evaluate existing scientific information for the purpose of decision making. In 2009, the NRC issued an updated version its risk assessment guidance titled "Science and Decisions: Advancing Risk Assessment" (NRC, 2009). This report reiterated the value of the framework illustrated in Figure 6.2-1, but expanded it to include a solutions-based format that integrates planning and decision making with the risk characterization process. The NRC risk framework illustrates the parallel activities that take place during risk assessment and the reliance of all activities on existing research. These activities combine through the risk characterization process to support risk management.

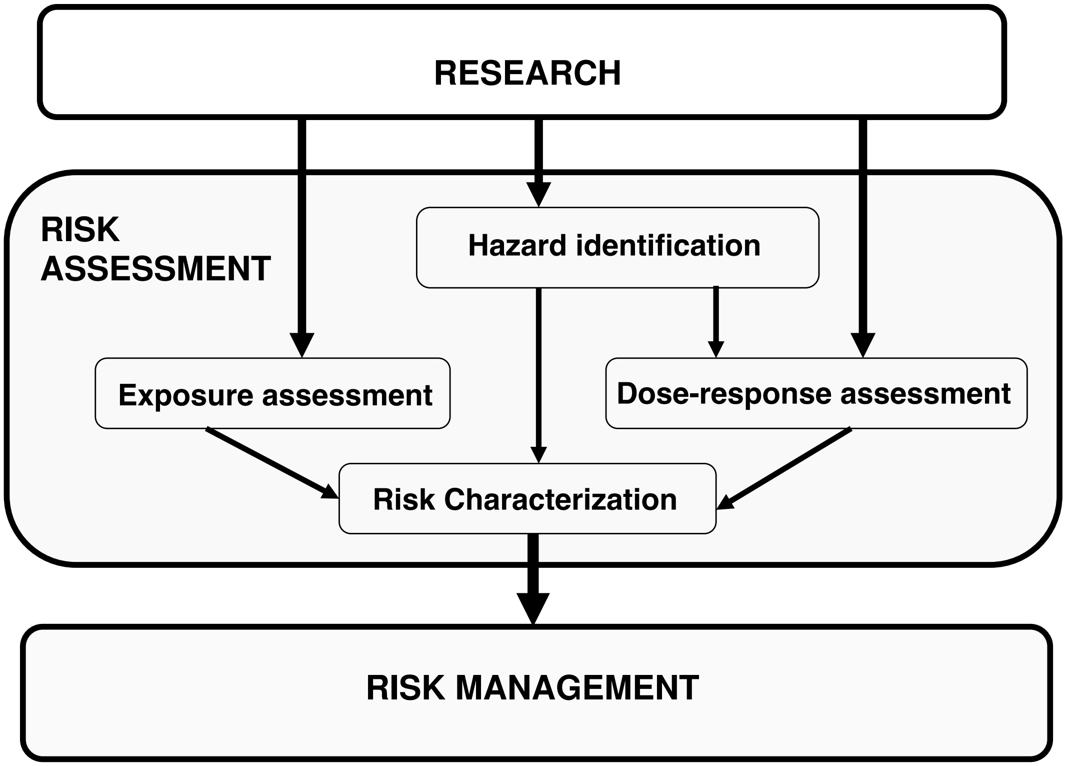

Figure 6.2-1. The NRC (1983) Risk Analysis Framework.

In using the framework in Figure 6.2-1, the first task in the risk analysis process is to identify any feature, event, or process associated with an activity that could cause harm. These are called "hazards." Any given hazard may or may not be a problem. It depends on the answers for two additional questions. First, is the hazardous condition likely to result in a population being exposed to the hazard? Second, what will be the impact if the hazardous exposure does occur (dose-response)? If we know the magnitude of a specific hazard exposure and the relationship between the magnitude of exposure and response or harm, then we can estimate the risk associated with that hazard. In cases where the hazardous condition is unlikely or where, even if it did occur, the harm is insignificant, then the risk is low. Risk is only high when the hazardous condition is both likely to occur and would cause significant harm if it did occur. Of course, there are many combinations of likelihood and harm possible.

Formal risk analysis presents difficulties, because we often lack:

- Data on all the possible hazards;
- Information on the likelihood and magnitude of exposure; and
- Data to support an understanding the relationship between exposure (dose) and harm (response).

If a hazard has not been identified, then it is difficult to develop steps to mitigate potential harm. In this case, a useful approach is to avoid the problem where possible, for example by choosing chemicals that are better understood, less toxic, or more controllable rather than choosing ones for which there is little toxicity information or poor understanding of the relationship between the hazard and risk to the environment and/or to public health. These options for both known and unknown hazards are discussed further in the mitigation section of this chapter as well as in Volume II, Chapter 2, Section 2.4 and in the Summary Report Conclusions.

Although one can attempt to identify *all* hazards associated with well-stimulation-enabled oil and gas development in California, it is important to note that this does not mean that all hazards that are identified present risks. A formal risk assessment is required to estimate risk associated with any given hazard. Although operators can make use of chemicals identified "acceptable" by programs such as the U.S. Environmental Protection Agency (U.S. EPA) Design for Environment Program or the North Sea Gold Ban list, uncertainties about exposure and impact can remain. A formal risk assessment is a significant undertaking that is beyond what was possible in this report. Among the goals of this chapter are to identify community and occupational hazards and highlight those where additional study may be warranted in the context of developing and implementing policies for well stimulation operations.

6.2.2. Scope of Community and Occupational Health Assessment

We consider and include both intentional and unintentional releases of chemical hazards to surface water, groundwater, and air as a direct and indirect result of well stimulation activities. These activities include the transport of equipment and materials to and from the well pad; mixing, handling, and injection of chemicals; and management of recovered fluids/produced water, drill cuttings, and other waste products (NRC, 2014; Shonkoff et al., 2014). In addition, we consider chemical hazards that are produced and/or released during support activities for well stimulation and from stimulated wells, such as: reaction products and mobilized chemical and/or radioactive hazards from the stimulated wells; emissions from generators, compressors, and other equipment during and after stimulation activity; leakage from transfer lines and infrastructure; and accidental spills. Finally, we consider other physical hazards related to well stimulation activity, including elevated noise and light. These hazards are relevant to both community and occupational health.

We exclude hazards associated with the manufacturing of materials, supplies, or equipment that are used in well stimulation activity; hazards from transport of oil and gas to refineries; hazards related to refining; or hazards from the combustion of hydrocarbons as fuel. These hazards, though important, are far removed both temporally and geographically from activities related to the well-stimulation-enabled oil and gas development process. We also exclude economic and psychosocial hazards that may be related to oil and gas development activities and may be important considerations in specific areas, but are beyond the scope of this chapter.

We focus primarily on hazards identified in relevant California-specific datasets and/or in the peer-reviewed literature that is specific to California. We augment this information with hazards identified in peer-reviewed studies conducted outside of California. As pointed out in Volume I and in other chapters in Volume II, geologic conditions and current practice with well stimulation in California can be different from that performed in other states, so not all hazards associated with well-stimulation-enabled oil and gas development outside of California are generally applicable to the California context.

6.2.3. Overview of Approach and Chapter Organization

The objective of this chapter is to catalogue and highlight important *community and occupational health hazards* associated with well stimulation activity in California. This is in contrast to earlier chapters of this volume that focused on environmental hazards in general and specifically those with water, air, and ecological pathways. There is significant overlap among the water, air, and ecological hazards described in earlier chapters and human-health-relevant hazards discussed in this chapter. Therefore, we begin in Section 6.2.4 with a summary of all hazards that have been described in earlier chapters of this volume, with an emphasis on human health aspects and risk factors, and we merge these with hazards that are identified and described in subsequent sections of this chapter. This provides a single list of human-health-relevant risk factors and hazards for well-stimulation-enabled oil and gas development activities in California, with reference to the specific locations in the report where each hazard is discussed. We also link the identified human health hazards to the case studies in Volume III of this report, where some of these hazards are illustrated and/or assessed in specific geographic places. Following the table of human-health-relevant hazards, we provide additional details on each risk factor/hazard combination from the list as well as other hazard/risk factors that are not listed (e.g., coccidiomycosis from exposure to San Joaquin Valley dust) along with recommendations for mitigating of risk.

After reporting and reviewing all human-health-relevant hazards in Section 6.2.4, we conduct a more detailed assessment of human-health-relevant hazards. The remainder of this chapter follows the issues summarized in the table, with the human health hazards (both community and occupational) defined and grouped into the following categories (and the section in which they are discussed):

- *Well stimulation chemicals* (Section 6.3)—includes both hydraulic fracturing and acidization chemicals intentionally injected to stimulate the reservoir or to improve oil and gas production. These chemicals are known and reported by industry on a mostly voluntary basis and more recently under Senate Bill 4 (SB 4, 2014) on a compulsory basis.

- *Recovered fluids and produced water* (Section 6.4)—includes some fraction of the well stimulation chemicals but can also include mobilized chemical compounds, naturally occurring toxic materials (such as radionuclides), and degradation and synergistic by-products from well stimulation chemicals, naturally occurring chemical constituents, and hydrocarbons.

- *Air pollutant emissions associated with well stimulation-enabled oil and gas development* (Section 6.5)—includes combustion products and/or chemical emissions from pumps, generators, compressors and equipment; venting and flaring emissions; dust from well stimulation and land-clearing activities; leaks from transfer lines and/or well heads; longer-term leakage of oil and gas from stimulated wells. (This category does not include emissions from refining and use of the hydrocarbon products.)

- *Occupational Health* (Section 6.6) —includes hazards such as exposure to respirable silica, volatile organic compounds (VOCs), and acids.

- *Other* (Section 6.7)—includes physical hazards such as light and noise and heavy equipment activity, industrial accidents (e.g., loss of well control, explosions), biological hazards such as valley fever in areas where surface soil is disturbed by well stimulation activity, spills from trucks transporting chemicals that can contaminate private wells.

We use the above categories to differentiate hazards that have similar release mechanisms and time of release, such that all chemicals in a given category are likely to be released into the environment by the same mechanism or activity and in the same location. These categories enable us to group hazards identified in this report that are relevant to human and occupational health risk in the summary table below (Table 6.2-1). The specific hazards are listed in terms of the four categories above, along with California-specific factors or conditions (risk factors) that are expected to increase or decrease the human health risk associated with the hazards. All of these risk factors identified in the summary table are applicable to the San Joaquin Valley (SJV), where more than 85% of the well stimulation events in California occur. Some factors also apply to other oil and gas producing regions where well stimulation is used.

In the sections that follow the summary table, we expand on the specific human health hazard categories identified above. In general, when evaluating population-level human-health impacts, it is extremely difficult to identify specific causal relationships for a given health hazard and impact. As a result, risk assessors consider alternative approaches to assess the likelihood of harm. The first approach, sometimes referred to as "bottom-up," starts with a cause, such as chemical hazard, and attempts to track emissions and exposure pathways along with dose-response modeling to characterize population impact. This approach often must confront uncertainties identifying exposures and actual health impacts. The second approach, sometimes referred to as "top-down," starts with an impact—for example disease incidence—and attempts to track it back to some source chemical or activity. For the "top-down" approach, uncertainty arises from the lack of statistical power in making associations with low disease rates, as well as from the considerable lag times between exposure and occurrence of diseases (e.g., cancer). Because of their significant but different types of limitations, it is useful to consider both approaches. These alternate ways of exploring hazard are illustrated in Figure 6.2-2. In this chapter, we use both approaches.

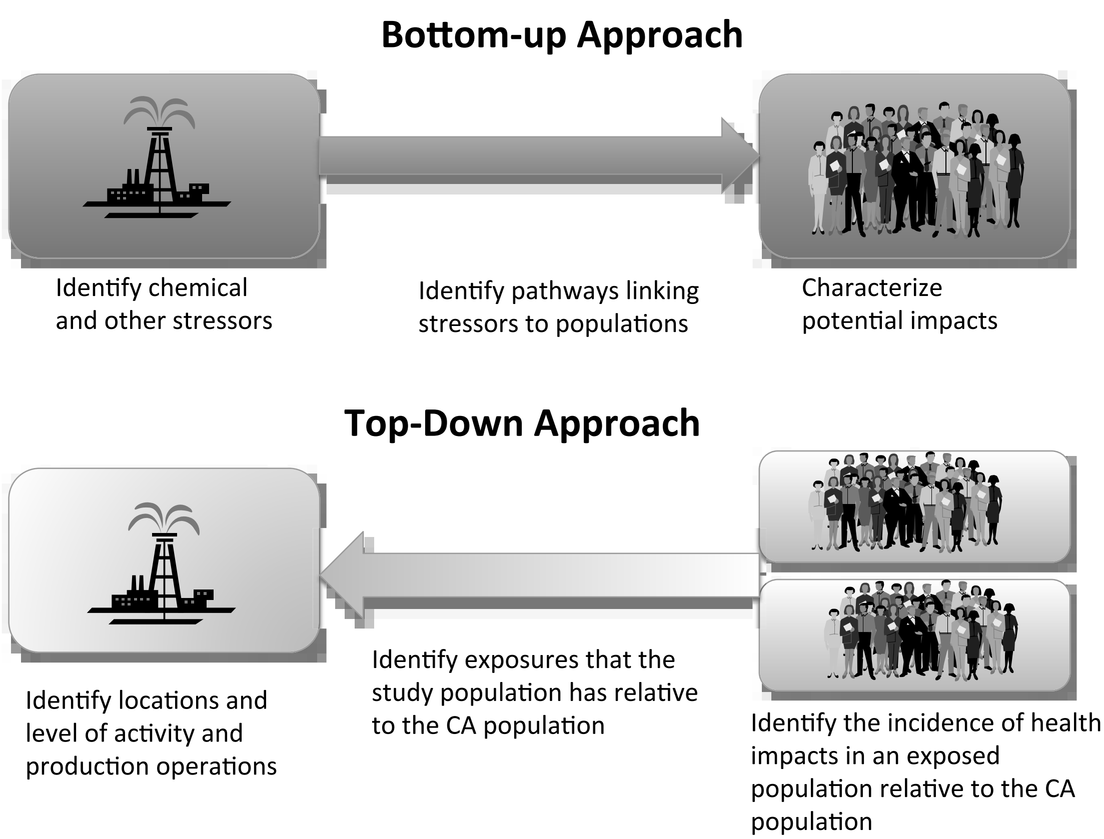

Figure 6.2-2. Illustration of two approaches used to identify human health hazards associated with an activity.

We conduct a bottom-up assessment in Section 6.3, 6.4, and 6.5 where we evaluate chemical and physical hazards associated with well stimulation chemicals and potential contamination pathways. We build on the discussions in Volume II Chapters 2 and 3 that characterize the composition of well stimulation fluids and toxic air contaminants associated with well stimulation activity. We extend this data by identifying chemical-specific human-health-relevant dose-response information where available, and rank the chemical hazards based on a combined hazard metric that includes frequency of use, mass used, and toxicity. We also discuss potential exposure factors to further extend the bottom-up assessment.

The most relevant approach for top-down hazard assessment would be to conduct a formal epidemiological study that attempts to pull out specific cause-effect relationships within a population. However, these studies require that the "effect" already be expressed (and measured) in the population, and that the effect is both unique and common enough to identify. A more general top-down approach draws from the peer-reviewed literature, where individual outcomes and potential hazards are studied, and findings provide evidence of possible associations between hazard and public health risk. We include a top-down hazard assessment in support of each section focusing primarily on California and health-outcome studies and, where studies from outside of California are relevant, we review and summarize the evidence for hazards based on experience and observations from outside California. A detailed summary compilation of the literature is provided in Appendix 6.A for public health, Appendix 6.D for occupational health and Appendix 6.F for noise.

We wrap up the chapter with a summary of critical data gaps (in addition to those identified in earlier chapters) and then with conclusions and recommendations for community and occupation health.

6.2.4. Summary of Environmental Public Health Hazards and Risk Factors

The geology and history of hydrocarbon development, along with current practices and current regulatory framework for well stimulation-enabled oil and gas development in California, give rise to the potential public health risks associated with well stimulation activities. Table 6.2-1 summarizes all human health relevant hazards identified in this chapter and in previous chapters of this volume. We also provide reference to the location in this volume where each risk factor and hazard is discussed in more detail. Although we include possible mitigation strategies in Table 6.2-1, data on the *adequacy and effectiveness* of regulations to achieve these goals is often not available, requires more study, and/or is beyond the scope of this report.

Table 6.2-1. Summary of human health hazards and risk factors in California substantiated with California-specific data.

Risk Factor	**Hazard**	**Description of the issue**	**How the risk factor is expected to influence public health risks**	**Possible mitigation**	**Volume and Section in this Report**
Number and toxicity of chemicals in well stimulation fluids	Well stimulation chemicals	Operators have few restrictions on the types of chemicals they can use for hydraulic fracturing and acid stimulation. In California, oil and gas operators have reported the use of over 300 chemical additives. About 1/3 have not been assessed for toxicity. Of the chemicals for which there is basic environmental and health information, only a few are known to be highly toxic, but many are moderately toxic. There is incomplete information on which of the chemicals used have the potential to persist or bio-accumulate in the environment and may present risks from chronic low-level exposure.	If these chemicals are not released into usable water, including agricultural water and to the atmosphere then the risk is minimal. However, if there are leakage and emission pathways then it is nearly impossible to assess the risk because of the large number of chemicals, incomplete knowledge about which chemicals are present, how long these compounds persist and what their environmental and human health impacts are. Researchers and the public need access to sufficient levels of information on all chemicals involved in well stimulation, to begin an assessment of the toxicity, environmental profiles, and human health hazards associated with hydraulic fracturing and acidizing stimulation fluids.	Invoke Green Chemistry principles to reduce risk, that is, use smaller numbers and amounts of less toxic chemicals, and avoid chemicals with unknown impacts. Mitigate exposure pathways. Limit the chemical use in hydraulic fracturing to those on an approved list that would consist only of those chemicals with known and acceptable toxicity profiles.	Vol. II Ch. 2 S. 2.4 & Summary Report S. 3.1
Disposal of water in unlined sumps	Recovered fluids & produced water	The disposal of contaminated water in unlined pits is banned in nearly all other states because such fluids can migrate out of sumps into groundwater and move along with this water to wells or surface water where contamination can be a serious problem. Nearly 60% of wastewater from stimulated wells in California was disposed in unlined sumps.	Well stimulation and naturally occurring chemical constituents can evaporate from these ponds to the atmosphere as air pollutants, leak into aquifers, or migrate through the soil which could lead to food chain exposure to biota and humans. Chemicals in recovered fluids and produced water may be toxic, persistent, or bioaccumulative.	Test and appropriately treat water going in to unlined pits, or phase out the use of unlined sumps in the SJV for wastewater disposal.	Summary Report S. 3.2 & Vol. II Ch. 2 S. 2.6.2.1 & Vol. III Ch. 5

Risk Factor	Hazard	Description of the issue	How the risk factor is expected to influence public health risks	Possible mitigation	Volume and Section in this Report
Beneficial use of produced water	Recovered fluids & produced water	California is a water-short state and California's oil reservoirs produce about 10 times more water than oil. Produced water is sometimes reused, for example to irrigate crops. If this produced water comes from stimulated wells or oil wells producing from a reservoir where stimulation was used, stimulation chemicals could be present in the produced water.	Well stimulation chemicals and their reaction products may be toxic, persistent or bioaccumulative. Current water district requirements for testing such waters before they are used for irrigation are not sufficient to guarantee that stimulation chemicals are removed, although some local treatment plants do use adequate protocols. If produced water used in irrigation water contains well stimulation and other chemicals, this would provide a possible exposure pathway for farmworker and animals and could lead to exposure through the food chain. Currently, more than 60% of the fruits and vegetables consumed domestically come from the Central Valley.	Water districts in the SJV should explicitly disallow the use for irrigation of produced water from wells that have been hydraulically fractured, or demonstrate that their monitoring and treatment methods ensure that hydraulic fracturing chemicals and other contaminants are not present in water destined for irrigation.	Vol. II Ch. 2 S. 2.6.2.3 & Summary Report S. 3.2
Shallow hydraulic fracturing	Well stimulation chemicals	The majority of hydraulic fracturing in California is conducted from shallow vertical wells. These operations present a larger probability of fractures intersecting near-surface groundwater compared to high volume fracturing from deep long-reach horizontal wells commonly used elsewhere.	The groundwater in the vicinity of some shallow fracturing is protected. Contamination of usable groundwater presents environmental public health risks. Groundwater monitoring requirements are likely insufficient to determine whether water has been contaminated by well stimulation-enabled oil and gas development or not. The groundwater in the vicinity of much of the shallow hydraulic fracturing operations in California has high salinity and has no beneficial uses that might constitute environmental exposure pathways to humans.	The focus of regulations should be on preventing contamination of aquifers, not just monitoring for it. Operators should be required to demonstrate that stimulations could not intersect usable groundwater to receive a permit. A higher level of scrutiny should be applied to shallow stimulations. Groundwater monitoring plans should be adapted as part of the corrective action, to improve the monitoring system and specifically look for contamination in close proximity to possible fracture extensions into groundwater.	Vol. I Ch. 3 S. 3.2.3.3 & Vol. II Ch. 2 S. 2.6.2.5

Risk Factor	Hazard	Description of the issue	How the risk factor is expected to influence public health risks	Possible mitigation	Volume and Section in this Report
Hydraulic fracturing in reservoirs with long history of oil and gas production	Well stimulation chemicals Recovered fluids & produced water Air and other pollutant emissions associated with well-stimulation-enabled oil development Other	Many of the issues faced by other states arise because hydraulic fracturing has opened up oil and gas development in regions that previously had little or no experience with production. When the US Energy Information Agency issued a report indicating that a large amount of such development was also possible in California from the Monterey Formation (subsequently revised dramatically downward), many were concerned about the development of oil and gas in new geographies. This assessment finds that the most likely future use of hydraulic fracturing is in and around the reservoirs where it is currently being used.	New production in developed fields can use the existing roads, platforms and infrastructure already in place. As a result, the impacts caused by construction and traffic are much less than in new, previously undeveloped regions.	Existing infrastructure reduces the need for new pads pipelines and other stationary infrastructure. Existing infrastructure can often transport fluids to and from the pad, reducing the need for truck trips. This reduces traffic accidents and the emission of diesel particulates and other health-damaging air pollutants	Vol. II, Ch.. 3
			Old reservoirs have many existing wells. If hydraulic fractures intersect or come near these old wells, the wells could form leakage pathways for stimulation fluids. Older existing infrastructure (e.g. pipelines, storage tanks) may increase the likelihood of failures or leakage.	Locate and seal old wells in the vicinity of hydraulic fracturing if they would provide leakage paths to air and usable groundwater. Regulations should explicitly require an assessment of the integrity and leakage risk of existing wells that might be encountered by a hydraulic fracture, and remediation of wells which create a high risk of leakage into water less than 10,000 milligrams/liter total dissolved solids.	Vol. II Ch. 2 S. 2.6.2.6 & Vol. III Ch. 5 and Ch.2, Table 2.6-1.
Injection of recovered fluids and produced water into aquifers used for drinking, agriculture, and other direct and indirect uses by humans	Well stimulation chemicals Recovered fluids & produced water	Produced water from stimulated fields has been injected into aquifers that are suitable for drinking water, irrigation, and other beneficial uses.	If water from contaminated aquifers is used, it could expose humans to unsafe concentrations of toxic compounds.	Prevent injection of well stimulation chemicals to usable groundwater in the future. In the process, of reviewing, analyzing and remediating the potential impacts of wastewater injection into protected groundwater, consider the possibility that stimulation chemicals may have been present in these wastewaters.	Vol. II Ch. 2 S. 2.6.2.2 & Vol. III Ch. 5

Risk Factor	Hazard	Description of the issue	How the risk factor is expected to influence public health risks	Possible mitigation	Volume and Section in this Report
Spills and leaks	Well stimulation chemicals Recovered fluids & produced water	Surface spills and leaks are common occurrences in the oil and gas industry and must be reported and cleaned up.	Information recorded on spills and leaks is insufficient to determine whether stimulation chemicals could be involved.	Require public reporting about whether the source of the leak could contain well stimulation chemicals.	Vol. II Ch. 2 S 2.6.2.9
Oil and gas development near human populations	Well stimulation chemicals Recovered fluids & produced water Air and other pollutant emissions associated with hydraulic fracturing-enabled development Other	California has large oil reserves located under densely populated areas primarily in the San Joaquin and Los Angeles Basins. In Los Angeles, oil and gas production developed simultaneously with the growth of the city. The Los Angeles Basin has world-class oil reservoirs, with the most concentrated oil in the world. Los Angeles is also a global megacity.	Proximity to production increases exposures to air pollutant emissions and other results of oil and gas development activities (e.g., dust, chemicals, noise, light). Households that use groundwater from private drinking water wells in close proximity to oil and gas development may be at increased risk of exposure to potential water contamination.	Identify and apply appropriate measures to limit exposure by residents and sensitive receptors such as schools, daycare facilities and elderly care facilities such as scientifically based setback requirements.	Vol. II Ch. 6S. 6.8.1 & Vol. III Ch. 4.3 & Summary Report S. 3.2
Acid use	Well stimulation chemical	Operators in California commonly use mixtures of hydrochloric acid and hydrofluoric acid with other sources of fluoride anions as the most economical reagent for cleaning out wells or enhancing geological formation permeability. Reported use of hydrofluoric acid in the SCAQMD data lists the concentration as percent mass in the ingredient as 1%-3%.	Spills and leaks of undiluted acids may present an acute toxicity and corrosivity hazard. The use of acid can also mobilize naturally occurring heavy metals and other compounds that are known to be health hazards and these compounds could therefore be present in recovered fluids and produced water which humans could be exposed to if treatment and disposal is not sufficiently undertaken.	Evaluate the chemistry of recovered fluids and produced water for wells that have used acids and the potential consequences for the environment. Require reporting of significant chemical use for oil and gas development based on these results.	Vol. II Ch. 2 Ss. 2.4.3.2, 2.6.2.9 & Summary Report S. 3.2

6.3. Public Health Hazards of Unrestricted Well Stimulation Chemical Use

Previous chapters have considered environmental and ecological hazards. In this section, we examine the potential impact of well stimulation chemicals on human health, based on reported use information (frequency and quantity) and on published toxicity information.

The majority of important potential direct impacts of well stimulation result from the use of chemicals. Operators have few restrictions on the types of chemicals they use for hydraulic fracturing and acid treatments. In California, oil and gas operators have reported, on voluntary and mandated bases, the use of over 300 chemical additives (see Volume II, Chapter 2 for detailed description of chemicals). Although SB 4 (2014) now mandates reporting of chemical use by operators, the data are not subject to independent verification, and chemicals can be reported as "trade secrets," meaning they need not be fully identified. The many chemicals used in well stimulation makes it very difficult to judge the public health risks posed by releases of stimulation fluids.

In addition to the sheer number of known and unknown (trade-secret) chemical additives used, we often lack information on potential release mechanisms and important physical and chemical properties needed to characterize environmental fate and exposure pathways, and toxicological characteristics (acute and chronic) needed to fully understand chemical hazards.

The most common toxicity information about chemicals is from standardized mammalian acute toxicity tests that measure the short-term (minutes to hours) exposure concentration or one-time dose of a chemical required to induce a well-defined response (death, narcosis, paralysis, respiratory failure, etc.) of a test animal, most commonly rats and mice. Such tests are used to assess toxicity of inhalation, ingestion, and/or uptake through the skin. Acute toxicity tests measure extreme outcomes, but the tests are useful for ranking chemicals against each other and identifying chemicals that are clearly dangerous if taken into the body.

More useful but less commonly available tests for health impacts are chronic toxicity tests. These are long-term studies (often lifetime or multi-generation studies) with small mammals to observe any increases in chronic disease incidence—including tumors and cancer, reproductive/developmental changes, neurological damage, respiratory damage, life shortening. Animal-based chronic toxicity results are used for assessing the hazards and risks to communities and workers from long-term (up to lifetime) exposures to relatively low concentrations or doses of chemicals. In addition to toxicity tests with animals, some chemicals have occupational or community epidemiological studies that provide useful information on chronic toxicity. Because these studies are the result of accidents or from improperly regulated chemicals or air contaminants, there are limited numbers of chemicals that have human-based chronic health data. Approximately two-thirds of the reported chemicals used in well stimulation have publically available results from acute mammalian toxicity tests (excluding material safety data sheets (MSDS) data), and only about one-fifth of the reported chemicals have associated chronic toxicity information.

Of the chemicals for which there is basic environmental and health information, only a few are known to be highly toxic, but many are moderately toxic. For most substances we consider, there is lack of toxicological testing for long-term chronic exposure at very low levels. There is also a lack of testing on mixtures. Some of the chemicals used may have the potential to persist or bio-accumulate in the environment and present risks from chronic low-level exposure. Because the toxicology for multiple routes of exposure—inhalation, ingestion, skin contact, etc.—is rarely reported, cumulative exposure assessment is beyond the scope of our analysis.

In this section, we develop and apply a semi-quantitative ranking system for chemical hazards associated with well stimulation activity. The ranking system is not a substitute for field observations or a full risk assessment, but provides an initial focus on which chemicals are of highest concern and which are of lower priority. Section 6.3.1 describes the approach, followed by results for hydraulic fracturing chemicals, acidization chemicals, and toxic air contaminants in Section 6.3.2, finishing with a summary of relevant literature in Section 6.3.3.

6.3.1. Approach for Human Health Hazard Ranking of Well Stimulation Chemicals

Chemical hazards include both hydraulic fracturing and acidization chemicals that are intentionally injected to stimulate the reservoir or to improve oil and gas production (see Volume I, Chapter 2 for the engineering purpose of these chemicals) and unintentional releases from spills or leaks. Chemicals are used in the drilling and well stimulation processes for a variety of purposes, including as corrosion inhibitors, biocides, surfactants, friction reducers, viscosity control, and scale inhibitors (Southwest Energy, 2012; Stringfellow et al., 2014) (Section 2.4.4.1). Hydraulic fracturing uses fluids or gels that contain organic and inorganic chemical compounds, a number of which are known to be health damaging (Aminto and Olson, 2012).

In this section, we provide a bottom-up assessment to develop hazard priorities for chemicals that are used in well stimulation. The ranking is based on reported information about the specific chemical identity, the quantity and frequency of use, and available information on both acute and chronic toxicity.

6.3.1.1. Chemical Hazard Ranking Approach

Well stimulation (e.g., hydraulic fracturing and acidization) includes processes that use, generate, and release (intentionally and unintentionally) a wide range of chemical, physical, and, in some cases, biological stressors. To organize the large and diverse number of potential stressors, we use a hazard-ranking scheme that begins with a list of all identifiable stressors, and then records for each stressor our attempts to characterize potential outcomes, using measures of toxicity combined with information representing the frequency and magnitude of use. Sections 6.4 and 6.5 describe potential exposure pathways that would bring chemicals to a human population through water supply or air.

The hazard-ranking scheme used here gives weight to three factors— the number of times a chemical is reported in the database (a surrogate for frequency of use), mass or mass fraction (concentration) used, and toxic hazard screening criterion. So it is not the most toxic substances that always rank high, because weight is also given to substances of intermediate toxicity (or even relatively low toxicity) that are used frequently and/or in large quantities. Even with high mass and frequent use of compounds with elevated toxicity, an exposure pathway is required to bring the compound into contact with the human receptor for an adverse effect to be realized.

The disclosed mass and frequency of chemical use (as described in Section 2.4.3 for hydraulic fracturing and in Section 6.3.2.2 for acidization) provides a surrogate for potential chemical release and exposure, but this is only part of the hazard picture. It is also important to consider the impact of exposure to a chemical. Impacts considered in this assessment include both acute and chronic toxicity outcomes for individual chemicals. As noted above in Section 6.3, toxicity can be characterized as acute (short-term consequences from a single exposure or multiple exposures over a short period) or chronic (long-term consequences from continuous or repeated exposures over a longer period). It is not possible to evaluate potential synergistic hazards with multiple pollutants at this time.

For acute toxicity, we use a screening hazard criterion based on the Global Harmonization Score (GHS) that combines all acute toxicity information into a single screening value (UN, 2011). For chronic toxicity, we use published regulatory reference levels that consider information reported for different routes of exposure (inhalation, ingestion, dermal) and different health outcomes.

The ultimate goal of the hazard ranking is to combine the different elements that relate to increasing hazard. In considering specific chemical stressors, we used the information on frequency of use, mass or mass fraction used per treatment, and acute and/or chronic health hazard criteria, to develop a potential hazard score that could be used to assign a rank for each substance. In cases where all three pieces of information are available, the hazard score is calculated as an Estimated Hazard Metric (EHM) given by:

EHM = (frequency of use) × (mass or mass fraction used)/(toxicity criterion)

The calculated EHM are used to rank all substances from highest estimated hazard to lowest. For chemicals that lack sufficient information to calculate an EHM, we ranked from most toxic to least toxic, and when toxicity information is lacking we rank from most to least reported use. The resulting sorted list provides an indication of level of concern for each compound.

The development of acute and chronic toxicity criteria used for calculating the EHM are discussed in Sections 6.3.1.2 and 6.3.1.3, respectively, with the hazard ranking results for hydraulic fracturing and acidization presented in Sections 6.3.2.1 and 6.3.2.2, respectively.

6.3.1.2. Acute Toxicity Hazard Screening Criterion

Human hazards associated with acute or short-term exposures are inferred from laboratory studies that examine the acute toxicity of an individual compound or chemical formulations through standardized testing procedures using mammals—typically mice, rats, and rabbits. In these studies, the test animals are exposed to high concentrations of the test chemical and the response of the animals as a function of the exposure is determined, with the metric being the concentration at which some significant fraction of the animals have a measurable outcome (05%, 10%, 50%). These effective concentrations (EC) or effective doses (ED) are reported as respectively EC05 (EC05), EC10 (ED10), and EC50 (ED50).

We collected acute toxicity data for the chemicals that have been disclosed in well stimulation fluid in California that were definitively identified by their Chemical Abstract Service Registration Numbers (CASRN). Toxicity data were gathered from publicly available sources as described in Volume II, Chapter 2 and from MSDS. Acute toxicity data is available for a number of exposure routes and a range of effects. To merge this diverse data set into a single health-screening criterion, we used the United Nations Globally Harmonized System of Classification and Labeling of Chemicals (GHS). The GHS is a system for categorizing chemicals based upon their LD50 (lethal dose) or EC50 values (UN, 2011). In the GHS system, lower numbers indicate more toxicity, with a designation of “1” indicating the most toxic compounds. Chemicals for which the LD50 or EC50 exceeded the highest GHS category were assigned a value of 6 and classified as non-toxic. Chemicals that lack data on acute effects were assigned a GHS value of zero.

We also reviewed material safety data sheets (MSDS) for each chemical and recorded GHS values for a range of outcomes, including acute dermal, skin irritation, eye effects, respiratory sensitization, and skin sensitization. The GHS values from publicly available sources (oral and inhalation) were assessed separately from the GHS scores reported in MSDS.

Because the GHS is reported on a scale of 1 to 5, we found it to be ineffective for sorting out highly toxic chemicals. To address this issue for human health impacts, we converted the GHS category scores back to the midpoint exposure concentration for animal oral toxicity in milligrams per kilogram (mg/kg) for the given category, based on the definitions provided for GHS categories (Table 3.3-1 in UN, 2011). GHS categories 1, 2, 3, 4, and 5 were assigned equivalent toxicity criteria of 2.5, 25, 200, 1,150, and 3,500 mg/kg, respectively. We refer to this as the GHS-surrogate-concentration or “GHS-sc.”

Most stimulation chemicals are used at fairly low concentrations, usually less than 0.1%. These concentrations can be well below concentrations that would cause test animals to have a measureable acute response. However, most chemicals that have been assessed for toxicity are assessed with acute toxicity tests. Low-concentration responses are difficult to measure but highly relevant to efforts to protect human health. Public health actions are intended to prevent harm before it happens, rather than provide methods to monitor harm as it happens. This goal reflects the need for chronic hazard screening as a key supplement to acute hazard screening.

6.3.1.3. Chronic Toxicity Hazard Screening Criterion

Chronic toxicity values are typically expressed using a long-term average intake that is considered a “safe” or no-effect dose, expressed in mg/kg (body weight) per day. For example, the state of California issues reference exposure levels (RELs) in milligrams per kilogram per day (mg/kg/d) for a number of non-cancer chemicals. Acceptable chronic exposure levels for cancer-causing chemicals are selected to assure a minimum cancer risk, such as below 1 in 100,000. In developing a screening criterion for chronic toxicity, we select a single chronic screening score (CSS), which reflects the lowest acceptable chronic exposure in mg/kg/d across a broad range of chronic outcomes. Chronic health hazard screening values for hydraulic fracturing and acidizing fluid-treatment chemicals were developed from several sources of chronic toxicity information compiled by California and federal health agencies. These values indicate the likelihood of an adverse health outcome from repeated or continuous exposure over the long term.

Chronic toxicity screening criteria were developed separately for inhalation and oral exposure. Details on the compilation of chronic screening scores (CSS) for well stimulation chemicals are provided for the inhalation and oral routes of exposure in the following sections.

6.3.1.3.1. Chronic Screen Scores for the Inhalation Route

The following sources were used to identify screening values for the inhalation route of exposure.

1. Office of Environmental Health Hazard Assessment-derived (OEHHA) Reference Exposure Levels (RELs) for non-carcinogenic toxicants, and inhalation Unit Risk values (URs) for carcinogens (OEHHA, 2008; 2014a);

2. U.S. EPA toxicity criteria, which are similar to the OEHHA criteria in both form and method of derivation. U.S. EPA develops Reference Concentrations (RfCs) for non-carcinogens and Unit Risk Estimates (UREs) for carcinogens1 (U.S. EPA, 2014a; 2014b);

3. Agency for Toxic Substances and Disease Registry (ATSDR) Minimal Risk Levels (MRLs) for non-carcinogens, also similar to the OEHHA REL values (ATSDR, 2014).

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1. U.S. EPA’s Integrated Risk Information System (IRIS) was used as the primary source of information from U.S. EPA. In some cases, additional values were based on Provisional Peer Reviewed Toxicity Values (PPRTVs) derived by U.S. EPA’s Superfund Health Risk Technical Support Center, or U.S. EPA’s Health Effects Assessment Summary Tables.

For purposes of comparison, the available dose-response values were converted into a consistent scale of measurement, namely, a reference concentration in units of milligrams per cubic meter (mg/m3). Details and assumptions for calculating screening level dose-response values for chronic inhalation exposure are provided in Appendix 6.B. The reference concentrations were then converted to mg/kg/d equivalent dose, assuming a 20 m^3(5,283 gallons)/day inhalation rate and 70 kg (154 lbs) body weight. This value is meant only for ranking hazards across different routes of exposure; the original regulatory reference concentrations should be used in any subsequent assessment of risk.

6.3.1.3.2. Screening Values for the Oral Route

The following sources of toxicity information were used to identify hazard-screening values for the oral route of exposure:

1. OEHHA-derived values: Public Health Goals (PHGs) and Maximum Contaminant Levels (MCLs) for drinking water, “No Significant Risk Levels” (NSRLs), and Maximum Allowable Dose Levels (MADLs) for carcinogens and reproductive toxicants listed under Proposition 65 (OEHHA, 2014a; 2014b);

2. U.S. EPA: oral Reference Doses (RfDs) and cancer Slope Factors (SFs) (U.S. EPA, 2014a; 2014b);

3. ATSDR MRLs for oral exposure (ATSDR, 2014).

Oral route toxicity screening values are presented as mg/kg/d of oral intake. For details on derivation of chronic toxicity screening value for oral dose in this report, see Appendix 6.B.

6.3.2. Results of Human-Health Hazard Ranking of Stimulation Chemicals

This section provides results ranking hazards for chemical additives in hydraulic fracturing fluids (Section 6.3.2.1) and in acidization fluids (Section 6.3.2.2). In addition, we review hazards for chemicals released during well stimulation activity that are not directly added to the well (Section 6.3.2.3).

6.3.2.1. Hazard Ranking of Chemicals Added to Hydraulic Fracturing Fluids

The hazard ranking for hydraulic fracturing fluids is derived for all substances reported to be used in hydraulic fracturing that were definitely identified by CASRN. Additives without CASRN identification could not be assessed for toxicity screening values and thus were not included in the hazard ranking analysis. However, the absence of definitive identification for a chemical should not be interpreted as an indication that the specific additive is not hazardous.

For each disclosed additive, we use the available information on frequency of use in well stimulation (Section 2.4.3.1), quantity used (median concentration used across all well stimulation events) (Section 2.4.3.2), along with the GHS-based toxicity screening criterion for acute mammalian toxicity (normalized to exposure concentration as described in Section 6.3.1.2), and chronic screening values normalized to dose as derived from published values and regulatory values. We rank the acute and chronic hazards separately, and we include separate chronic rankings to reflect intake by inhalation or oral routes. For the acute toxicity information, we often had to rely on information that was only on material safety data sheets (MSDS), which is not always reliable but often the only toxicity information for specific health outcomes (e.g., eye irritation or sensitization). In cases where toxicity information from other published sources is available, we include separate hazard rankings using for results from material safety data sheets (MSDS) and from published sources. We base the ranking on the minimum, or most conservative, acute hazard value for each hazard ranking (i.e., with and without using MSDS data).

Out of 320 substances identified in the chemical disclosures (Table 2.A-1), 227 were definitively identified. We identified acute hazard screening values for 176 substances and chronic screening values for 56. The acute screening values are reported in Appendix 6.C Table 6.C-1. The chronic screening values are reported in Appendix 6.C Table 6.C-2. Four of the 56 compounds with chronic screening values did not have acute screening values, so we had a total of 176 compounds out of 320 (55%) for which we could develop a complete hazard ranking. There are an additional 23 compounds reported for which we have CASRN, but no information on frequency of use or mass used. Of these 23, we have an acute and/or chronic hazard screening value for 17. There are 121 substances for which we have generic descriptors ("trade secrets") and frequency of use information, but no CASRN identifications or toxicity information (note that chemicals without CASRN were not reviewed for toxicity). In Table 6.3-1 below, we summarize our findings regarding the different combinations of known versus unknown factors for reported hydraulic fracturing chemical additives.

Table 6.3-1. Available and unavailable information for characterizing the hazard of stimulation chemicals used in hydraulic fracturing.

Number of chemicals	Proportion of all chemicals	Identified by unique CASRN	Impact or toxicity	Quantity of use or emissions
176	55%	Available	Available	Available
17	5%	Available	Available	Unavailable
6	2%	Available	Unavailable	Available
121	38%	Unavailable	Unavailable	Available

Following the approach described above, we used information on frequency of use, quantity used, and health hazard screening criterion to derive an estimated acute hazard metric (EHM_{acute}) score for each of the 176 substances used in hydraulic fracturing that had sufficient information to make this calculation. All 176 EHM_{acute} scores are provided in

Table 6.C-1. The scores range over six orders of magnitude from 0.003 to 4,000. These are relative scores with higher values associated with higher concern. We used these scores to sort the substances from high to low. Table 6.3.2 lists the 12 substances with the highest EHM_{acute} values and identifies what factor(s) contribute most to this score—frequency of use, quantity used, and/or toxicity. The footnote to Table 6.3-2 indicates the acute toxicity and source of information for each chemical. Substances that did not have sufficient information to calculate EHM_{acute} values are sorted from low to high on a toxicity criterion; then for chemicals that lack a toxicity criterion, we sorted from high to low on frequency of use, then mass used, and finally the last chemicals are simply sorted alphabetically in Table 6.C-1.

Table 6.3-2. A list of the 12 substances used in hydraulic fracturing with the highest acute Estimated Hazard Metric (EHM_{acute}) values along with an indication of what factor(s) contribute most to their ranking (from high to low).

Chemical Name	**Reported frequency of use**	**Reported median mass fraction per WST (mg/kg)**	**Acute Toxicity**
Distillates, petroleum, hydrotreated light paraffinic	✔	✔	
Isotridecanol, ethoxylated	✔		✔[1]
Hydrochloric acid		✔	✔[2]
Polyethylene-polypropylene glycol	✔		✔[3]
Sodium hydroxide			✔[4]
Glyoxal		✔	✔[5]
Potassium carbonate	✔	✔	
Glutaraldehyde			✔[6]
Ammonium Persulfate	✔		✔[7]
Hydrofluoric acid		✔	✔[8]
Sodium tetraborate decahydrate	✔	✔	
5-Chloro-2-methyl-3(2H)-isothiazolone	✔		✔[9]

[1] Skin corrosion/irritation GHS = 1 per MSDS; [2] Skin sensitization and eye effects GHS = 1 per MSDS; [3] Inhalation LC50 for rats of 45 ppm equivalent to GHS 1 from published data; [4] Skin corrosion/irritation GHS = 1 per MSDS; [5] Eye effects GHS = 1 per MSDS; [6] Inhalation equivalent to GHS 1 per published values and Eye effects GHS = 1 per MSDS; [7] Respiratory sensitization GHS = 1 per MSDS; [8] Inhalation equivalent to GHS 2 per published values and dermal, skin corrosion/irritation and eye effects per MSDS; [9] Inhalation equivalent to GHS 1 per published values

In developing a chronic hazard metric (EHMchronic) score, we again make use of frequency of use, mass used per treatment, and health-hazard screening criterion for each of 55 substances used in hydraulic fracturing that had sufficient information to make this calculation. All 55 EHMchronic scores are provided in Table 6.C-2. The scores range over nine orders of magnitude from 200 to 400,000,000,000 and tend to be higher for the inhalation route compared to the oral route. These are relative scores with higher values associated with higher concern. We used these scores to sort the substances from the highest to lowest estimated EHMchronic sorted on the average rank across inhalation and oral routes. The median chronic score is around 1 million. The top 12 substances for chronic hazard all have EHMchronic values over 1 million. Table 6.3-3 lists the 12 substances with the highest EHMchronic values and identifies what factor(s) contribute most to this score—frequency of use, quantity used, or toxicity. Substances with neither an EHM_{acute} or EHMchronic value are listed in Table 6.C-1, but not repeated in Table 6.C-3.

Table 6.3-3. A list of the 12 substances used in hydraulic fracturing with the highest chronic Estimated Hazard Metric (EHMchronic) values along with an indication of what factor(s) contribute most to their ranking (from high to low).

Chemical Name	Reported frequency of use	Reported median conc. per WST (mg/kg)	Chronic[8] Toxicity
Proppant material[1]		✔	✔[1]
Glutaraldehyde	✔	✔	✔
Zirconium oxychloride	✔	✔	✔[2]
Bromic acid, sodium salt (1:1)		✔	✔[3]
Hydrochloric acid	✔	✔	✔
Boron sodium oxide	✔	✔	✔[4]
Ethylbenzene		✔	✔
Naphthalene	✔		✔
Sodium tetraborate decahydrate	✔	✔	✔[5]
Boric acid, dipotassium salt		✔	✔[6]
Aluminum oxide		✔	✔[7]
Diethanolamine		✔	✔[6]

[1] Proppant materials reported that might include Crystalline silica impurity (Mullite, Kyanite, Silicon dioxide) use Crystalline silica impurity as reference chemical for hazard screening (inhalation); [2] Soluble Zirconium compounds used as reference chemical for hazard screening (oral); [3] Boric Acid and Bromate used as reference compound for hazard screening (oral) and (inhalation) respectively; [4] Boric acid used as reference chemical for hazard screening (oral); [5] Boric Acid used as reference compound for hazard screening (oral); [6] Boric acid used as reference chemical for hazard screening (oral); [7] The toxicity value used is only for non-fibrous forms of aluminum oxide, and does not apply to fibrous forms; [8] Screening toxicity values for aluminum oxide, titanium oxide, propargyl alcohol, glyoxal, butyl glycidyl ether, hydrogen peroxide, and ethanol are available for occupational health criteria but screening values are not provided because for each of these substances, there was an indication in the literature of possible mutagenicity or carcinogenicity such that the available occupational health criteria might not be sufficiently health protective of workers and the general population.

6.3.2.2. Hazard Ranking of Acidization Chemicals

The data used to characterize hydraulic fracturing fluids did not include disclosed acidization events. However, the South Coast Air Quality Management District (SCAQMD) rule 1148.2 mandates that operators disclose the chemicals used in oil and gas development activities that include acidization. Acidization events are defined for the purpose of this review as events that include hydrochloric acid (HCl) and/or hydrofluoric acid (HF). The data that meets the definition of an acidization event were exported from data entered into the SCAQMD database between July 2013 and May 2014. The data include 243 events in 243 wells with a total of 8,549 entries for individual chemicals or "trade secrets" (listed by chemical family). The actual date of each event is not listed, but it appears that most of the data was entered into the database between March and May of 2014.

As with the hydraulic fracturing fluid disclosures, not all additives in the acidization events were clearly identified. Between 3 and 21 lines (ingredients in the acidization event) for each event are reported as trade secret, with no information provided on mass, composition, or definitive chemical identification. A total of 87 definitively identified chemicals are listed for the acidization events with 33 chemicals unique to acidization (i.e., not used in hydraulic fracturing). The remaining 54 chemicals are used in both acidization (per SCAQMD disclosures) and hydraulic fracturing (per FracFocus disclosures). It is unclear which if any disclosures for specific events are included in both databases.

Twenty-six chemicals were listed more than 50 times in the acidization notices, with methanol (n = 532), hydrochloric acid (n = 436) and propargyl alcohol (n = 272) being the most commonly reported chemicals used in acidization events (excluding water). There are 45 chemicals listed fewer than five times. Data are not available to assess the coverage of the SCAQMD disclosures relative to all acidization treatments in California, but clearly the data provided in the SCAQMD database are specific for activity in the South Coast Air Basin which includes Orange County and the non-desert regions of Los Angeles and Los Angeles County, San Bernardino County, and Riverside County.

Twelve chemicals are reported with median application rate greater than 200 kg per event, but several of these are either base fluid or proppant material. The reporting of proppant indicates that there may be some overlap between acidization treatments and fracturing treatments in the SCAQMD database. The remaining high-use chemicals in the list include primarily acids and buffering compounds. For chemicals that are used in both hydraulic fracturing and in acidization treatments, a comparison of the reported mass used indicates that there is no correlation ($r^2 = 0.01$) between median mass reported for specific compound used in the SCAQMD acidization treatments and the FracFocus/ DOGGR (Division of Oil, Gas and Geothermal Resources) hydraulic fracturing treatments.

In order to develop a hazard ranking for acidizing fluids, we follow the procedure outlined above for hydraulic fracturing fluids to compile a list of all substances for which we had CASRN and provided, for each chemical, any available information on frequency of use in well stimulation, quantity used in each well stimulation, the GHS screen criterion for acute toxicity, and available chronic screening criteria. The frequency used and quantity used are specific to the acidization treatments and differ from values reported for the same chemical in the assessment of hazard for stimulation chemicals used in hydraulic fracturing (previous section). The data used to assess acidization did not provide information that would allow the calculation of mass fraction or concentration as used in the hydraulic fracturing assessment above, so the media mass (kg) used across all events was used as a surrogate for quantity. The acute screening values for acidization chemicals are reported in Appendix 6.C, Table 6.C-3. The chronic screening values are reported in Appendix 6.C, Table 6.C-4. Out of 165 uniquely identified additives (or products), 78 compounds were identified with CASRN, 48 had both quantity and toxicity information, and 39 had only quantity information. In Table 6.3-4 below, we summarize our findings regarding these different combinations of known versus unknown factors.

Table 6.3-4. Available and unavailable information for characterizing the hazard of stimulation chemicals use in acidizing.

Number of chemicals	Proportion of all chemicals	Identified by unique CASRN	Impact or toxicity	Quantity of use or emissions
48	29%	Available	Available	Available
0	0%	Available	Available	Unavailable
39	24%	Available	Unavailable	Available
78	47%	Unavailable	Unavailable	Unavailable

Following the approach described above and used for hydraulic fracturing chemicals, we used the information on frequency of use, quantity used, and toxicity screening criterion to derive an estimated acute hazard metric (EHM_{acute}) score for each of the 48 substances used in acidization that had sufficient information to make this calculation. All 48 EHM_{acute} scores are provided in Table 6.C-3 along with information for other substances for which the score could not be determined. The scores range over eight orders of magnitude from 0.002 to 150,000. These are relative scores with higher values associated with higher concern. We used these scores to sort the substances from high to low on the average EHM between results, including MSDS data and results based on published toxicity data. The median score is around 1. Table 6.3-5 lists the 10 substances with the highest EHM_{acute} values and identifies what factor(s) contribute most to this score—frequency of use, quantity used, or toxicity. Substances with no EHM_{acute} are sorted by decreasing concentration.

In developing a chronic hazard metric (EHMchronic) score for acidization chemicals, we again make use of frequency of use, mass used per treatment, and health hazard screening values for each of 17 substances used in acidization that had sufficient information to make this calculation. All 17 EHMchronic scores, along with toxicity and use-frequency data for substances that did have reported mass used, are provided in Table 6.C-6. The scores range over eight orders of magnitude from 10 to 800,000,000, and tend to be higher for the inhalation route than the oral route. These are relative scores with higher values associated with higher concern. We used these scores to rank the substances from 1 to 17, with 1 being the greatest estimated hazard rank. The median chronic score is around 10,000. Table 6.3-6 lists the 10 substances with the highest EHMchronic values and identifies what factor(s) contribute most to this score—frequency of use, quantity used, or toxicity.

Table 6.3-5. A list of the 10 substances used in acidization with the highest acute Estimated Hazard Metric (EHM_{acute}) values, along with an indication of what factor(s) contribute most to their ranking (from high to low).

Chemical Name	**Reported frequency of use**	**Reported median mass per WST (kg)**	**Acute Toxicity**
Hydrochloric acid	✔		✔[1]
Hydrofluoric acid	✔		✔[2]
Potassium chloride		✔	
Ammonium Chloride	✔	✔	✔[3]
Citrus Terpenes			✔[4]
2-Butoxyethanol (Ethylene glycol butyl ether)	✔		✔[5]
Propargyl alcohol	✔		✔[6]
Acetic Acid			✔[7]
Crystalline silica quartz		✔	
Citric acid	✔	✔	✔[8]

[1] Skin sensitization and eye effects GHS = 1 per MSDS; [2] Inhalation equivalent to GHS 2 per published values and dermal, skin corrosion/irritation and eye effects per MSDS; [3] Eye effects GHS = 2 per MSDS; [4] Skin corrosion/irritation GHS = 1 and eye effects GHS = 2 per MSDS; [5] Inhalation effects GHS 2 from published data and eye effects GHS = 2 per MSDS; [6] Oral effects GHS 2 from published data and numerous effects with GHS = 1 or 2 per MSDS; [7] Skin corrosion/irritation GHS = 1 and eye effects GHS = 1 per MSDS; [8] Eye effects GHS = 2 per MSDS

Table 6.3-6. A list of the 10 substances used in acidization with the highest chronic Estimated Hazard Metric (EHMchronic) values along with an indication of what factor(s) contribute most to their ranking (from high to low).

Chemical Name	Reported frequency of use	Reported median mass per WST (kg)	Chronic Toxicity
Hydrochloric acid	✔		✔
Propargyl alcohol			✔
Crystalline silica quartz		✔	✔
Ethylbenzene			✔
Ammonium Chloride		✔	✔
Hydrofluoric acid			✔
2-Butoxyethanol (Ethylene glycol butyl ether)			✔
Acetic Acid		✔	
Methanol	✔		
Phosphoric acid, calcium salt (2:3)			✔

6.3.2.3. Hazard Summary of Air Pollutants that are Related to Well Stimulation Fluid

There are fifteen chemicals listed in Tables 6.C.1– 6.C.4 for hydraulic fracturing and acidization activity that are also listed on the California Air Resources Board (CARB) Toxic Air Contaminant (TAC) Identification List (CARB, 2015). These compounds are listed in Table 6.3-7, along with an indication of the well stimulation activity that they are reportedly used in. Five of the compounds listed on the TACs list are already identified in the previous tables, but all compounds listed as TACs should be considered hazardous and included in subsequent risk assessments. The California TACs list (CARB, 2015) includes all Hazardous Air Pollutants (HAPs) listed by the U.S. EPA and are heavily regulated compounds.

Table 6.3-7. The substances used in hydraulic fracturing and acidization that are also listed on the California TAC Identification List (http://www.arb.ca.gov/toxics/id/taclist.htm).

Chemical Name	**CASRN**	**Used in Hydraulic Fracturing**	**Used in Acidization**
Hydrochloric acid	7647-01-0	✔	✔
Methanol	67-56-1	✔	✔
Toluene	108-88-3		✔
Acetophenone	98-86-2		✔
Ethylene Glycol	107-21-1	✔	✔
Formaldehyde	50-00-0	✔	✔
Naphthalene	91-20-3	✔	✔
Diethanolamine	111-42-2	✔	
Benzyl Chloride	100-44-7	✔	
Acrylamide	79-06-1	✔	

Volume III, Chapter 3 summarizes a list of all CARB-reported TACs air emissions associated with all oil-well production activities including well stimulation fluids (Chapter 3, Section 3.3.2.2). We noted that not all of the TACs listed above are reported emissions—likely as a result of different requirements for reported use versus reported emissions. It is not possible at this point to allocate the CARB-reported emissions specifically to the use well stimulation fluids. In addition to chemicals added to well stimulation fluids, there a number of TACs released during well stimulation activities that are not added directly to the well. As TACs, these substances have all been identified as posing human health hazards, with the actual health risk dependent on the magnitude and duration of exposure. Among this substance list are combustion products and/or chemical emissions from pumps, generators, compressors, and equipment; venting and flaring; dust from well stimulation activity; leaks from transfer lines and/or well heads; and emissions related to leakage of oil and gas from stimulated wells (this category does not include emissions from refining and use of the hydrocarbon products). A variety of mobile sources relevant to oil and gas (and presumably to well stimulation) activities are tracked by CARB in its emissions inventories (See Chapter 3, Section 3.3.2.2), especially for off-road diesel equipment. However, it is not clear how to apportion these activities between conventional oil production and well stimulation activities without a much more detailed study.

Several criteria pollutants (particulate matter, carbon monoxide, nitrogen oxides, and sulfur dioxide) as well as reactive organic gases are associated with well stimulation activities (see Section 3.3.2.2 for details on emissions estimates). Criteria pollutants are heavily regulated and should be included in any hazard or risk assessment associated with well stimulation. Given the known and accepted hazards associated with criteria pollutants, no further hazard assessment is provided for these compounds in this chapter.

6.3.3. Literature Summary of Human Health Hazards Specific to Well Stimulation

In the sections above, we made bottom-up characterizations and rankings of chemicals used and/or emitted during well stimulation operations in California. This section reviews and analyzes the chemical hazards of well stimulation chemicals based primarily on published source categories related to well stimulation activities and associated equipment. Much of the literature discussed below is associated with activities outside of California, but offers insights on what is or could be done in California.

Colborn et al. (2011) used Chemical Abstract Service (CAS) numbers and systematic searches in the National Library of Medicine, Toxicology Data Network (TOXNET) and other databases to determine that (a) 75% of the identified compounds from fracturing fluids in samples from Colorado are known to negatively impact sensory organs, the gastrointestinal system, and/or the liver; (b) 52% of the identified chemicals have the potential to adversely affect the nervous system; and (c) 37% are candidate endocrine disrupting chemicals (EDCs). EDCs present unique hazards compared to other toxins, because their effects at higher doses do not always predict their effects at lower doses (Vandenberg et al., 2012). They are particularly hazardous during fetal and early childhood growth and development (Diamanti-Kandarakis et al., 2009), can impact the reproductive system, and have epigenetic mechanisms that may lead to pathology decades after exposure (Zoeller et al., 2012).

In addition to the chemicals used in well stimulation, the major constituents of well acidization fluid are hydrochloric acid and hydrofluoric acid. Hydrochloric acid is used frequently in oil and gas wells in California and elsewhere as an additive to well-injection fluids during matrix acidizing, wellbore cleanout, and other forms of acid treatments of oil and gas wells (Colborn et al., 2011; Stringfellow et al., 2014) (also see Volume I for more details). Hydrochloric acid is corrosive to the skin, eyes, and mucous membranes, and is associated with a number of acute health effects (ATSDR, 2002). Oral exposure may result in the corrosion of mucous membranes, the esophagus, and the stomach. Symptoms may include nausea, vomiting, and diarrhea (U.S. EPA, 2000a). Dermal exposure may result in severe burns, ulceration, and scarring. Chronic exposures in occupational settings are associated with gastritis, chronic bronchitis, dermatitis, and photosensitization (U.S. EPA, 2000a). As discussed in the occupational health section below, we note that exposure to acid vapors resulting in acid-vapor inhalation is a hazard for any unprotected individuals close to the location of acid use or transfer.

Hydrofluoric acid is also used as an additive to well injection during matrix acidizing, wellbore cleanout, and other forms of acid treatments of oil and gas wells (Colborn et al., 2011; Stringfellow et al., 2014) (See Volume I). Acute exposure to hydrofluoric acid in liquid and gaseous form causes irritation of the eyes and nose, and can result in severe respiratory damage (Centers for Disease Control and Prevention (CDC), 2014). In high doses, exposure to hydrofluoric acid can lead to convulsions, cardiac arrhythmias, or death from cardiac or respiratory failure (U.S. EPA, 2000b). Chronic exposure to elevated concentrations of hydrofluoric acid is associated with adverse pulmonary effects, renal injury, thyroid injury, anemia, hypersensitivity, and dermatological reactions (U.S. EPA, 2000b). When inhaled at low concentrations, hydrofluoric acid can result in nose, throat, and bronchial irritation and congestion (ATSDR, 1993; CDC, 2014). To date, no studies on the public health dimensions of hydrofluoric and hydrochloric acid have been conducted in the upstream oil and gas context.

6.4. Water Contamination Hazards and Potential Human Exposures

This section reviews the transport mechanisms that could cause human exposures to stimulation chemicals through water contamination. Section 6.4.1 briefly reviews the pathways identified in Volume II, Chapter 2, and summarized in Table 6.2-1, and discusses implications for human health. This is followed by Section 6.4.2, which provides a literature survey of health issues attributed to water contamination due to stimulation.

A direct impact of concern from chemical use for well stimulation is the potential for water contamination and subsequent human exposure from accidental releases related to the handling of the well stimulation fluids and the management of produced water that may contain stimulation chemicals. Similarly, potential subsurface leakage pathways into protected groundwater present a potential impact of contamination by the petroleum constituents in the reservoir. This risk may be exacerbated by the presence of chemicals used in hydraulic fracturing. If chemicals contained in well stimulation fluids are well managed and not released into usable water, including agricultural water, then the public health risks would be reduced. Acid use increases the probability that naturally occurring heavy metals and other pollutants from the oil-bearing formation will be dissolved and mobilized. Assessment of the environmental public health risks posed by acid use along with commonly associated chemicals, such as corrosion inhibitors, cannot be undertaken without a more complete disclosure of chemical use, and a better understanding of the chemistry of treatment fluids and produced water returning to the surface, in order to understand the risks these fluids may pose. Risk assessment would also require better knowledge of potential transport mechanisms and pathways that could lead to human exposure, as well as how treatment chemicals are altered during transport.

6.4.1. Summary of Risk Issues Related to Water Contamination Pathways

The potential for surface and groundwater contamination from well stimulation activities (contamination with stimulation chemicals, recovered fluids and produced water, residual oil, methane and other compounds) was evaluated in great detail in Chapter 2 of this volume. Release mechanisms and environmental transport pathways associated with well stimulation and production that are relevant to California include spills and leaks, percolation of wastewater from unlined pits, siting of disposal wells near abandoned wells or into protected groundwater, reuse or disposal of inadequately treated wastewater; loss of wellbore integrity; subsurface leakage and migration through abandoned wells, migration though faults, fractures, or permeable regions, and illegal waste discharge

(Section 2.6.2). Some of these release mechanisms are primarily relevant to California, and are uncommon elsewhere, such as disposal of wastewater in unlined percolation pits, which has been banned in many states, and potential siting of disposal wells into protected groundwater. However, many of the release mechanisms have also been noted in other parts of the country. Below, we briefly summarize the main findings from Chapter 2 with regard to release mechanisms and transport pathways of concern for human health impacts.

Stimulation fluids can move through the environment and come into contact with human populations in a number of ways, including surface spills, accidental releases (Rozell and Reaven, 2012), loss of zonal isolation in wellbores (Chilingar and Endres, 2005; Darrah et al., 2014), venting and flaring of gases (Roy et al., 2013; Warneke et al., 2014), and transportation and disposal of wastes (Rozell and Reaven, 2012; Warner et al., 2012; Warner et al., 2013a; Fontenot et al., 2013).

6.4.1.1. Disposal of Produced Water in Unlined Pits

As noted in Volume II, Chapter 2, the most commonly reported recovered fluids and produced water disposal method for stimulated wells in California is by evaporation and percolation in unlined surface impoundments, also referred to as unlined sumps or pits. Operators report that nearly 60% of the produced water from stimulated wells was disposed of in unlined sumps during the first full month after stimulation. There is no testing required, or thresholds specified, for the contaminants found in well stimulation fluids or other naturally occurring chemical constituents in produced water, such as benzene, heavy metals, and naturally occurring radioactive materials (NORMs). The primary intent of unlined pits is to percolate water into the ground, and as a result, this practice provides a potentially direct subsurface pathway for the transport of produced water constituents, including returned stimulation fluids, into groundwater aquifers that are or may be used for human consumption and agricultural use. Where groundwater intercepts rivers and streams, surface water resources could also be affected. If protected water were contaminated and if plants (including food crops), humans, fish, and wildlife use this water, it could introduce contaminants into the food web and expose human populations to known and potentially unknown toxic substances.

6.4.1.2. Public Health Hazards of Produced Water Use for Irrigation of Agriculture

As noted in Volume II, Chapter 2, large volumes of water of various salinities and qualities are produced along with oil. Most produced water is re-injected into the oil and gas reservoirs to help produce more oil, maintain reservoir pressure, and prevent subsidence. But some of this produced water is not highly saline, and small quantities of it are now being used by farmers for irrigation. As discussed in Chapter 2 of this volume, concerns arise that stimulation chemicals could be mixed with produced water and thus end up in irrigation water. Because of the growing pressures on water resources in the state, there is increasing interest in whether produced water could be used for a range of beneficial purposes such as groundwater recharge, wildlife habitat, surface waterways, irrigation, and other uses. If produced water comes from an oil field where well stimulation has been used, stimulation chemicals could also be present in the produced water and would not necessarily be detected by current testing. The presence of stimulation chemicals and other naturally occurring constituents, such as heavy metals that could be mobilized by stimulation chemicals makes it far more difficult to determine if the produced water can be safely reused. The presence of stimulation chemicals also makes it more difficult to determine the amount and type of water treatment required to make the water safe for beneficial use in agriculture from a public health perspective.

6.4.1.3. Public Health Hazards of Shallow Hydraulic Fracturing

Deep fracturing operations are unlikely to produce fractures and conduits that intersect fresh water aquifers far above them (See Volume I of this study for more details). However, in California, about three quarters of the hydraulic fracturing takes place in shallow wells less than 600 m deep. Where drinking water aquifers exist above shallow fracturing operations, there is an inherent risk that hydraulic fractures could intersect aquifers used for drinking, agriculture, and other uses and contaminate them, thus introducing human exposure pathways and public health risks. To the extent that human populations are drinking, washing, or using water that has been contaminated via this environmental exposure pathway, there exists a public health risk (See Chapter 2 of this volume for me details water exposure pathways).

6.4.1.4. Leakage Through Wells

One of the problems faced in a number of other states is oil and gas development in regions that have not previously had intensive oil and gas development. California's experience with well stimulation is the opposite: most well stimulation is occurring in reservoirs where oil and gas has been produced for a long time. This means the operations are taking place where many wells have previously been drilled, plugged, abandoned, and orphaned. Leakage can occur if a hydraulic fracture intersects another well (offset well). Offset wells can also act as a conduit through which emissions to air and water resources can occur. If protected water is contaminated and if plants (including food crops), humans, fish, and wildlife use this water, it could introduce contaminants into the food web and expose human populations to known and potentially unknown toxic substances. Because geologic conditions in California result in almost no coal mining, we did not consider leakage facilitated by abandoned coal mines, which is a problem in other states.

6.4.1.5. Injection Into Usable Aquifers

In June 2014, the U.S. EPA expressed concerns to the state of California regarding an EPA evaluation of injection wells in California used to dispose of oil-field waste, primarily recovered fluids and produced water that returns to the wellhead along with oil (U.S. EPA, 2014c). The EPA found that some wells inappropriately allowed injection of waste into protected groundwater. The California Division of Oil, Gas and Geothermal Resources (DOGGR) has shut down some of these wells and is reviewing many more for possible violations. Some chemicals that are used in well-stimulation operations are known to be toxic, but more than 50% of reported well stimulation chemicals in California have unknown environmental and health profiles. Some of the naturally occurring constituents in produced water are also toxic. Introduction of recovered fluids or produced water into protected groundwater presents a risk to the health of human populations that may drink, bathe, or irrigate with these water supplies.

6.4.2. Literature on Water Contamination from Well Stimulation

6.4.2.1. Exposure to Water Pollutants

We identified original research, including modeling studies on the potential for exposures to water quality impairment associated with oil and gas development enabled by well stimulation. We excluded studies that explored only evaluative methodology or baseline assessments, as well as papers that simply comment on or review previous studies. Papers on the potential for exposure to well-stimulation-associated contaminated water (a) rely on empirical field measurements, (b) explore plausibility of mechanisms for contamination, or (c) use modeled data to determine hazard and risk associated with potential water exposure pathways. Some of these studies explore only one aspect of shale gas development, such as the well-stimulation process of hydraulic fracturing. These studies do not indicate whether well-stimulation-enabled oil and gas development as a whole is associated with water contamination and are therefore limited in their utility for gauging water quality impacts. We are only concerned with actual findings in the field or modeling studies that specifically identify hazard, or actually document the occurrence or non-occurrence of water contamination.

Surface and groundwater contamination from well-stimulation-enabled oil and gas development is extensively documented in Chapter 2 of this volume. But the question of potential health risks remains, especially given the dearth of investigations and monitoring on this issue in California. Some association studies have reported that well stimulation contributes to higher levels of methane in drinking-water wells within 1 km of active gas development sites (Darrah et al., 2014; Jackson et al., 2013; Osbourne et al., 2012). Other studies found no association and have suggested that methane contamination of shallow groundwater from oil and gas production may be less likely to occur in certain shale formations, owing in part to regional geological variations, including the presence of intermediate gas-bearing formations above target formations (e.g., in the Pennsylvania area of the Marcellus Shale region), but not others (e.g., in the Fayetteville shale region) (Warner et al., 2013b). The most recent study on fugitive gas contamination of drinking-water wells used noble gas data to implicate faulty well production casings in water contamination rather than upward migration of methane through geological strata triggered by hydraulic fracturing (Darrah et al., 2014). While methane is not considered to be toxic, these studies suggest that there are subsurface pathways through which gases and liquids, some of which may contain hazardous compounds, may be present. Methane—particularly thermogenic methane (Stolper et al., 2014)—can migrate and mix with protected water through natural seepages (Dusseault et al., 2014; Dusseault and Jackson, 2014). Such seepages are common in California. Investigations of aquifer contamination attributable to oil and gas development have not been conducted in California. There is a need for these investigations, including studies to determine the effect of natural seepages in methane migration.

Other studies that evaluated water quality in private drinking-water wells near natural gas operations found higher levels of arsenic, selenium, strontium, and total dissolved solids in water wells located within 3 km of active gas wells (Fontenot et al., 2013). While this study used historical data from the region as a baseline to link the water contamination to natural gas development, the specific mechanism responsible for contamination was not determined.

Water contamination events associated with well stimulation have been documented in geographically diverse parts of the country. In Colorado, an analysis of 77 reported surface spills (~0.5% of active wells) within Weld County and groundwater monitoring data revealed BTEX (benzene, toluene, ethylbenzene, xylene) contamination in groundwater (Gross et al., 2013). Another study in Colorado measured estrogen and androgen receptor activity in surface and groundwater samples, using reporter gene assays in human cell lines from drilling-dense areas in the Piceance basin (Kassotis et al., 2013). Water samples collected from the more intensive areas of natural gas extraction exhibited statistically significantly more estrogenic, antiestrogenic, or antiandrogenic activity than reference sites. Notably, the concentrations of chemicals detected by Kassotis and colleagues (2013) were high enough to potentially interfere with the response of human cells to male sex hormones and estrogen.

In August 2014, the Pennsylvania Department of Environmental Protection (PA DEP) announced that 243 cases of water contamination attributable to oil and gas development in the region had occurred since 2008, and as of 4 March 2015, the number of confirmed water contamination cases was 254 (PA DEP, 2014). While this database makes clear that these cases of water contamination were caused by oil and gas development, it is not clear which mechanisms were most prominent. However, the presence of methane and other VOCs in the aquifers suggests that loss of wellbore integrity was a likely mechanism among the many of the cases. The majority of the events occurred in the northeastern region of the state; however, reasons for this geographic trend are still unknown and are currently being investigated. More research is needed to determine if wellbore integrity is associated with these events and if that integrity is affected by hydraulic fracturing.

6.4.2.2. Oil and Gas Recovered and Produced Water

Well stimulation generates recovered fluids and produced water. Evidence indicates that approximately 35% of the initial fracturing fluid volume injected underground returns to the surface as recovered fluids and produced waters, although estimates range from 9% to

80% (U.S. EPA, 2004, 2010; Horn, 2009). Recovered fluids and produced water contain chemical compounds added to fracturing fluids as well as naturally occurring compounds that are mobilized from target geological features (Alley et al., 2011; Thurman et al., 2014; Warner, 2013a). Compounds hazardous to human health identified in produced waters include chlorides, heavy metals, and metalloids (e.g., cadmium, lead, arsenic), volatile organics (e.g., benzene, toluene, ethylbenzene, and xylene), bromide, barium, and, depending upon the geochemistry of the target reservoir, naturally occurring radioactive materials (e.g., radium-226 and radon) and other compounds (Alley et al., 2011; Maguire-Boyle and Barron, 2014; Nelson et al., 2014). Many of these naturally occurring compounds have moderate to high toxicity and can induce health effects when exposure is sufficiently elevated (Balaba and Smart, 2012; Haluszczak et al., 2013). It should be noted that no studies to date have analyzed the chemical constituents of recovered fluids and produced water from well-stimulation-enabled oil wells in California.

Recovered fluid and produced water are sometimes treated at publicly owned treatment works (POTWs) and then discharged into surface waters (Ferrar et al., 2013). This practice is currently applied to a subset of recovered fluid/produced water in California (DOGGR, 2014) (also see Chapter 2 on impacts to water resources). Warner et al. (2013a) examined water quality and isotopic compositions of discharged effluents, surface waters, and stream sediments associated with a Marcellus wastewater treatment facility site. This study reported that treated recovered fluid and produced water still contained some elevated concentrations of contaminants associated with shale gas development. The researchers also found elevated levels of chloride and bromide downstream, along with radium-226 levels in stream sediments at the point of discharge that were approximately 200 times greater than upstream and in background sediments, and well above regulatory standards (Warner et al., 2013a). The study did not differentiate what amounts of these elevated concentrations were directly attributable to hydraulic fracturing. Some papers have noted that these types of emissions to water supplies could increase the health risks of residents who rely on these surface and hydrologically contiguous groundwater sources for drinking, bathing, recreation (Wilson and VanBriesen, 2012), and sources of food (i.e., fish protein) (Papoulias and Velasco, 2013).

6.5. Air Emissions Hazards and Potential Human Exposures

In addition to the potential direct impacts of water contamination, there is the possibility of direct public health risks of exposures to stimulation chemicals that are known toxic air contaminants (TACs). In Volume II Chapter 3, we analyzed the SCAQMD mandatory oil and gas reporting database and noted TACs have been reported as used in hydraulic fracturing and acidizing fluids. All of these TACs are hazardous to human health, yet none of them have known emission factors. This makes it difficult to assess the extent to which populations may be exposed and at what concentrations. Section 6.5 below expands this topic. This section reviews the potential human health impact of air emissions associated with well stimulation in two parts. Section 6.5.1 reviews what is known about air emissions from the assessment in Chapter 3 and elsewhere. Section 6.5.2 reviews the literature on human health impacts.

6.5.1. Emissions Characterized in Chapter 3

As discussed in Chapter 3 of this volume, air emissions from oil and gas development can come from a variety of sources, including, but not limited to drilling, production processing, well completions, servicing, and transportation. Among *known* air contaminants, compounds of particular concern that are known to be emitted during the well-stimulation-enabled oil and gas development process (and from oil and gas development in general) are BTEX compounds (benzene, toluene, ethylbenzene, and xylene), formaldehyde; hydrogen sulfide; particulate matter (PM); nitrogen oxides (NO_x); sulfur dioxide (SO_2); polycyclic aromatic, aliphatic, and aromatic hydrocarbons; and volatile organic compounds (VOCs) that can contribute to tropospheric ozone formation.

Also discussed in Chapter 3 of this volume are methane emissions, which are currently assessed as greenhouse gases but can also be used as a predictor of many VOC emissions. Some VOCs are directly health damaging (e.g., benzene), and many others are precursors to regional tropospheric ozone, a strong respiratory irritant. In the San Joaquin Valley Unified Air Pollution Control District (APCD), 2012 oil and gas associated reactive organic gas (ROG) emissions were approximately 8% of total regional ROG emissions (see Chapter 3). In a field-based study in the San Joaquin Valley of California, Gentner et al. (2014) found that at least 22% of all anthropogenic VOC emissions are attributable to oil development.

The quantity of specific chemicals emitted to the atmosphere per unit of injected well stimulation fluid is completely lacking from the existing literature. Compounds noted in the previous paragraph can be emitted or released prior to use during transport, transfer, blending, and injection by accidental release, intentional release or by fugitive emission pathways. After injection of fluid into the well-bore, the release pathways and emission rates become even more uncertain, because of a lack of knowledge about the recovered fraction of well stimulation fluid and changes in composition of recovered fluid and produced water at stimulated wells. There are a number of potential release pathways to air for the stimulation fluids recovered from a treated well, including both intentional (evaporation ponds, agricultural use, re-injection) and accidental (spills, transportation, disposal and fugitive emissions). None of these potential emission pathways for down-hole TACs is sufficiently characterized beyond the frequency and total mass estimates derived in Chapter 2.

Emission rates for TACs that are indirectly related to well stimulation activity are based on activity-specific emission factors that report the quantity of a pollutant released to the atmosphere relative to an activity associated with the release of that pollutant. Emission factors are provided by regulatory agencies such as the U.S. EPA. Generic or generalizable emission rates are not available at the wellhead scale. Estimating emission rates depends on the combination of site-specific activities and equipment (e.g., number of stationary and mobile source, leakiness of transfer lines and connections). However, all TACs by definition are hazardous, so they should be included in any thorough risk assessment for well stimulation activity using case-specific conditions and emission factors to determine ultimate exposures and quantify risk.

6.5.2. Potential Health-Relevant Exposure Pathways Identified in the Current Literature

6.5.2.1. Air Emissions Exposure Potential

Based on the potential harm of a number of VOCs (i.e., benzene, toluene, ethylbenzene, xylene, etc.) and the role of VOCs in the production of tropospheric ozone, we considered studies that address methane *and* non-methane volatile organic compounds (VOC) emissions. We considered papers that specifically address human exposures from well stimulation (i.e., unconventional oil and gas development) at either a local or regional scale. These include local and regional measurements of non-methane volatile organic compounds and tropospheric ozone.

As discussed in Chapter 3 of this volume, emissions from oil and gas development can come from a variety of sources including, but not limited to, drilling, processing, well completions, servicing, and transportation. Of particular concern are BTEX compounds (benzene, toluene, ethylbenzene, and xylene), other VOCs; formaldehyde; hydrogen sulfide; methylene chloride; particulate matter (PM); nitrogen oxides (NO_x); sulfur dioxide (SO_x); polyaromatic, aliphatic, and aromatic hydrocarbons; and tropospheric ozone.

An issue of potential concern in California is tropospheric (ground-level) ozone, which is formed through the interaction of VOCs, and NO_x in the presence of sunlight (Jerrett et al., 2009; U.S. EPA, 2013). Tropospheric ozone is a strong respiratory irritant associated with increased respiratory and cardiovascular morbidity and mortality (Jerrett et al., 2009; UNEP, 2011). However, as noted in Chapter 3 of this volume, the oil and gas industry is currently not a major contributor to tropospheric precursors in California air basis. There is some research on tropospheric ozone production associated with oil and gas development operations in other states. Modeling studies in the Haynesville and Barnett shale plays have predicted substantially increased atmospheric ozone concentrations associated with oil and gas development in Texas (Kemball-Cook et al., 2010; Olaguer, 2012; Gilman et al., 2013). Some observations in oil and gas producing basins in the western U.S. have found high levels of ozone in the winter, often in excess of air quality standards (Edwards et al., 2014). Nevertheless, as discussed in Volume II Chapter 3 and in contrast to the studies noted above, the ozone levels in California air basins are mostly dependent on an abundance of ozone precursors from outside of oil production.

As discussed in Chapter 3 of this volume, methane emissions, which are currently assessed as greenhouse gases, can be used as a predictor of many VOC emissions. Some VOCs are directly health damaging (e.g., benzene), and many others are precursors to regional tropospheric ozone. In a field-based study in the San Joaquin Valley of California, Gentner et al. (2014) found that at least 22% of all anthropogenic VOC emissions are attributable to oil development.

Local human exposures to emissions from oil and gas development have not been well-characterized, but modeling and preliminary studies have indicated that intermittent spikes in emissions to the atmosphere may pose increased risks to local human populations through air pollution concentrations at the regional scale (Brown et al., 2014; Colborn et al., 2014). Few studies to date have investigated the frequency and magnitude of air pollution emission spikes from oil and gas development, but available studies document their occurrence and their potential frequency and magnitude (Allen et al., 2013; Macey et al., 2014; Helmig et al. 2014).

6.5.2.2. Emissions and Potential Exposures from Equipment and Infrastructure

Oil and gas development relies on a variety of ancillary infrastructure throughout the well stimulation and oil and gas production process. This equipment includes, but is not limited to, diesel-powered trucks, generators, and pumps, separator tanks, condensate tanks, pipelines, flaring/venting operations, and gas compressor stations. The deployment and use of each of these pieces of equipment act as emissions sources that can present risks through exposure to chemicals, air emissions, and physical stressors. Specific to well stimulation operations is the need for heavy truck traffic to transport water, proppant, chemicals, and equipment to and from the well pad. Well stimulation as practiced in California typically requires about a hundred to two hundred heavy truck trips per vertical well, and two hundred to four hundred trips per horizontal well, counting two trips for each truck traveling to the site. This is one-third to three-quarters of the heavy truck traffic required for well pad construction and drilling.

The pollutants of primary health concern identified in the scientific literature and attributable to transportation and other heavy machinery associated with well stimulation are emissions of dust, diesel particular matter (dPM), nitrogen oxides (NO_x), sulfur dioxide and secondary sulfate particles (SO_x), volatile organic compounds (VOCs), and secondarily tropospheric ozone (Roy et al., 2013; Kemball-Cook et al., 2010). A pollutant of primary health concern emitted from the transportation component of shale gas development is dPM with aerodynamic diameter less than 2.5 microns ($PM_{2.5}$). dPM is a California TAC and a well-studied health-damaging pollutant that contributes to cardiovascular illnesses, respiratory diseases (e.g., lung cancer) (Garshick et al., 2008), atherosclerosis, and premature death (Pope, 2002; Pope et al., 2004). A study by the California Air Resources Board indicates that for each 10 $\mu g/m^3$ increase in $PM_{2.5}$ exposure in California, there is an expected 10% (uncertainty interval: 3%, 20%) increase in the number of premature deaths (Tran et al., 2008). Particulate matter can also contain concentrated associated products of incomplete combustion (PICs), and when particle diameter is < 2.5 μm, they can act as a delivery system of these compounds to the alveoli of the human lung (Smith et al., 2009). In addition to dPM, NO_x and VOCs, other pollutants prevalent in diesel emissions react in the presence of sunlight and high day-time temperatures to produce tropospheric (ground-level) ozone. Tropospheric ozone is a well-established respiratory irritant associated with increased respiratory and cardiovascular morbidity and mortality (Jerrett et al., 2009). It should be noted that most of the places where well stimulation is known to take place in California—The San Joaquin Valley and the Los Angeles Basin—are also the regions that are consistently out of attainment for atmospheric concentrations of tropospheric ozone. As such, oil and gas developments in these regions are a potentially significant factor (Gentner et al., 2013) of cumulative environmental public health risks for populations in these areas.

Formaldehyde is a volatile compound with well-established health impacts that is produced all along the oil and gas production chain. Notably, it is formed by incomplete combustion emitted by natural gas-fired reciprocating engines at oil and gas compressor stations, as well as being a component of diesel combustion. It is a suspected human carcinogen, but it has also been associated with acute and chronic health effects (U.S. EPA, 2013). One community-based exploratory monitoring study determined that levels of formaldehyde exceeded health-based risk levels near compressor stations with gas developed from wells enabled by hydraulic fracturing in Arkansas, Pennsylvania, and Wyoming oil/gas production sites (Macey et al., 2014). It should be noted that formaldehyde is not added to stimulation fluids, but rather is a product of combustion associated with oil and gas development activity, including well stimulation activity.

6.5.3. Public Health Studies of Toxic Air Contaminants

Oil and gas development—including that enabled by well stimulation—creates the risk of exposing human populations to a broad range of toxic air contaminants (TACs). Data suggest that these TACs are likely more elevated close to compared to far from active oil and gas development, and that emissions of TACs in areas of high population density (e.g., the Los Angeles Basin) result in larger population exposures than when population density is lower (See Chapter 3 of this Volume for more details).

Many of the constituents used in and emitted by oil and gas development are known to be damaging to health, and place disproportionate risks on sensitive populations, including children, the elderly, those with pre-existing respiratory and cardiovascular conditions, and those exposed to multiple environmental stressors. Oil and gas development poses more elevated population health risks when conducted in areas of high population density, such as the Los Angeles Basin, because it results in larger population exposures to TACs (see Los Angeles Basin Case Study in Volume III for more details).

California has large developed oil reserves located in densely populated areas. For example, the Los Angeles Basin has the highest concentrations of oil in the world, but Los Angeles is also a global megacity, and oil and gas development occurs in close proximity to human populations. In the San Joaquin Valley, there are a number of communities that live, work, and play near oil and gas development. Approximately half a million people live within one mile of a stimulated well, and many more live near oil and gas development of any type. In addition, large numbers schools, elderly facilities, and daycare facilities are sited within a mile of a stimulated well. The closer citizens are to these industrial facilities, the more potentially elevated their exposure to TACs. Volume II, Chapter 3 indicates that stationary source oil and gas facilities in the San Joaquin Valley are responsible for over 70% of H2S emissions, and 2-5.5% of benzene, formaldehyde, hexane, and xylene emissions. In the South Coast region, stationary oil and gas sources are responsible for less than 0.25% of all ten indicator TACs studied. While these fractions are in many cases not large as a fraction of regional impacts, they can still have important health impacts on nearby populations.

Studies from out of state indicate that community public health risks of exposures to toxic air contaminants, such as benzene and aliphatic hydrocarbons, are most significant within 800 meters (½ mile) from active oil and gas development (McKenzie et al., 2012). Atmospheric data on dilution of conserved TACs indicate that potentially harmful community exposures can occur out to ~3 km (almost 2 miles) from the source. There are no studies from inside California that have measured the relationship between health impacts and the distance from active oil and gas development. The Los Angeles County Department of Public Health conducted a peer-reviewed public health outcome study near the Inglewood Oil Field in Los Angeles County (Rangan and Tayour, 2011). This study did not find any health effects in populations relative to proximity to oil and gas development. However, the study was not designed to see long-term outcomes with incidence rates below ~ 1%. Therefore, significant questions remain about the health effects of proximity to oil and gas production that should be the subject of further study.

6.5.3.1. Methods for Peer Review of Scientific Literature

We conducted a review of the peer-reviewed scientific literature on the environmental public health and occupational health dimensions of well stimulation. In contrast to the bottom-up approach based on moving from hazard to exposure to outcome, most of the public health-relevant literature focuses on known links between population health risks and environmental pollution that arises from the well-stimulation-enabled oil and gas development. The best information for evaluation of the public health and occupational health impacts of oil and gas development, including that enabled by well stimulation in California, should be from verified California-specific datasets and peer-reviewed scientific studies conducted in California. However, we found California-specific information on public health risks to be extremely limited in quantity, quality, and scope. As a result, we also assessed the relevance of environmental public health-relevant studies from outside of California.

We included papers that consider the question of public health in the broad context of shale gas development. Of course, research findings in other categories such as air quality and water quality are relevant to public health, but in this subsection we only include those studies that directly consider the health of individuals and human populations. We only consider research to be original if it measures health outcomes or complaints (i.e., not health research that only attempts to determine opinion or methods for future research agendas).

We organized this literature review in a framework that tracks pathways from community health to various well stimulation types, in order to investigate what is known about any associations between sources of environmental pollution, potential exposures, and human health hazards related to well stimulation. We restricted the boundaries of our literature review to upstream oil and gas development processes prior to hydrocarbons being sent to market. We also only included physical health outcomes. Although some of the literature suggests that social, psychological, and economic impacts of well stimulation are possibly important for community health, these studies are beyond the scope of this review.

The source-to-outcome pathway is commonly used to describe associations between pollutant sources and health effects. This approach addresses in sequence the emissions, environmental concentrations of pollutants, pollutant exposure pathways (ambient air, water, etc.), and dose (e.g., micrograms of pollutant ingested, inhaled or absorbed per unit body weight per day) (Figure 6.5-1) (ATSDR, 2005). Potential sources of health-relevant environmental pollution are present throughout the well stimulation and oil and gas production process. Sources of environmental pollution include hydrocarbon production and processing activities (e.g., drilling, well stimulation, hydrocarbon processing and production, and wastewater disposal) and the transportation of water, sand, chemicals, and wastewater before, during, and after well stimulation (Shonkoff et al., 2014).

As noted above, the best information for evaluation of the public health and occupational health impacts of oil and gas development, including that enabled by well stimulation in California, should be from verified California-specific datasets and peer-reviewed scientific studies. However, we found this California-specific information to be limited in quantity, quality, and scope. With the exception of the Inglewood study (Rangan and Tayour, 2011), which had limited scope and statistical power, there have been no comprehensive health outcome studies that focus directly on the health impacts of stimulated wells. As a result, we also assessed the relevance of environmental public-health studies and experience from outside of California. Since 2007, the rapid growth of hydrocarbon development in shale and other low-permeability (aka, "tight") formations across the U.S. has been accompanied by an increase in scientific investigations of the environmental and public health dimensions of oil and gas development, including that enabled by well stimulation, especially hydraulic fracturing. For example, approximately 70% of the peer-reviewed journal papers that are pertinent to the public health dimensions of onshore well-stimulation-enabled oil and gas development have been published between January 2009 and December 2014 (PSE Healthy Energy, 2014)[2]. This body of literature is still relatively new; many uncertainties and data gaps on the human health impacts persist on the national scale, and especially with application to California.

2. For a near-exhaustive collection of peer-reviewed scientific literature on the subject of shale gas and well-stimulation-enabled oil and gas development please see the PSE Healthy Energy Peer Reviewed Literature Database at http://psehealthyenergy.org/site/view/1180.

Some studies of well stimulation in other parts of the country, including Pennsylvania, Colorado, Utah, North Dakota, and Texas, may be relevant to California. There are notable differences between direct and indirect impacts of oil and gas development practices in California compared to those in other states, due to differences in geology, variability and tectonics, well-stimulation and drilling techniques, and oil production and transmission infrastructure, such as pipelines to transport fresh water, recovered fluids, and produced water (see Volume I).

However, in many cases, there are similarities between the *types* of hazards noted in other states and those in California, although the magnitude of risks associated with these hazards are not clear. For example, studies of oil and gas development with relevance to public health in Colorado, Utah, and Wyoming assess oil and gas development at the regional scale (Pétron et al., 2012; Pétron et al., 2014; Darrah et al., 2014; Thompson et al., 2014; Helmig et al. 2014) in the context of shale and source rock formations, but also of hydraulic-fracturing-enabled migrated oil development, much like the majority of production in California.

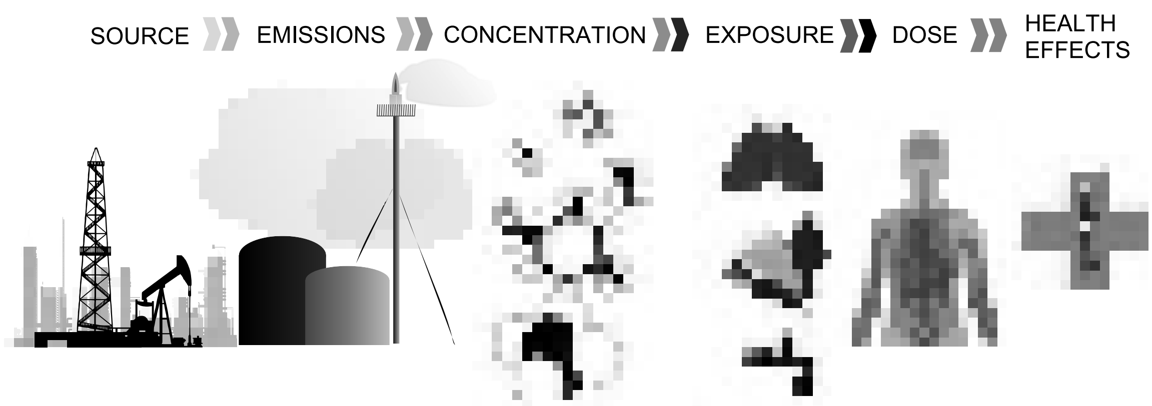

Figure 6.5-1. Simplified environmental exposure framework. Source: Shonkoff et al. (2014).

6.5.3.2. Results from the Environmental Public Health Literature Review

We divide the results for our literature review into three sections. The first section provides an overview of the peer-reviewed literature on well-stimulation-enabled shale and tight gas, and discusses the relevance of the current literature to well-stimulation-enabled oil and gas development in California. While the development of tight-gas resources is not a perfect proxy for the resources developed by means of well stimulation in California, the peer-reviewed literature between 1 January 2009 and 31 December 2014 (the time range we accessed) has a strong focus on tight-gas resources and provides useful but not necessarily relevant insight. We note, however, that there are fundamental differences between the production of tight gas and what is going on in California. Many of the volatile organic compounds found in tight gas are also produced from and emitted by California oil and gas development, but the relative concentrations of these compounds between different types of oil and gas development can differ widely, based on geology, geography, and hydrocarbon type. In the second section, we review epidemiologic and population health studies, and identify what these studies tell us about any potential impacts on public health. The third section examines what the wider literature says about health issues due to potential exposures to water and air emissions from well-stimulation-enabled oil and gas development.

6.5.3.3. Public Health Outcome Studies

Within California, we could only identify one public health outcome study that has relevance to well-stimulation-enabled oil production. This is the Inglewood study carried out by Los Angeles County (Rangan and Tayour, 2011), which is discussed below. Outside of California, health outcome studies and epidemiologic investigations continue to be particularly limited, and most of the peer-reviewed papers to date are commentaries and reviews of the environmental literature pertinent to environmental public health risks.

A cursory public health outcome study was conducted by the Los Angeles County Department of Public Health near the Inglewood Oil Field in Los Angeles County. This study compared incidence of a variety of health endpoints including all-cause mortality, low birth weight, birth defects, and all cancer among populations nearby the Inglewood Oil Field and Los Angeles County as a whole. The study found no statistically significant difference in these endpoints between the population near the Inglewood field and the overall county population. While this may seem to indicate that there is no health impact from oil and gas development, as the study notes, the epidemiological methods employed in this study do not allow it to pick up changes in "rare events" such as cancer and birth defects in small sample sizes, as is the case in this study (Rangan and Tayour, 2011). In addition, lacking statistical power, the Inglewood Oil Field Study is a cluster investigation with exposure assigned at the group level (i.e., an ecological study). It also appears that only crude incidence ratios were calculated. This type of study design is insufficient for establishing causality and has many major limitations, including exposure misclassification and confounding, which may have obscured associations between exposure to environmental stressors from oil and gas development and health outcomes.

Health assessments have been confounded by the dearth of well-designed human-population studies that measure both human exposure and impacts. While a number of studies have found environmental and exposure pathways and health-damaging compounds in environmental concentrations sufficiently elevated to induce health effects, epidemiological studies aimed to assess and quantify the population health burden (i.e., impact severity) of oil and gas production remain in their infancy.

In a study that analyzed air samples from locations in five different states using a community-based monitoring approach, it was found that levels for eight volatile chemicals, including benzene, formaldehyde, hexane, and hydrogen sulfide, exceeded federal guidelines (ATSDR minimal risk levels (MRLs) (ATSDR, 2014) and EPA Integrated Risk Information System (IRIS) cancer risk levels) in a number of instances (Macey et al., 2014). Notably, the residents who collected the grab samples reported a number of common health symptoms, including "headaches, dizziness or light-headedness, irritated, burning, or running nose, nausea, and sore or irritated throat" (Macey et al., 2014). We note that this was not a formal outcomes-based study, and the authors did not attempt to associate the reported health effects with the chemicals measured in the samples. But the study suggests that concentrations of hazardous air pollutants near well-stimulation-enabled oil and gas operations can be elevated to levels where health impacts could occur. We further note that such elevated levels may not be due to well stimulation itself, but to existing petroleum production combined with enhanced petroleum production.

There have been health complaints associated with oil and gas development documented in the peer-reviewed literature. These studies have limitations because they are mainly provide self-reported outcomes and are based on convenience samples, which are collected for other purposes or easily collected by or from local populations. However, many of the reported health outcomes are consistent with what would be expected from exposure to some of the known contaminants associated with oil and gas development, and are consistent across geographic space. In a 2012 survey of Pennsylvania citizens, more than half of the participants surveyed who live in close proximity to well-stimulation-enabled oil and gas development reported increased fatigue, nasal irritation, throat irritation, sinus problems, burning eyes, shortness of breath, joint pain, feeling weak and tired, severe headaches, and sleep disturbance (Steinzor et al., 2013). The survey also found that the number of reported health problems decreased with distance from facilities.

Some research has attempted to assess human-health risks related to air pollutant emissions associated with hydraulic-fracturing-enabled oil and natural gas development. Using U.S. EPA guidance to estimate chronic and subchronic non-cancer hazard indices (HIs) as well as excess lifetime cancer risks, a study in Colorado suggested that those living in closer geographical proximity to active oil and gas wells (≤ 0.8 km [0.5 mile]) were at an increased risk of acute and sub-chronic respiratory, neurological, and reproductive health effects, driven primarily by exposure to trimethyl-benzenes, xylenes, and aliphatic hydrocarbons. It also suggested that slightly elevated excess lifetime cancer risk estimates were driven by exposure to benzene and aliphatic hydrocarbons (McKenzie et al., 2012). The findings of this study are corroborated with atmospheric dilution data of conserved pollutants; for instance, a U.S. EPA report on dilution of conserved toxic air contaminants indicates that the dilution at 800 m (0.5 mile) is on the order of 0.1 mg/m^3 per g/s (U.S. EPA, 1992). Going out to 2,000 m increases this dilution to 0.015 mg/m^3 per g/s, and going out to 3,000 m increases dilution to 0.007 mg/m^3 per g/s. Given that, for benzene, there is increased risk at a dilution of 0.1, it is not clear that concentrations out to 2,000 m (1.25 miles) and 3,000 m (1.86 miles) can necessarily be considered as presenting acceptable risk. However, beyond 3,000 m (1.86 miles), where concentrations fall more than two orders of magnitude via dilution relative to the ½ mile radius, there is likely to be a sufficient margin of safety. Nevertheless, these results indicated that any potentially harmful community exposures could occur at 2,000 meters (1.25 miles) and as much as almost ~3,000 meters (~2 miles) from the source. In considering these dilution assessments, we note that—based on wind, topography, and inversion layers--dilution can increase or decrease, and that increasing density of oil and gas development will require greater dilution to attain the same level of risk as lower density.

In contrast, an oil and gas industry study in Texas compared VOC concentration data from seven air monitors at six locations in the Barnett Shale with federal and state health-based air concentration values (HBACVs) to determine possible acute and chronic health effects (Bunch et al., 2014). The study found that shale gas activities did not result in community-wide exposures to concentrations of VOCs at levels that would pose a health concern. The key distinction between McKenzie et al. (2012) and Bunch et al. (2014) is that Bunch et al. (2014) used air quality data generated from monitors focused on regional atmospheric concentrations of pollutants in Texas, while McKenzie et al. (2012) included samples at the community level. Finer geographically scaled samples can often capture local atmospheric concentrations that are more relevant to human exposure (Shonkoff et al., 2014).

This geographical correlation has been observed in random sampling efforts as well. In a recent study in Pennsylvania, researchers evaluated the relationship between household proximity to natural gas wells and reported health symptoms for 492 people in 180 randomly selected homes with ground-fed wells in an area of active drilling (Rabinowitz et al., 2014). The results suggest that close proximity to gas development is associated with prevalence of dermal and respiratory health symptoms.

In addition to population health hazards in varying distances from active oil and gas development, other studies have assessed the effect of the *density* of oil and gas development on health outcomes. In a retrospective cohort study in Colorado, McKenzie et al. (2014) examined associations between maternal residential location and density of oil and gas development. The researchers found a positive dose-response association between the prevalence of some adverse birth outcomes, including congenital heart defects and possibly neural tube defects and increasing density of development (McKenzie et al., 2014). For instance, the observed risk of congenital heart defects in neonates was 30% (OR = 1.3 (95% CI: 1.2, 1.5)) greater among those born to mothers who lived in the highest density of oil and gas development (> 125 wells per mile), compared to those neonates born to mothers who lived with no oil and gas wells within a 16 km (10-mile) radius. Similarly, the data suggest that neonates born to mothers in the highest density of oil and gas development were twice as likely (OR = 2.0, 95% CI: 1.0, 3.9) to be born with neural tube defects than those born to mothers living with no wells in a 10-mile radius (McKenzie et al., 2014). The study, however, showed no positive association between the density and proximity of wells and maternal residence for oral clefts, preterm birth, or term low birth weight. We also note that these indirect effects, by definition, cannot be directly linked to stimulation technology, but to existing and well-stimulation-enhanced petroleum production.

6.5.4. Summary of Public Health Outcome Studies

There have been few epidemiological studies that measure health effects associated with oil and gas development, whether enabled by well stimulation or not. The studies that have been published have been heavily focused on exposures to toxic air contaminants (hazardous air pollutants), while fewer studies have evaluated associations between oil and gas development and water contamination.

Each of the studies discussed above have limitations to their study designs, their geographic focus, and their statistical power to evaluate associations. These studies suggests that health concerns about oil and gas development may not be **direct** effects specific to the well stimulation process, but rather are associated with **indirect** effects of oil and gas development. For example, the studies in Colorado (McKenzie et al., 2012; McKenzie et al., 2014) found that the most likely driver of poor health outcomes were aliphatic hydrocarbons and benzene. Neither of these compounds is added to stimulation fluids, but rather are mobilized in the subsurface and co-produced (and co-emitted) with oil and gas production, processing, transmission, and consumption.

6.6. Occupational Health-Hazard Assessment Studies

Due to their proximity to hazards, workers directly involved in well stimulation processes may have exposure to chemical and physical hazards larger than those of the surrounding communities, and therefore have the greatest likelihood of any resulting acute and/or chronic health effects. The expansion of well stimulation in California has the potential to expose workers in this industry to a range of existing hazards related to oil and gas development, and additional hazards specific to well stimulation such as elevated VOC exposures during injection and flowback operations (Esswein et al., 2014) and the use of proppant, which has been noted to subject workers to elevated silica exposure (Esswein et al., 2013). Silica exposure is a major risk factor for the development of the lung disease silicosis.

An adequate understanding of occupational health hazards requires information about the quantities and composition of materials used, handling protocols, and emissions factors of operations in addition to information about the tasks, protocols, and exposure reduction control measures for activity on well pads, in and around trucks and machinery, and in other locations throughout the oil development process related to well stimulation. Employers can and often do implement comprehensive worker protection programs that substantially reduce worker exposure and likelihood of illness and injury. Employers in the oil and gas industry are required to comply with existing California occupational safety and health regulations, and follow best practices to significantly reduce and/or eliminate illness and injury risk to their employees (California Occupational Safety and Health Act of 1973 and Title 8 of the California Code of Regulations). In following these standards and best practices in protecting workers from chemical exposures while they are involved in well stimulation operations, employers in this industry may also reduce the likelihood of chemical exposure to the surrounding community.

There is a large California workforce engaged in the oil development and production industry. We reviewed available literature and the scope of this occupation group (and the hazards they face). Although data are available on health risks faced by this work population, little data is available on the hazards directly associated with well stimulation activities.

6.6.1. Scope of Industry and Workforce in California

Employment numbers and occupations involved in well stimulation are impossible to ascertain with precision, as companies engaged in drilling and support activities in well stimulation are also involved with overall oil and gas development in California. Any workers engaged in well stimulation are typically part of the broader oil and gas well development/production industry. This is an industry where workers can be exposed to a range of hazards in addition to those directly associated with well stimulation. Table 6.6-1 provides a summary of the employment in the oil and gas extraction industry in California.

Table 6.6-1. Employment in oil and gas extraction – California 2014.

Industry Title	Establishments	Average Monthly Employment
2111111 Crude Petroleum and natural gas extraction	179	9,669
2111112 Natural Gas Liquid Extraction	10	193
213111 Drilling Oil and Gas Wells	91	3,419
213112 Support Activities, Oil/Gas Operations	240	9,162
Total	**520**	**22,443**

Source: http://www.labormarketinfo.edd.ca.gov/

A review of all data on occupational health for the oil and gas extraction industry indicates that this industry has a high rate of worker injury and death relative to other industries, but does not collect publicly available data on the fraction of oil and gas development that is enabled by well stimulation (NIOSH, 2015a; 2015b; 2015c; 2015d). According to NIOSH (2015d), the oil and gas extraction industry had an annual occupational fatality rate of 27.5 per 100,000 workers (2003-2009)—more than seven times higher than the rate for all U.S. workers. The annual occupational fatality rate is highly variable, and correlates with the level of drilling activity. For example, the numbers of fatalities increased by 23% between 2011 and 2012 to the largest number of deaths of oil and gas workers since 2003. Appendix 6.D provides details on occupational health data we compiled for the U.S. oil and gas extraction industry. In the sections below, we summarize studies that address the direct impacts of well stimulation within the oil and gas industry. This is U.S. data, which is relevant to California operations, but not necessary fully representative of current or future California well stimulation activities.

6.6.2. Processes and Work Practices

In seeking insight on occupational hazards from well stimulation, we identified two review papers useful for describing occupational exposures in oil and gas development (Mulloy, 2013; Witter, 2014), but these papers do not include job or process descriptions. We identified two additional peer-reviewed papers describing the work processes in oil and gas extraction that evaluate occupational exposure for silica and VOCs attributable directly to well stimulation (Esswein et al., 2013; 2014). The Esswein et al. papers (2013; 2014) report results from the National Institute for Occupational Safety and Health study that collected 111 personal-breathing-zone samples at 11 sites in five states during four seasons, for investigation of crystalline silica exposure and personal and environmental measurements at six sites in two states, for investigation of chemical exposures. We found no other publicly available data sources that include job titles or work activities during oil and gas extraction or well stimulation.

In the first of these two papers, Esswein et al. (2013) describe the processes of hydraulic fracturing, in terms of the workers involved and their typical roles as:

> *At a typical site, 10 to 12 driver/operators position and set up equipment, configure and connect piping, pressure test, then operate the equipment (e.g., sand movers, blender, and chemical trucks) required for hydraulic fracturing. Other employees operate water tanks and water transport systems, and several control on-site traffic, including sand delivery trucks and other vehicles. An additional crew includes well liners (typically 3–5) who configure and assemble well casing perforation tools and operate cranes to move tools and equipment into and out of the well. … Moving proppant along transfer belts, pneumatically filling and operating sand movers, involves displacement of hundreds of thousands of pounds of sand per stage, which creates airborne dusts at the work site (Esswein et al., 2013).*

Similarly, in the second paper, Esswein et al. (2014) describe flowback operations and the associated exposures to VOCs from these operations as:

> *Typical flowback operations have two to four flowback personnel performing flowback tasks; these were the typical number of workers at each of the sites visited. Air sampling, typically collected over two days, included workers with the following job titles and descriptions:*

- *Flowback lead: recorded well pressures and temperatures, monitored separators and other equipment*
- *Flowback tech: gauged flowback tanks 1–4 times per hr., recorded volumes, assisted in tank pumping and fluid transfers to trucks*
- *Production watch lead: monitored rate and volume of natural gas and liquid hydrocarbons*
- *Production watch technician: gauged production tanks*
- *Water management operator: gauged water tanks, ran pumps*

Workers access the tanks through hatches located on the tops of tanks. Periodically, recovered liquid hydrocarbons/condensate is pumped to production tanks or to trucks, which collect and transport process fluids off the well pad; natural gas is typically piped to gas gathering operations. Tank gauging and other tasks required during flowback can present exposure risks for workers from alkane and aromatic hydrocarbons produced by the well and diluted treatment chemicals used during hydraulic fracturing (typically a combination of acid, pH adjusters, surfactant, biocides, scale and corrosion inhibitors, and, in some cases, gels, gel demulsifiers, and cross-linking agents) (Esswein et al., 2014).

6.6.3. Acid Used in Oil and Gas Wells

The oil and gas industry commonly uses strong acids along with other toxic substances, such as corrosion inhibitors, for both routine maintenance and well stimulation (see Volume I, Chapter 2 and 3 & Volume I). These acids pose occupational hazards relevant to well stimulation. Well acidizing requires the use of hydrochloric (HCl) and hydrofluoric (HF) acid. In many cases, HF is created at the oilfield by mixing hydrochloric acid with ammonium fluoride and immediately injecting the mix down the well (Collier, 2013). Creating the HF on site may be safer than offsite production, because it reduces the risk of transport accidents. In all uses of HF, there is the potential for worker exposure to acid gases. According to industry protocols, safety precautions for those on site during an acid treatment concern detection of leaks and proper handling of acid (SPE, 2015; API, 1985). As also reported in Volume II Chapter 2, due to the absence of state-wide mandatory reporting on chemical use in the oil and gas industry, it is not known how much acid is used for oil and gas development throughout California.

Well-established procedures exist for mixing and handling acids (NACE, 2007). The parent acids do not generally migrate long distances from the well, but acids formed through a complex series of reactions during acidization can migrate deeper into the formation (Weidner, 2011). If the acidization fluids are introduced into the well in the right proportions and order, and sufficient time and conditions allowed for reactions to proceed, then the original acids are used up during the acidization process (Shuchart, 1995). The reaction of strong acids with the rock minerals, corrosion products, petroleum, and other injected chemicals can also release contaminants of concern, such as hydrogen sulfide from acid reaction with iron sulfides, that have not been characterized or quantified. These chemicals may be present in recovered fluids and produced water (NACE, 2007). We do not have data to determine how much strong acid, including hydrochloric and hydrofluoric acid, is used in oil and gas development in California. DOGGR has only recently required reporting of all acid use that will result in a better understanding in the future. Hydraulic fracturing operations have only infrequently incorporated acid use (11 voluntarily reported applications between January 2011 and May 2014). Industry has voluntarily reported approximately twenty matrix-acidizing treatments per month throughout California, but has not revealed detailed chemical information. The South Coast Air Quality District requires reporting on the use of all chemicals by the oil and gas industry. Their data suggest widespread and common use of acid for many applications in the industry.

Environmental public health exposures to strong acids are only likely to occur at the surface, given that migration of acids in the subsurface are limited by relatively rapid reactions. The most likely human exposures to strong acids are to workers. The opportunities for exposure are predominantly the following: (1) handling and mixing of acids prior to well injection, (2) during flowback following an acid treatment, and (3) during accidents and spills.

State and federal agencies regulate spills of acids and other hazardous chemicals, and existing industry standards dictate standard safety protocols for handling acids (see Section 6.6.3.4). The Office of Emergency Services (OES) between January 2009 and December 2014 reported nine spills of acid that can be attributed to oil and gas development in California. Reports indicate the spills did not involve any injuries or deaths. These acid spill reports represents less than 1% of all reported spills of any kind attributed to the oil and gas development sector in the same period, and suggest that spills of acid associated with oil and gas development are infrequent. Given the lack of Occupational Safety and Health Administration (OSHA) reporting of worker exposures to acids, to the extent that this reporting is comprehensive, it appears that industry protocols for handling acids likely are protecting workers from such acute exposures.

Chapter 2 of this volume reports chemical spills in California oil fields, including spills of hydrochloric, hydrofluoric, and sulfuric acids. Of the 31 spills reported between January 2009 and December 2014, nine were acid spills. Among these was a storage tank at a soft water treatment plant containing 20 m^3(5,500 gallons) of hydrochloric acid in the Midway-Sunset Oil Field in Kern County that ruptured violently, releasing the acid beyond a secondary containment wall. No injuries or deaths were associated with this or any other acid spill.

Work processes and health hazards associated with well stimulation are summarized in Table 6.6-2.

The physical hazard associated with a chemical used on the job is most often characterized by evaluating a standard selection of properties associated with the individual chemical or chemical mixture. These properties include inflammability, corrosivity, and reactivity.

There are a number of different systems for classifying the hazardous properties of chemicals. The American Coatings Association, Inc. developed the Hazardous Materials Identification System (HMIS) (ACS, 2015) to aid its members in the implementation of an effective Hazard Communication Program as required by law. Another system developed by the National Fire Protection Association (NFPA) is directed at communicating potential hazards during emergency situations (NFPA, 2013.) Both systems have a "0 to 4" ranking system with a chemical ranked "4" having a severe hazard, "3" representing a serious hazard, "2" representing a moderate hazard, and "1" a slight hazard. Materials ranked "0" are of minimal or no hazard for the category ranked.

All of the chemicals reportedly in well stimulation in California (see Chapter 2, Appendix 2.A, Tables 2.A-3 and 2.A-5) were evaluated for this report using both the HMIS and the NFPA systems. Approximately 20% to 30% of the additives were not categorized under either the HMIS or NFPA systems for different hazards. Overall, only approximately 5% of the well stimulation fluid additives were considered flammable or fire hazard, and only a few compounds were ranked as physical or reactivity hazards (Figure 6.6-1).

Well stimulation fluid additives categorized as severe (4) or serious hazards (3) are listed in Chapter 2, Appendix 2.A, Table 2.A-8 (Chapter 2). Since chemical hazards and fire hazards are integral to both conventional and unconventional oil and gas extraction, the well stimulation additives illustrated in Figure 6.6-1 are not likely to pose new or unusual hazards that are specific to unconventional oil and gas production. However, the additives should be considered in evaluation of occupational exposure and in assessment of the risks associated with oil and gas production.

Table 6.6-2. Work processes and health hazards associated with well stimulation.

Work processes	Health hazards	Fed OSHA Standards
Mixing and injecting of chemicals and dusts - i.e., proppants, acids, pH adjustment agents, biocides etc.	Irritation and burns to skin and eyes Acute and chronic respiratory disease (COPD, asthma, silicosis, lung cancer) Low pH recovered fluid	Hazard Communication, Safety Data Sheets - 29 CFR 1910.1200(g) Personal Protective Equipment - 29 CFR Subpart I Specifications for Accident Prevention Signs and Tags -29 CFR 1910.145 Toxic and Hazardous Substances - 29 CFR 1910 Subpart Z Hazard Communication - 29 CFR 1910.1200 Emergency Response Program to Hazardous Substance Releases - 29 CFR 1910.120(q) Medical Services and First Aid - 29 CFR 1910.151(c)
Pressure pumping	Explosions Acute and chronic inhalation exposure due to high pressure from uncontrolled releases, use of flammable fluids, gases, and materials	Personal Protective Equipment, General Requirements - 29 CFR 1910.132
Recovered fluids	Explosions Acute and chronic inhalation exposure due to high pressure from uncontrolled releases, use of flammable fluids, gases and materials	Personal Protective Equipment - 29 CFR 1910 Subpart I Portable Fire Extinguishers - 29 CFR 1910.157 Welding, Cutting, and Brazing - 29 CFR Subpart Q, 29 CFR 1910.252, General Requirements
Multiple operations: hydrogen sulfide, volatile organic compounds (VOCs), combustion products and elevated noise	Asphyxia Nervous system, liver and kidney damage Cancer (blood)	Respiratory Protection, General Requirements - 29 CFR 1910.134(d)(iii) Air contaminants - 29 CFR 1910.1000
Transport, Rig-Up, and Rig-Down	Injuries and fatalities (struck-by, caught-in, crushing hazards, and musculoskeletal injuries) from off-site and on-site vehicle and machinery traffic or movement; heavy equipment, mechanical material handling, manual lifting, and ergonomic hazards (these are mostly indirect hazards with respect to well stimulation)	Electrical - 29 CFR 1910.307 – Hazardous (Classified) Locations Powered Industrial Trucks - 29 CFR 1910.178 Crawler, Locomotive, and Truck Cranes - 29 CFR 1910.180 Slings - 29 CFR 1910.184(c)(9) Walking-Working Surfaces - 29 CFR 1910 Subpart D Permit-Required Confined Spaces - 29 CFR 1910.146 Occupational Noise Exposure - 29 CFR 1910.95 Electrical: Selection and Use of Work Practices - 29 CFR 1910.33

Source: Adapted from U.S. OSHA (2014) and Esswein et al. (2013; 2014)

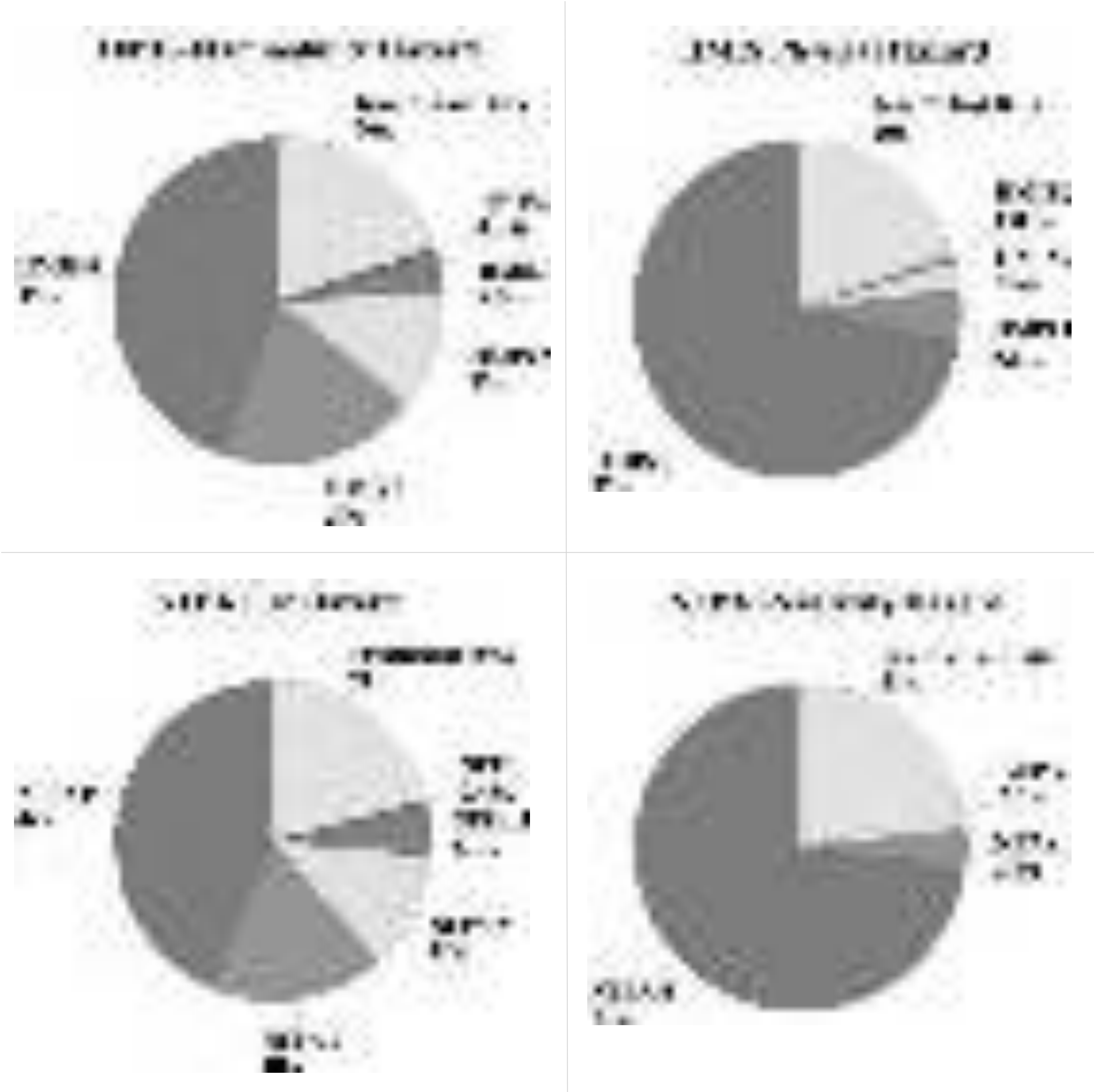

Figure 6.6-1. Evaluation of the flammability, reactivity, and physical hazards of chemical additives reported for hydraulic fracturing in California using the Hazardous Materials Identification System (HMIS) and the National Fire Protection Association (NFPA) classification system.

6.6.3.1. Occupational Health Outcomes Associated With Well Stimulation-Enabled Oil and Gas Development

There are few peer-reviewed health outcomes studies among workers in the oil and gas development industry that are specific to well-stimulation-enabled oil and gas development. For well stimulation, there are effectively no health outcome studies and only two studies addressing health risks (Esswein et al., 2013; 2014). The results of these two studies are summarized above.

6.6.3.2. Worker Protection Standards, Enforcement, and Guidelines for Well Stimulation Activities

The U.S. Occupational Safety and Health Administration (OSHA) has identified multiple hazards and enforces numerous standards for oil and gas extraction (OSHA, 2015a; 2015b). There are several specific OSHA exemptions for the oil and gas development industry, including:

- Process safety management (PSM) of highly hazardous and explosive chemicals (29 CFR 1910.119). The PSM standard requires affected facilities to implement a systematic program to identify, evaluate, prevent, and respond to releases of hazardous chemicals in the workplace. The PSM standard exempts oil and gas well drilling and servicing operations (OSHA, 2015c)

- Comprehensive General Industry Benzene Standard (29 CFR 1910.1028). Under the Comprehensive Standard, the limit for workers' exposure is 1 part per million (ppm)—the occupational exposure limit is the same. The exemption allows worker exposures up to 10 ppm in oil and gas. The exemption also eliminates requirements for medical monitoring, exposure assessments, and training (OSHA, 2015d).

- Hearing Conservation Standard (29 CFR 1910.95). This standard, designed to protect general industry employees, establishes permissible noise exposure limits and outlines requirements for controls, hearing protection, training, and annual audiograms for workers. Many sections of the standard do not apply to employers engaged in oil and gas well drilling and servicing operations (OSHA, 2015e).

- Control of Hazardous Energy Sources, or "Lockout/Tagout" (29 CFR 1910.147). The standard requires specific practices and procedures to safeguard employees from the unexpected energization or startup of machinery and equipment, or the release of hazardous energy during service or maintenance activities. The standard does not cover the oil and gas well drilling and servicing industry (OSHA, 2015f).

The U.S. OSHA has issued an alert on the hazards of silica exposure (OSHA, 2015g) and guidance to employers on other safety and health hazards during hydraulic fracturing and fluid recovery (OSHA, 2015h). The National Institute for Occupational Safety and

Health (NIOSH) has identified exposure to silica dust and volatile organic compounds as significant health hazards during oil and gas extraction (NIOSH, 2015a; 2015b; 2015c), and recommends additional quantification of exposure to diesel particulate and exhaust gases from equipment, high or low temperature extremes, noise, hydrocarbons, hydrogen sulfide, heavy metal exposure, and naturally occurring radioactive material (NIOSH, 2015d).

The California Division of Occupational Safety and Health (CalOSHA) has specific enforceable regulations pertaining to petroleum drilling and production (CalOSHA, 2015a; 2015b). For the ten-year period January 1, 2004–December 31, 2013, there were 281 inspections in oil and gas extraction: 77 inspections in NAICS 211, 98 inspections in NAICS 213111, and 106 inspections in NAICS 213112 (OSHA, 2015i). Of the 281 inspections, 153 (54%) were in response to an accident, 47 (17%) were planned, and 36 (13%) were due to complaints. Cal/OSHA is required to investigate all work-related amputations, hospitalizations for greater than 24 hours, and traumatic fatalities. There are 104 cases in which a detailed narrative is available regarding these incidents, including 16 work-related fatalities (Appendix 6.E).

The American Petroleum Institute has also published comprehensive safety and health guidelines for oil and gas well drilling and servicing operations, and includes recommended best practices from the American Conference of Governmental Industrial Hygienists and American National Standards Institute (API, 2007).

The American Petroleum Institute (API) and the Society of Petroleum Engineers have established protocols and safety precautions for those on site during an acid treatment (SPE, 2015; API, 1985). These guidelines state that (a) pressure tests with water or brine are used to ensure the absence of leaks in pressure piping, tubing, and packer; (b) anyone around acid tanks or pressure connections should wear safety goggles for eye protection; (c) those handling chemicals and valves should wear protective gauntlet-type, acid-resistant gloves; (d) water and spray washing equipment should be available at the job site; (e) when potential hydrogen sulfide gas hazards exist, workers need contained, full-face, fresh-air masks; (f) testing equipment and appropriate safety equipment should be on hand to monitor the working area and protect personnel in the area; and (g) special scrubbing equipment may be required for removal of toxic gases.

6.7. Other Hazards

Oil and gas development, including those enabled by well stimulation, creates a number of physical stressors, including noise and light pollution. Although noise pollution and light pollution are often thought of as mere nuisances, data suggest that these physical stressors can be detrimental to human health. Noise pollution is associated with truck traffic, drilling, pumps, flaring of gases, and other processes associated with well stimulation-enabled oil and gas development and oil and gas development in general.

6.7.1. Noise Pollution

While no peer-reviewed studies to date examine the public health implications of communities exposed to elevated noise from oil and gas development in California, numerous large-scale epidemiological studies have found positive associations between elevated environmental noise and adverse health outcomes. (See Noise Literature Review in Appendix 6.F.) Noise is a biological stressor that modifies the function of the human organs and nervous systems, and can contribute to the development and aggravation of medical conditions related to stress, most notably hypertension and cardiovascular diseases (Munzel et al., 2014). The World Health Organization (WHO, 2014) has noise thresholds, measured in decibels (dB), and their effect on population health, with noise levels above 55 dB considered dangerous for the general population (Table 6.7-1). A number of activities associated with drilling and production activity (Table 6.7-2), some of which could also be associated with well stimulation, generate noise levels greater than those considered dangerous to public health. Dose-response data indicate that noise during well stimulation in California and elsewhere is associated with sleep disturbance and cardiovascular disease (McCawley, 2013). These findings are corroborated by estimates from the New York State Department of Environmental Conservation on the development of shale gas (NYSDEC, 2011).

Table 6.7-1. WHO thresholds levels for effects of night noise on population health.

Average night noise level over a year $L_{night,outside}$	**Health effects observed in the population**
Up to 30 dB	Although individual sensitivities and circumstances may differ, it appears that up to this level no substantial biological effects are observed. $L_{night,outside}$ of 30 dB is equivalent to the no-observed-effect level (NOEL) for night noise.
30 to 40 dB	A number of effects on sleep are observed from this range: body movements, awakening, self-reported sleep disturbance, and arousals. The intensity of the effect depends on the nature of the source and the number of events. Vulnerable groups (for example children, the chronically ill and the elderly) are more susceptible. However, even in the worst cases the effects seem modest. $L_{night,outside}$ of 40 dB is equivalent to the lowest-observed-adverse-effect level (LOAEL) for night noise.
40 to 55 dB	Adverse health effects are observed among the exposed population. Many people have to adapt their lives to cope with the noise at night. Vulnerable groups are more severely affected.
Above 55 dB	The situation is considered increasingly dangerous for public health. Adverse health effects occur frequently, a sizeable proportion of the population is highly annoyed and sleep-disturbed. There is evidence that the risk of cardiovascular disease increases.

Source: Adapted from the WHO (2014)

Table 6.7-2. Equipment Noise Levels for Drilling and Production in Hermosa Beach, California.

Work Stage	Equipment	Sound Power Level† (dBA)
Drilling (30 month scheduled duration)	Hydraulic Power Unit	110.7
	Mud Pump	105.4
	Drill Rig	93.3
	Shaker	75.3
	Pipe Handling (Quiet Mode)	107.5
Production (at rate of 800 barrels per day)	Well Pumps	97.7
	Produced Oil Pump	77.7
	Produced Water Pump	86.7
	Shipping Pump	92.8
	Water Booster Pump	86.7
	Water Injection Pumps (2)	102.8
	Vapor Recovery Compressor	88.6
	Vapor Recovery Unit Cooler	90.2
	1st Stage Compressor (2)	96.2
	2nd Stage Compressor (2)	96.2
	Compressor Cooler	102.0
	Amine Cooler	102.1
	DEA Charge Pump	77.7
	Regenerator Reflux Pump	77.7
	Chiller	85.0
	Glycol Regenerator	92.4
	Micro-turbines (5)	92.9
	Variable Frequency Drives	83.3

Source: Adapted from Hermosa (2014) based on field measurements and identified as Source Noise Levels (measured in decibels (dBA)) used in modeling noise contour maps.

While noise mitigation measures are undertaken in some California oil fields, including Hermosa Beach (Hermosa, 2014) and Inglewood (Cardno ENTRIX, 2012), there are no data available as to their effectiveness and adherence. The City of Hermosa Beach allows noise levels in the 40-60 dB range (Appendix 6.F, Table 6.F-8a and Table 6.F-9).

6.7.2. Light Pollution

Light pollution is reported as a nuisance in communities undergoing well stimulation, because activities occur during both daytime and nighttime hours (Witter et al., 2013). While little research has been conducted on the public health implications of exposures to light pollution from oil and gas development, some epidemiologic studies of light pollution from other sources suggests a positive association between indoor artificial light and poor health outcomes (Chepesiuk, 2009). Further, other studies suggest that night-time light exposure can disrupt circadian and neuroendocrine physiology (Chepesiuk, 2009; Davis and Mirick, 2006). Hurley et al. (2014) found that women living in areas with high levels of artificial ambient light at night may be at an increased risk of breast cancer, although how these findings translate to the levels of night-time light exposure to oil and gas development remains understudied.

6.7.3. Biological Hazards

Coccidioides immitis (*C. immitis*) is a soil fungus that causes Valley Fever and is endemic to the soils of the southwest. The San Joaquin Valley is an area where the fungal spores live in the top 2”-12” of soil. Soil disturbance associated with developing and maintaining oil field infrastructure may generate airborne *C. immitis* and expose workers and nearby residents. Cases of Valley Fever are not uncommon among workers in the oil fields of Kern County (Hirshmann, 2007).

While over 60% of people exposed to *C. immitis* never have symptoms, symptomatic infection can result in those who are exposed to the spores through inhalation. Symptoms range from mild, influenza-like illness to systemic fungal infection and severe disease, particularly in those who are immune-compromised. Coccidioidomycosis is considered an occupational hazard in endemic regions, particularly for workers who are exposed to spores through earth-moving activities or who are exposed to dusty conditions (Friedlander, 2014). In California, Cal/OSHA issued a fact sheet to employers to outline the health hazards of Valley Fever and preventative measures, focusing on worker education, adopting site plans to reduce exposure, and protecting workers against exposure with NIOSH-approved respiratory protection filters (Friedlander, 2014).

While the health hazards of Valley Fever have been outlined, no data have been published on the rates of infection among workers specifically in the oil and gas industry in California. Valley Fever remains an important occupational health hazard, as much of the well-stimulation-enabled oil and gas extraction activities take place in California’s Central Valley.

6.8. Community and Occupational Health Hazard Mitigation Strategies

A number of strategies exist to reduce potential public health hazards and risks associated with well-stimulation-enabled oil and gas development activities. Most hazards have not been observed or measured in California, rendering it difficult to determine which hazards present risks at any given site in California. The most important hazards will not be identified until California-based studies document chemical compositions and release mechanisms, emission intensities, and potential for human exposure. As site-specific information becomes available, hazard mitigation strategies can be considered.

The following sections catalogue several potential community health and occupational hazard mitigation strategies. The strategies noted below highlight those among the more detailed mitigation recommendations provided above in this chapter as well as in Volume

II, Chapters 2 and 3. These strategies are to be considered in addition to employment of best practices in well-stimulation-enabled oil and gas development, which are employed to avoid exposure to a given hazard in the first place. It should be noted that mitigation and “best practices” should be systematically evaluated for effectiveness in the field, and even those mitigation practices with high efficacy are not effective if they are not properly executed and enforced.

6.8.1. Community Health Mitigation Practices

6.8.1.1. Setbacks

Exposures to environmental pollution and physical hazards such as light and noise falls off with distance from the source. The literature on oil and gas production suggests that the closer a population is to active oil and gas development, the more elevated the exposure, primarily to air pollutants but also to water pollutants, if a community relies on local aquifers for their drinking water, and zonal isolation of gases and fluids from aquifers is not achieved (see Section 6.4.1 above). While some California counties and municipalities have minimum surface setback requirements between oil and gas development and residences, schools, and other sensitive receptors, there are no such regulations at the state level. Further, the scientific literature is clear that certain sensitive and vulnerable populations (e.g., children, asthmatics, those with pre-existing cardiovascular or respiratory conditions, and populations already disproportionately exposed to elevated air pollution) are more susceptible to health effects from exposures to environmental pollutants known to be associated with oil and gas development (e.g., benzene) than others. The determination of sufficient setback distances should consider these sensitive populations.

Setback requirements have been instituted in some locales to decrease exposures to air pollutants, especially to VOCs that are known to be health damaging (e.g., benzene). The Dallas-Fort Worth area recently instituted a 460 meters (1,500 foot) minimum setback requirement between oil and gas wells and residences, schools, and other sensitive receptors. In summary, the scientific literature supports the recommendation for setbacks (City of Dallas, 2015). The distance of a setback would depend on factors such as the presence of sensitive receptors, such as schools, daycare centers, and residential elderly care facilities. The need for setbacks applies to all oil and gas wells, not just those that are stimulated.

6.8.1.2. Reduced Emission Completions and Other Air Pollutant Emission Reduction Technological Retrofits

As discussed in Volume II, Chapter 3, reductions of air pollutant emissions from well completions and other components of ancillary infrastructure have been demonstrated to reduce emission of methane, non-methane hydrocarbons, and VOCs during the oil and gas development process. Many of the non-methane VOCs contribute to background and regional tropospheric ozone concentrations and some are directly health damaging (e.g., benzene, toluene, ethylbenzene, xylene, formaldehyde, and hydrogen sulfide). Therefore, a reduction in emissions could decrease exposure of populations, especially at the local level, to harmful air pollutants. For a more complete discussion of these types of air pollutant emission mitigation technologies, please refer to Volume II, Chapter 3.

The deployment of mitigation technologies that have a demonstrated ability to reduce emissions in the laboratory or in small studies in the field do not necessary translate to actual reductions in air pollutants at scale if the sources of pollution increase. For example, Thompson et al. (2014) found that although regulations that strengthen rules about emission-reducing technologies in Colorado are much more stringent today than in 2008, emissions of VOCs have increased because of expansion of oil and gas development.

6.8.1.3. Use of Produced Water for Agricultural Irrigation

As noted in Chapter 2 of this volume, at least seven cases were identified that allow produced water to be used in agricultural irrigation in the San Joaquin Valley, with testing and treatment protocols that are insufficient to guarantee that well stimulation and other chemical constituents are at sufficiently low concentrations not to pose public health and occupational (farm worker) risks. To reduce public health risks that are potentially associated with the use of produced water for irrigation, prior to authorization to use produced water for irrigation, California should develop and implement testing and treatment protocols which account for stimulation chemicals and the other possible chemicals mobilized in the subsurface, prior to approving beneficial reuse of water produced from fields with well stimulation (and logically any produced water).

6.8.1.4. Water Source Switching

As noted in Chapter 2 of this volume, subsurface disposal of recovered fluid and produced water (Class II Underground Injection Control (UIC) wells) has been conducted in aquifers that are suitable for drinking water and other beneficial uses. The majority of Californians do not source their drinking water from such wells, and there has been no groundwater monitoring in the state to determine the number or the extent to which drinking water aquifers may be contaminated by well-stimulation-enabled oil development. Concerned households can eliminate their potential exposure by being provided with alternative drinking water sources that are known to be safe. It should be noted that water source switching is not be an alternative to the protection of drinking water resources.

6.8.2. Occupational Health Mitigation Practices

6.8.2.1. Personal Protective Equipment

The research is limited on the use of personal protective equipment (PPE) in the oil and gas extraction industry. A study on worker health and safety during flowback noted the routine use of PPE by workers at all sites, depending on work task (Esswein et al., 2014).

The PPE observed in use included flame-retardant clothing, steel toe boots, safety glasses, hard hats, and occasional use of fall protection, riggers gloves, and hearing protection. None of the workers observed in this study who experienced the highest exposure to silica sand and chemicals (flowback technicians, production watch technicians, or water management technicians) was observed wearing respirators, nor were they clean-shaven, which is necessary for proper respirator protection. Workers who wore half mask respirators during mixing of crystalline silica proppant were also not sufficiently protected, indicating that a similar study to this NIOSH assessment should be performed in California to assess worker exposure on the well pad.

6.8.2.2. Reducing Occupational Exposure to Silica

Mulloy (2014) identified opportunities for reducing silica exposure, including: elimination; substitution of ceramic or alternative proppants; proper engineering controls that minimize respiratory exposure; administrative control that limit worker time on site; and personal protection. Other recommendations included conducting workplace exposure assessments to characterize exposures to respirable crystalline silica; controlling exposures to the lowest concentrations achievable (and lower than the OSHA PEL or NIOSH REL); and ensuring that an effective respiratory protection program is in place that meets the OSHA Respiratory Protection Standards (Esswein et al., 2013).

6.9. Data Gaps

We need four types of information to assess environmental public health hazards:

1. The source and identity of the chemical substances (or stressor such as noise, traffic, etc.) of concern

2. A qualitative or quantitative measure of the outcome of the stressor, such as an acute or chronic toxicity factor,

3. Quantification of an emissions factor to air and/or water or a reporting of the quantity used.

4. Information about the number and plausibility of human exposure pathways associated either with emissions or quantities used. This factor is useful for hazard assessments and essential for risk assessments.

In preparing this hazard assessment, we have found that only for a minority of cases do we have information for items (1) identity, (2) outcome measure, (3) quantity/emission, and (4) exposure pathways. It is more common that we have (1) but not (2) or (3); (1) and (3) but not (2); or (1) and (2) and not (3). In some cases, for example some of the unidentified or ambiguously described components for the well treatment mixtures, we lack information on (1), (2) and (3). To add to our uncertainty, we find that even in cases where we have information about identity, toxicity, and/or quantity/emissions, there are significant concerns about the accuracy of the information.

6.10. Conclusions

The majority of important potential direct impacts of well stimulation result from the use of well stimulation chemicals. The large number of chemicals used in well stimulation makes it very difficult to judge the risks posed by accidental releases of stimulation fluids, such as those related to surface spills or unexpected subsurface pathways. Of the chemicals used, many are not sufficiently characterized to allow a full risk analysis.

There is a lack of information related to human exposure pathways for well-stimulation-enabled oil and gas development in California. For example, it is known that some produced water is diverted for agricultural use (see Chapter 2 in this volume); however, information regarding the composition of the fluids at the point of release and the environmental persistence, toxicity, and bioavailability of specific compounds in agricultural systems has not been studied. There is also a need to design and/or expand monitoring studies to better evaluate time activity patterns and personal exposure on and off-site for well-stimulation-enabled oil and gas development activities. Finally, it is important to extend the characterization of some on-site (occupational) exposures to off-site (community) exposures, i.e., for airborne silica proppant.

California-specific studies on the epidemiology of exposures to stimulation chemicals and stressors remain, by and large, non-existent. Although air and water quality studies suggest public health hazards exist, many data gaps remain, and more research is needed to clarify the magnitude of human-health risks and potential existing and future morbidity and mortality burdens associated with these concerns. It is clear that environmental public health science is playing catch up with well stimulation-enabled oil and gas development—and oil and gas development in general—across the country, and this is particularly notable in California.

Most of the studies included in this review of the literature were conducted in geographically and geologically diverse areas of the U.S., and may or may not be directly generalizable to the California context. Furthermore, much of the research on health risks has been conducted on the development of hydrocarbons from shale. While there are many similarities between the processes involved in the development of shale across the country and in the development of diatomite and other oil reservoirs in California, there are also a number of differences that increase and decrease public health hazards and potential public health risks (See Volume I).

There is no data on work-related fatalities related specifically to oil and gas development enabled by well stimulation, but the types of hazardous work activities during well stimulation are similar to those seen in general oil and gas extraction operations. Work-related fatality rates are significantly higher in the oil and gas development industry compared to the general industry average.

Work processes in oil and gas development, including that enabled by well stimulation, should be fully characterized to determine the specific risk factors for work-related injury and illness relative to risk factors for oil and gas production in general. Health effects among oil and gas development workers engaged in well stimulation should be monitored and evaluated to determine specific occupational health risk factors and harm-mitigation strategies to reduce the risk of deaths and serious injuries.

The current scientific literature and well stimulation chemical data available in California reveals that many of the well-stimulation-associated hazards have not been adequately characterized, nor have the associated environmental public health or occupational health risks been adequately analyzed—an observation that has been made by others (Adgate et al., 2014; Law et al., 2014; Kovats et al., 2014; New York Department of Health, 2014; NRC, 2014; Shonkoff et al., 2014). Studies of public health risk have failed to make clear whether the impact is caused by well stimulation or by oil development that is enabled by stimulation. Studies of health risks that differentiate the cause of the hazard would remedy this.

One of the most prominent key findings from our efforts to assess hazards is the significance of data gaps and the uncertainty that arises from these gaps in our confidence about characterizing human health risks for California.

This scientific literature review and hazard assessment, as well as other chapters in this volume, indicates that there are a number of potential human health hazards associated with well-stimulation-enabled oil and gas development in California with regards to air quality, water quality, and environmental exposure pathways. Our review also found that California-specific scientific assessments and datasets more generally on air, water, and human health are sparse. Additionally, human health monitoring data have not been adequately collected, let alone pursued. The hazard assessment of California-specific datasets on well stimulation chemistry indicates that more than half of the chemical constituents of stimulation fluids in California do not have any toxicity and/or use frequency or quantity information available, rendering it challenging to conclusively assess the magnitude of human health hazards associated with these processes. The emission of criteria and hazardous air pollutants have also only been monitored on the regional scale, and even in cases when these air pollutant emission factors are known, it is not possible, with the data available, to determine local emissions, community exposures, and subsequent population health risks.

We identified mitigation options that may reduce the magnitude of public health risks associated with well-stimulation-enabled oil and gas development in California; however, proper monitoring and enforcement are important components of sound mitigation that are often overlooked. Moreover, the data gaps that we identified create challenges in producing an adequately detailed assessment to provide clear guidance on the protection of public health, in the context of well-stimulation-enabled oil and gas development in California.

6.11. Recommendations

This chapter provides findings about what can and cannot be determined about potential impacts of well stimulation technology on human health, based on currently available information. One of the challenges that arise in efforts to study health risks for well-stimulation-enabled oil and gas development is the lack information available to carry out a standard hazard assessment and a broader risk characterization that requires information on exposure and dose-response. Here, we provide recommendations to address these information gaps.

6.11.1. Recommendation Regarding Chemical Use

The majority of important potential direct impacts of well stimulation result from the use of well stimulation chemicals. The large number of chemicals used in well stimulation makes it very difficult to judge the risks posed by accidental releases of stimulation fluids, such as those related to surface spills or unexpected subsurface pathways. Of the chemicals used, many are not sufficiently characterized to allow a full risk analysis.

Recommendation: *Operators should report the unique CASRN identification for all chemicals used in hydraulic fracturing and acid stimulation and the use of chemicals with unknown environmental profiles should be disallowed. The overall number of different chemicals should be reduced, and the use of more hazardous chemicals and chemicals with poor environmental profiles should be reduced, avoided or disallowed. The chemicals used in hydraulic fracturing could be limited to those on an approved list that would consist only of those chemicals with known and acceptable environmental hazard profiles. Operators should apply Green Chemistry principles to the formulation of hydraulic fracturing fluids.*

6.11.2. Recommendation Regarding Exposure and Health-Risk Information Gaps

This chapter identifies information gaps on hazards of substances used, the quantities and, in some cases, the identity of chemicals used for acidization and hydraulic fracturing, the magnitude of air emissions of well stimulation chemicals and fugitive emissions of oil and gas constituents, exposure pathways, and availability of acute and (in particular) chronic dose-response information.

Recommendation: *Conduct integrated research that cuts across multiple scientific disciplines and policy interests at relevant temporal and spatial scales in California, to answer key questions about the community and occupational impacts of oil and gas production enabled by well stimulation. Provide verification and validation of reported chemical use data, and conduct research to characterize the fate and transport of both intentional and unintentional chemical releases during well stimulation activities.*

6.11.3. Recommendation on Community Health

Oil and gas development—including that enabled by well stimulation—creates the risk of exposing human populations to a broad range of potentially hazardous substances (chemical and biological) or physical hazards (e.g., light and noise). For many of these hazards, we conclude that regional impacts associated with well stimulation activity are likely to be low, but exposures that can occur near well stimulation activity and enabled oil and gas development may result in elevated community health risks.

Recommendation: *Initiate studies in California to assess public health as a function of proximity to all oil and gas development, not just stimulated wells, and develop policies, for example science-based surface setbacks, to limit exposures.*

6.11.4. Recommendation on Occupational Health

Workers who are involved in oil and gas operations are exposed to chemical and physical hazards, some of which are specific to well stimulation activities, and many of which are general to the industry. Our review identified studies confirming occupational hazards related to well stimulation in states outside of California. There have been two peer-reviewed studies of occupational exposures attributable to hydraulic fracturing conducted by the National Institute for Occupational Safety and Health (NIOSH) across multiple states (not including California) and times of year. One of the studies found that respirable silica (silica sand is used as a proppant to hold open fractures formed in hydraulic fracturing) was in concentrations well in excess of occupational health and safety standards, in this case permissible exposure limits (PELs), by factors of as much as ten. Exposures exceeded PELs even when workers reported use of personal protective equipment. The second study found exposure to VOCs, especially benzene, above recommended occupational levels. The NIOSH studies are relevant for identifying hazards that could be significant for California workers, but no study to date has addressed occupational hazards associated with hydraulic fracturing and other forms of well stimulation in California.

Employers in the oil and gas industry must comply with existing California occupational safety and health regulations, and follow best practices to reduce and eliminate illness and injury risk to their employees. Employers can and often do implement comprehensive worker-protection programs that substantially reduce worker exposure and likelihood of illness and injury, but the effectiveness of these programs in California has not been evaluated. Engineering controls that reduce emissions could protect workers involved in well stimulation operations from chemical exposures and potentially reduce the likelihood of chemical exposure to the surrounding community.

Recommendation: *Design and execute California-based studies focused on silica and volatile organic compound exposures to workers engaged in hydraulic-fracturing-enabled oil and gas development processes, based on the NIOSH occupational health findings and protocols.*

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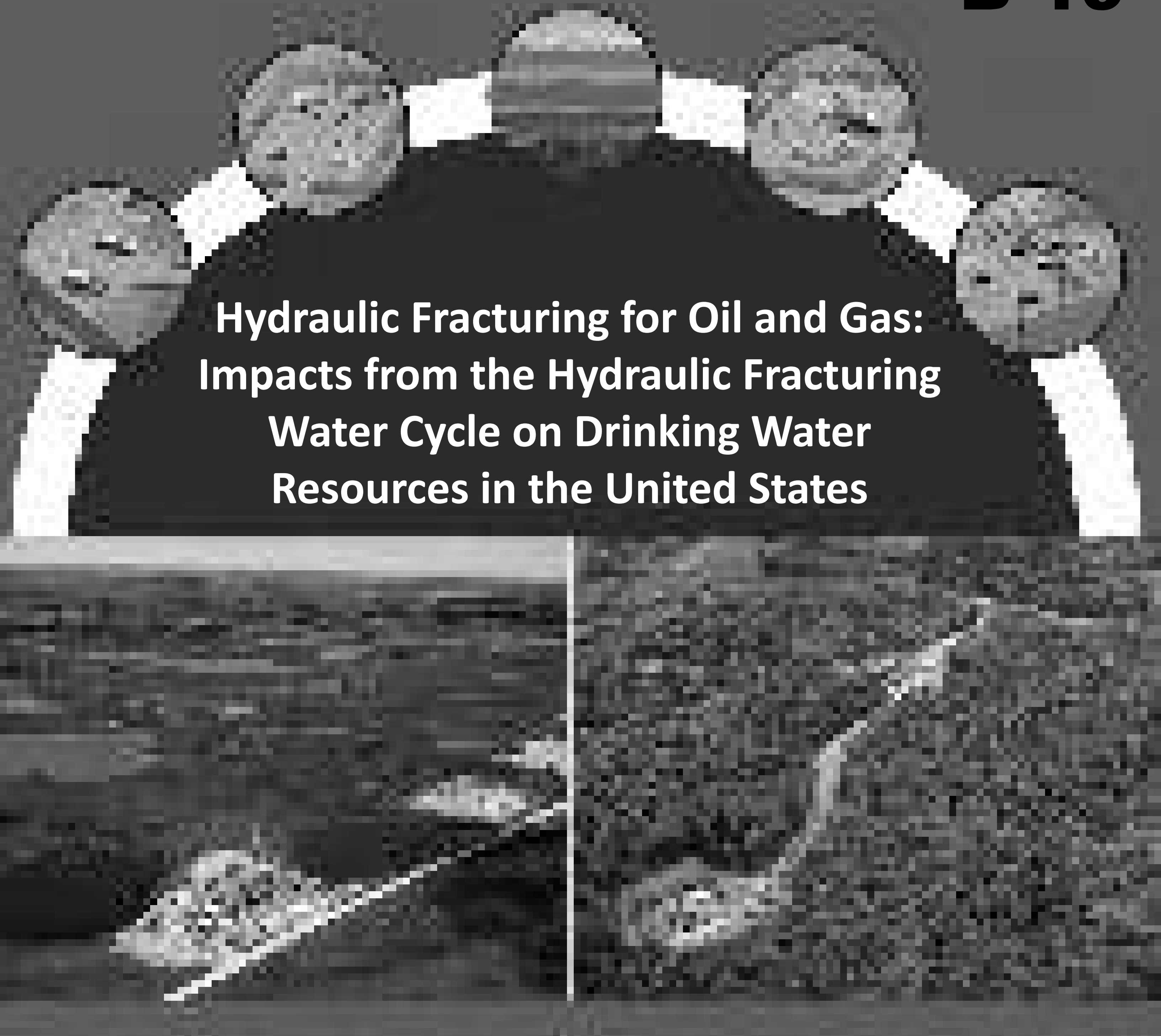
Hydraulic Fracturing for Oil and Gas: Impacts from the Hydraulic Fracturing Water Cycle on Drinking Water Resources in the United States



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List of Acronyms/Abbreviations

Acronym	Definition
2BE	2-butoxyethanol
ACToR	Aggregated Computational Toxicology Resource database
AME	Acton Mickelson Environmental, Inc.
AMEC	AMEC Environment & Infrastructure, Inc.
ANRC	Arkansas Natural Resources Commission
AO	administrative order
AOGC	Arkansas Oil and Gas Commission
API	American Petroleum Institute
ATSDR	Agency for Toxic Substance and Disease Registry
AWWA	American Water Works Association
BLM	Bureau of Land Management
BTEX	benzene, toluene, ethylbenzene, and xylenes
CARES	Casella Altela Regional Environmental Services
CASRN	chemical abstract services registration number
CBI	confidential business information
CBM	coalbed methane
CCST	California Council on Science and Technology
CDC	Centers for Disease Control and Prevention
CDWR	Colorado Division of Water Resources
CFR	Code of Federal Regulations
CICAD	Concise International Chemical Assessment Document
CM	chemical mixing
CMV	commercial motor vehicle
COGCC	Colorado Oil and Gas Conservation Commission
COWDF	Commercial Oil Field Waste Disposal Facilities
CWA	Clean Water Act
CWCB	Colorado Water Conservation Board
CWT	centralized waste treatment
CWTF	centralized water treatment facility
DBNM	dibromochloronitromethane
DBP	disinfection byproduct
DecaBDE	decabromodipheyl ether
DfE	Design for the Environment
DI	Drilling Info, Inc.
DMA	dimethylamine
DMR	Discharge Monitoring Report
DNR	Department of Natural Resources
DO	dissolved oxygen
DOC	dissolved organic carbon

DOE	U.S. Department of Energy
DOGGR	California Department of Conservation's Division of Oil, Gas & Geothermal Resources
DOI	U.S. Department of the Interior
DOJ	U.S. Department of Justice
DOT	U.S. Department of Transportation
DRB	Delaware River Basin
DRO	diesel range organics
DWSHA	Drinking Water Standards and Health Advisories
ECHA	European Chemicals Agency
EERC	Energy and Environmental Research Center, University of North Dakota
EIA	U.S. Energy Information Administration
EPA	U.S. Environmental Protection Agency
EPA OW	U.S. Environmental Protection Agency, Office of Water
EPI	estimation programs interface
EPWU	El Paso Water Utility
ERCB	Energy Resource Conservation Board
ERG	Eastern Research Group
ESN	Environmental Services Network
ESOD	erythrocyte Cu, Zn-superoxide dismutase
EWI	Energy Water Initiative
FDA	U.S. Food and Drug Administration
FOIA	Freedom of Information Act
FRS	fluids recovery services
GES	Groundwater & Environmental Services, Inc.
GHGRP	Greenhouse Gas Reporting Program
GNB	Government of New Brunswick
GRAS	generally recognized as safe
GRO	gasoline range organics
GTI	Gas Technology Institute
GWPC	Ground Water Protection Council
HBCD	hexabromocyclododecane
HDPE	high-density polyethylene
HF	hydraulic fracturing
HHBP	Human Health Benchmarks for Pesticides
HISA	Highly Influential Scientific Assessment
HPG	hydroxypropylguar
HTS	high throughput screening
HUC	hydrological unit code
IAEA	International Atomic Energy Agency
IARC	International Agency for Research on Cancer
IHS	Information Handling Services
IOGCC	Interstate Oil and Gas Compact Commission

IPCC	Intergovernmental Panel on Climate Change
IPCS	International Programme on Chemical Safety
IRIS	Integrated Risk Information System
IUPAC	International Union of Pure and Applied Chemistry
KWO	Kansas Water Office
LDEQ	Louisiana Department of Environmental Quality
LOAEL	lowest observed adverse effect level
MCDA	multicriteria decision analysis
MCL	maximum contaminant level
MCLG	maximum containment level goal
MCOR	Marcellus Center for Outreach and Research
MGD	million gallons per day
MIT	mechanical integrity test
MMCF	million cubic feet
MRL	minimum risk level
MSC	Marcellus shale coalition
MTBE	methyl tert-butyl ether
MVR	mechanical vapor recompression
NAS	National Academy of Sciences
NDDMR	North Dakota Department of Mineral Resources
NDDOH	North Dakota Department of Health
NDMA	N-nitrosodimethylamine
NDPES	National Pollution Discharge Elimination System
NDSWC	North Dakota State Water Commission
NETL	National Energy Technology Laboratory
NGO	non-governmental organization
NIH	National Institutes of Health
NM OCD	New Mexico Oil Conservation Division
NM OSE	New Mexico Office of the State Engineer
NOAEL	no observed adverse effect level
NORM	naturally occurring radioactive material
NPC	National Petroleum Council
NPDES	National Pollution Discharge Elimination System
NPDWR	National Primary Drinking Water Regulations
NRC	National Resource Council
NTP	U.S. National Toxicology Program
NYSDEC	New York State Department of Environmental Conservation
O&G	oil and gas
ODNR	Ohio Department of Natural Resources
DMRM	Division of Mineral Resources Management
OECD	The Organisation for Economic Co-operation and Development

OEPA	Ohio Environmental Protection Agency
OMB	Office of Management and Budget
ORB	Ohio River Basin
ORD	Office of Research and Development
OSF	oral slope factor
OSHA	Occupational Safety & Health Administration
OSWER	Office of Solid Water and Emergency Response
OWRB	Oklahoma Water Resources Board
PA DCNR	Pennsylvania Department of Conservation and Natural Resources
PA DEP	Pennsylvania Department of Environmental Protection
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PFBC	Pennsylvania Fish and Boat Commission
PDL	positive determination letter
PMF	Positive Matrix Factorization
PMN	pre-manufacturing notices
POD	point-of-departure
POTW	publicly owned treatment work
PPRTV	provisional peer-reviewed toxicity value
PVC	polyvinyl chloride
PWS	public water system
PWSA	Pittsburgh Water and Sewer Authority
QA	quality assurance
QAPP	quality assurance project plan
QC	quality control
QSAR	Quantitative Structure Activity Relationship
RAHC	reasonably anticipated to be a human carcinogen
RBC	red blood cells
RfD	reference dose
RfV	reference value
RO	reverse osmosis
SAB	Science Advisory Board
SAIC	Science Applications International Corporation
SAR	sodium adsorption ratio
SCN	thiocyanates
SDWA	Safe Drinking Water Act
SDWIS	Safe Drinking Water Information System
SEECO	Southern Electrical Equipment Company
SGEIS	supplemented generic environmental impact statement
SHS MSC	statewide health standards for medium-specific concentrations
SMCL	secondary maximum contaminant level
SPE	Society of Petroleum Engineers

SRB	Susquehanna River basin
SRBC	Susquehanna River Basin Commission
STO	Statoil
STRONGER	State review of oil and natural gas environmental regulations
SVOC	semi-volatile organic compounds
SWE	Southwestern Energy
TARM	TerrAqua Resource Management
TBA	tert-butyl alcohol
TDI	tolerable daily intake
TDS	total dissolved solids
TENORM	technologically enhanced naturally occurring radioactive material
THM	trihalomethane
TIPRO	Texas Independent Producers and Royalty Owners Association
TMDL	total maximum daily load
TOC	total organic carbon
TOPKAT	Toxicity Prediction by Komputer Assisted Technology
TPH	total petroleum hydrocarbons
TPHWG	Total Petroleum Hydrocarbon Criteria Working Group
TSS	total suspended solids
TTC	Threshold of Toxicological Concern
TTHM	total trihalomethane
TWDB	Texas Water Development Board
TXRRC	Texas Railroad Commission
UCRB	Upper Colorado River basin
UIC	underground injection control
UOG	unconventional oil and gas
USGAO	U.S. Government Accountability Office
USGS	U.S. Geological Survey
UWS	Universal Well Services
VES	viscoelastic surfactant
VOC	volatile organic compounds
WAWSA	Western Area Water Supply Authority
WFR	Well File Review
WHO	World Health Organization
WOE	weight of evidence
WRF	Water Research Foundation
WVDEP	West Virginia Department of Environmental Protection
WWTP	wastewater treatment plant
WYOGCC	Wyoming Oil and Gas Conservation Commission

Preface

Hydraulic fracturing is a technique used to increase oil and gas production from underground oil- or gas-bearing rock formations. Since the mid-2000s, the combination of hydraulic fracturing and directional drilling has become widespread, raising concerns about the potential impacts of hydraulic fracturing on drinking water resources. This concern is the focus of this report.

In 2010, the U.S. Environmental Protection Agency (EPA) initiated a study of the potential impacts of hydraulic fracturing activities on drinking water resources. The EPA defined the scope of its study to focus on the acquisition, use, disposal, and reuse of water used for hydraulic fracturing—what we call the hydraulic fracturing water cycle. This was done in recognition that concerns raised about potential impacts were not limited to the relatively short-term act of fracturing rock, but can include impacts related to other activities associated with hydraulic fracturing.

The EPA's study included the development of multiple research projects using the following research approaches: the analysis of existing data, scenario and modeling evaluations, laboratory studies, toxicological assessments, and five case studies. Throughout the study, the EPA engaged with stakeholders, including industry, the states, tribal nations, academia, and others, for input on the scope, approach, and initial results. To date, the study has resulted in the publication of multiple peer-reviewed scientific products, including 13 EPA technical reports and 14 journal articles.

This report represents the capstone product of the EPA's hydraulic fracturing drinking water study. It captures the state-of-the-science concerning drinking water impacts from activities in the hydraulic fracturing activities water cycle and integrates the results of the EPA's study of the subject with approximately 1,200 other publications and sources of information. The goals of this report were to assess the potential for activities in the hydraulic fracturing water cycle to impact the quality or quantity of drinking water resources and to identify factors that affect the frequency or severity of those impacts.

This report is a science document and does not present or evaluate policy options or make policy recommendations. A draft of this report was reviewed by the EPA's independent Science Advisory Board (SAB). Reflecting the complexity of the subject, the expert ad hoc panel formed by the SAB was the largest ever convened for the review of a scientific product. Combined with over 100,000 comments submitted by members of the public, SAB comments helped the EPA to refine, clarify, and better support the final conclusions presented in this report.

The release of this final assessment report marks the completion of the EPA's hydraulic fracturing drinking water study. The study has already prompted increased dialogue among industry, the states, tribal nations, the public, and others concerning how drinking water resources can be better protected in areas where hydraulic fracturing is occurring or being considered. However, there are data gaps and uncertainties limiting our understanding of the impacts of hydraulic fracturing activities on drinking water resources. As additional data become available, and with continued dialogue among stakeholders, our understanding of the potential impacts of hydraulic fracturing on drinking water resources will improve.

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Executive Summary

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Executive Summary

People rely on clean and plentiful water resources to meet their basic needs, including drinking, bathing, and cooking. In the early 2000s, members of the public began to raise concerns about potential impacts on their drinking water from hydraulic fracturing at nearby oil and gas production wells. In response to these concerns, Congress urged the U.S. Environmental Protection Agency (EPA) to study the relationship between hydraulic fracturing for oil and gas and drinking water in the United States.

The goals of the study were to assess the potential for activities in the hydraulic fracturing water cycle to impact the quality or quantity of drinking water resources and to identify factors that affect the frequency or severity of those impacts. To achieve these goals, the EPA conducted independent research, engaged stakeholders through technical workshops and roundtables, and reviewed approximately 1,200 cited sources of data and information. The data and information gathered through these efforts served as the basis for this report, which represents the culmination of the EPA's study of the potential impacts of hydraulic fracturing for oil and gas on drinking water resources.

The hydraulic fracturing water cycle describes the use of water in hydraulic fracturing, from water withdrawals to make hydraulic fracturing fluids, through the mixing and injection of hydraulic fracturing fluids in oil and gas production wells, to the collection and disposal or reuse of produced water. These activities can impact drinking water resources under some circumstances. Impacts can range in frequency and severity, depending on the combination of hydraulic fracturing water cycle activities and local- or regional-scale factors. The following combinations of activities and factors are more likely than others to result in more frequent or more severe impacts:

- Water withdrawals for hydraulic fracturing in times or areas of low water availability, particularly in areas with limited or declining groundwater resources;
- Spills during the management of hydraulic fracturing fluids and chemicals or produced water that result in large volumes or high concentrations of chemicals reaching groundwater resources;
- Injection of hydraulic fracturing fluids into wells with inadequate mechanical integrity, allowing gases or liquids to move to groundwater resources;
- Injection of hydraulic fracturing fluids directly into groundwater resources;
- Discharge of inadequately treated hydraulic fracturing wastewater to surface water resources; and
- Disposal or storage of hydraulic fracturing wastewater in unlined pits, resulting in contamination of groundwater resources.

The above conclusions are based on cases of identified impacts and other data, information, and analyses presented in this report. Cases of impacts were identified for all stages of the hydraulic fracturing water cycle. Identified impacts generally occurred near hydraulically fractured oil and gas production wells and ranged in severity, from temporary changes in water quality to contamination that made private drinking water wells unusable.

The available data and information allowed us to qualitatively describe factors that affect the frequency or severity of impacts at the local level. However, significant data gaps and uncertainties in the available data prevented us from calculating or estimating the national frequency of impacts on drinking water resources from activities in the hydraulic fracturing water cycle. The data gaps and uncertainties described in this report also precluded a full characterization of the severity of impacts.

The scientific information in this report can help inform decisions by federal, state, tribal, and local officials; industry; and communities. In the short-term, attention could be focused on the combinations of activities and factors outlined above. In the longer-term, attention could be focused on reducing the data gaps and uncertainties identified in this report. Through these efforts, current and future drinking water resources can be better protected in areas where hydraulic fracturing is occurring or being considered.

Drinking Water Resources in the United States

In this report, drinking water resources are defined as any water that now serves, or in the future could serve, as a source of drinking water for public or private use. This includes both surface water resources and groundwater resources (Text Box ES-1). In 2010, approximately 58% of the total volume of water withdrawn for public and non-public water supplies came from surface water resources and approximately 42% came from groundwater resources (Maupin et al., 2014).[1] Most people (86% of the population) in the United States relied on public water supplies for their drinking water in 2010, and approximately 14% of the population obtained drinking water from non-public water supplies. Non-public water supplies are often private water wells that supply drinking water to a residence.

Future access to high-quality drinking water in the United States will likely be affected by changes in climate and water use. Since 2000, about 30% of the total area of the contiguous United States has experienced moderate drought conditions and about 20% has experienced severe drought conditions. Declines in surface water resources have led to increased withdrawals and net depletions of groundwater in some areas. As a result, non-fresh water resources (e.g., wastewater from sewage treatment plants, brackish groundwater and surface water, and seawater) are increasingly treated and used to meet drinking water demand.

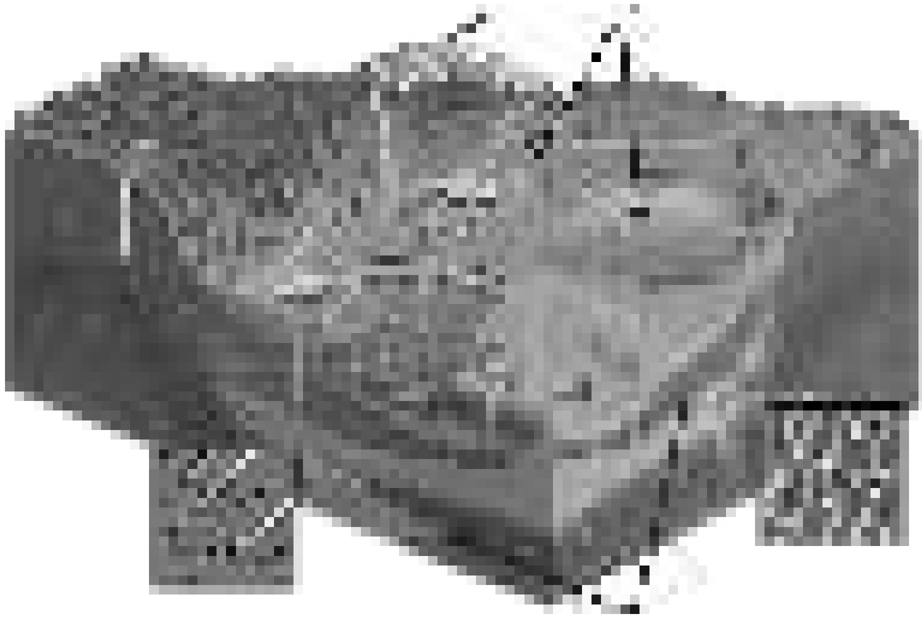
Natural processes and human activities can affect the quality and quantity of current and future drinking water resources. This report focuses on the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources; other processes or activities are not discussed.

[1] Public water systems provide water for human consumption from surface or groundwater through pipes or other infrastructure to at least 15 service connections or serve an average of at least 25 people for at least 60 days a year. Non-public water systems have fewer than 15 service connections and serve fewer than 25 individuals.

Text Box ES-1. Drinking Water Resources.

In this report, drinking water resources are considered to be any water that now serves, or in the future could serve, as a source of drinking water for public or private use. This includes both surface water bodies and underground rock formations that contain water.

Surface water resources include water bodies located on the surface of the Earth. Rivers, springs, lakes, and reservoirs are examples of surface water resources. Water quality and quantity are often considered when determining whether a surface water resource could be used as a drinking water resource.



Groundwater resources are underground rock formations that contain water. Groundwater resources are found at different depths nearly everywhere in the United States. Resource depth, water quality, and water yield are often considered when determining whether a groundwater resource could be used as a drinking water resource.

Hydraulic Fracturing for Oil and Gas in the United States

Hydraulic fracturing is frequently used to enhance oil and gas production from underground rock formations and is one of many activities that occur during the life of an oil and gas production well (Figure ES-1). During hydraulic fracturing, hydraulic fracturing fluid is injected down an oil or gas production well and into the targeted rock formation under pressures great enough to fracture the oil- and gas-bearing rock.[1] The hydraulic fracturing fluid usually carries proppant (typically sand) into the newly-created fractures to keep the fractures "propped" open. After hydraulic fracturing, oil, gas, and other fluids flow through the fractures and up the production well to the surface, where they are collected and managed.

[1] The targeted rock formation (sometimes called the "target zone" or "production zone") is the portion of a subsurface rock formation that contains the oil or gas to be extracted.

Figure ES-1. General timeline and summary of activities at a hydraulically fractured oil or gas production well.

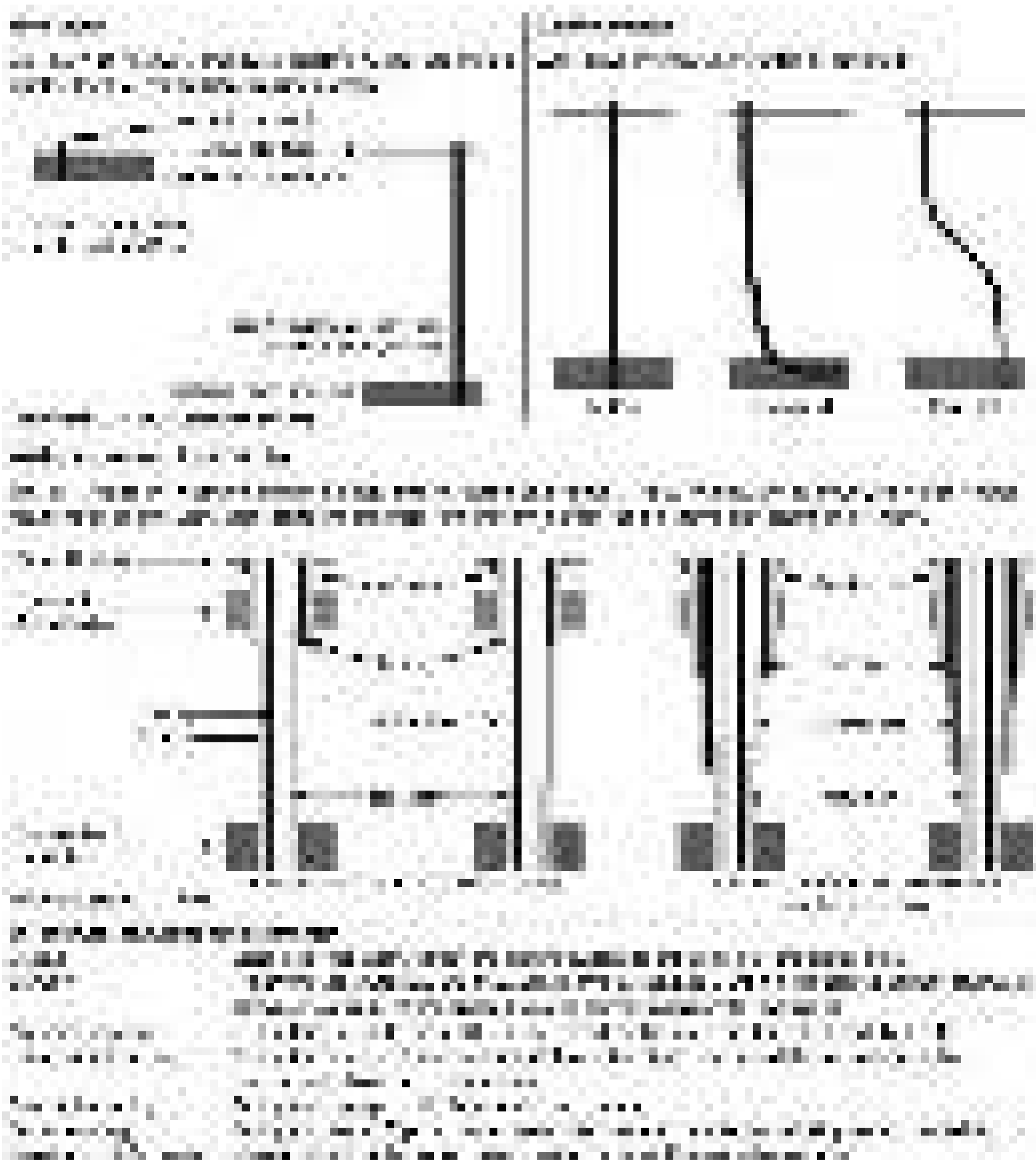
Hydraulically fractured oil and gas production wells have significantly contributed to the surge in domestic oil and gas production, accounting for slightly more than 50% of oil production and nearly 70% of gas production in 2015 (EIA, 2016c, d). The surge occurred when hydraulic fracturing was combined with directional drilling technologies around 2000. Directional drilling allows oil and gas production wells to be drilled horizontally or directionally along the targeted rock formation, exposing more of the oil- or gas-bearing rock formation to the production well. When combined with directional drilling technologies, hydraulic fracturing expanded oil and gas production to oil- and gas-bearing rock formations previously considered uneconomical. Although hydraulic fracturing is commonly associated with oil and gas production from deep, horizontal wells drilled into shale (e.g., the Marcellus Shale in Pennsylvania or the Bakken Shale in North Dakota), it has been used in a variety of oil and gas production wells (Text Box ES-2) and other types of oil- or gas-bearing rock (e.g., sandstone, carbonate, and coal).

Approximately 1 million wells have been hydraulically fractured since the technique was first developed in the late 1940s (Gallegos and Varela, 2015; IOGCC, 2002). Roughly one third of those wells were hydraulically fractured between 2000 and approximately 2014. Wells hydraulically fractured between 2000 and 2013 were located in pockets of activity across the United States (Figure ES-2). Based on several different data compilations, we estimate that 25,000 to 30,000 new wells were drilled and hydraulically fractured in the United States each year between 2011 and 2014, in addition to existing wells that were hydraulically fractured to increase production.[1] Following the decline in oil and gas prices, the number of new wells drilled and hydraulically fractured appears to have decreased, with about 20,000 new wells drilled and hydraulically fractured in 2015.

[1] See Table 3-1 in Chapter 3.

Text Box ES-2. Hydraulically Fractured Oil and Gas Production Wells.

Hydraulically fractured oil and gas production wells come in different shapes and sizes. They can have different depths, orientations, and construction characteristics. They can include new wells (i.e., wells that are hydraulically fractured soon after construction) and old wells (i.e., wells that are hydraulically fractured after producing oil and gas for some time).



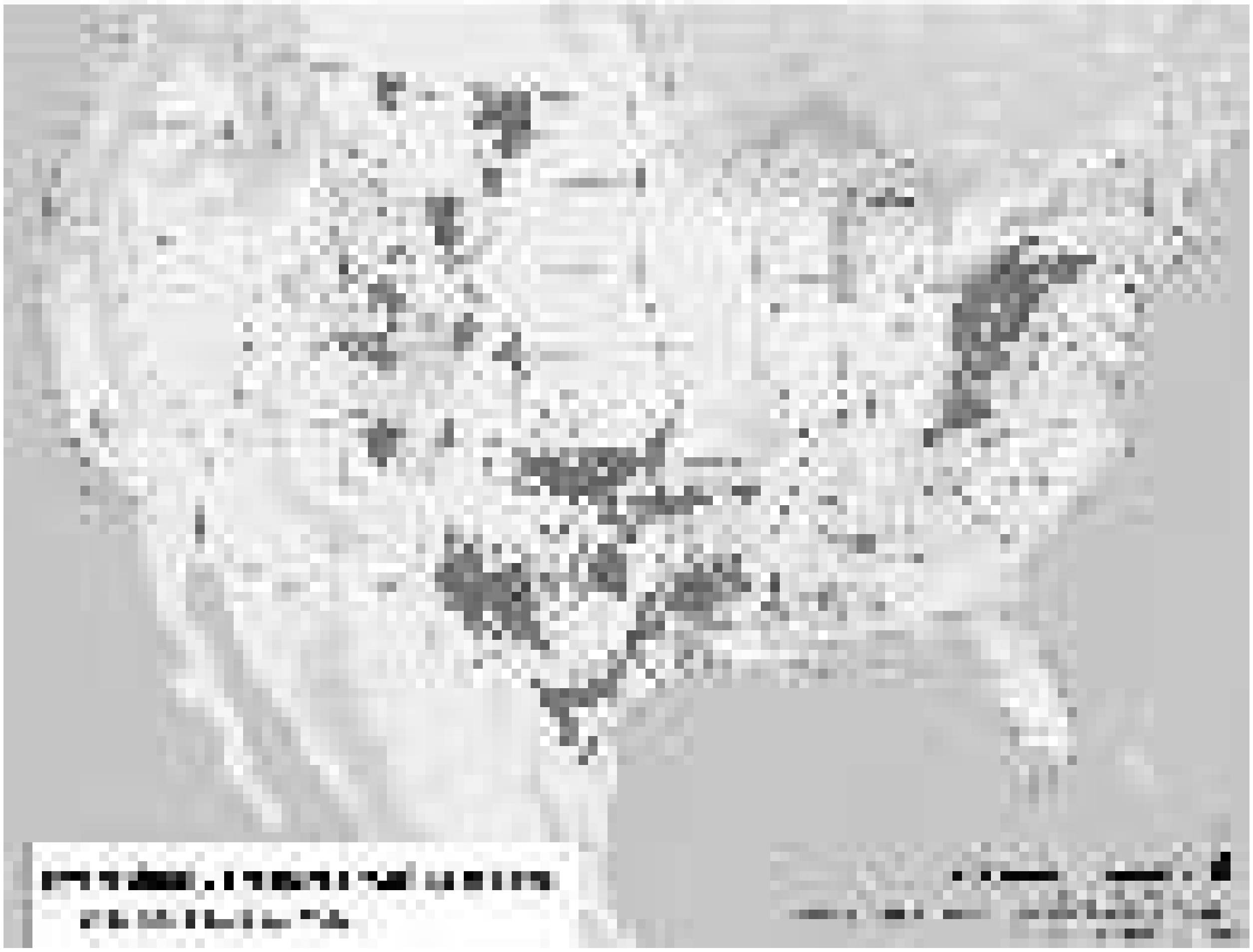


Figure ES-2. Locations of approximately 275,000 wells that were drilled and likely hydraulically fractured between 2000 and 2013.
Data from DrillingInfo (2014a).

Hydraulically fractured oil and gas production wells can be located near or within sources of drinking water. Between 2000 and 2013, approximately 3,900 public water systems were estimated to have had at least one hydraulically fractured well within 1 mile of their water source; these public water systems served more than 8.6 million people year-round in 2013. An additional 3.6 million people were estimated to have obtained drinking water from non-public water supplies in counties with at least one hydraulically fractured well.[1] Underground, hydraulic fracturing can occur in close vertical proximity to drinking water resources. In some parts of the United States (e.g., the Powder River Basin in Montana and Wyoming), there is no vertical distance between the top of the hydraulically fractured oil- or gas-bearing rock formation and the bottom of treatable water, as determined by data from state oil and gas agencies and state geological survey data.[2] In other parts of the country (e.g., the Eagle Ford Shale in Texas), there can be thousands of feet of

[1] This estimate only includes counties in which 30% or more of the population (i.e., two or more times the national average) relied on non-public water supplies in 2010. See Section 2.5 in Chapter 2.

[2] In these cases, water that is naturally found in the oil- and gas-bearing rock formation meets the definition of drinking water in some parts of the basin. See Section 6.3.2 in Chapter 6.

rock that separate treatable water from the hydraulically fractured oil- or gas-bearing rock formation. When hydraulically fractured oil and gas production wells are located near or within drinking water resources, there is a greater potential for activities in the hydraulic fracturing water cycle to impact those resources.

Approach: The Hydraulic Fracturing Water Cycle

The EPA studied the relationship between hydraulic fracturing for oil and gas and drinking water resources using the hydraulic fracturing water cycle (Figure ES-3). The hydraulic fracturing water cycle has five stages; each stage is defined by an activity involving water that supports hydraulic fracturing. The stages and activities of the hydraulic fracturing water cycle include:

- **Water Acquisition:** the withdrawal of groundwater or surface water to make hydraulic fracturing fluids;
- **Chemical Mixing:** the mixing of a base fluid (typically water), proppant, and additives at the well site to create hydraulic fracturing fluids;[1]
- **Well Injection:** the injection and movement of hydraulic fracturing fluids through the oil and gas production well and in the targeted rock formation;
- **Produced Water Handling:** the on-site collection and handling of water that returns to the surface after hydraulic fracturing and the transportation of that water for disposal or reuse;[2] and
- **Wastewater Disposal and Reuse:** the disposal and reuse of hydraulic fracturing wastewater.[3]

Potential impacts on drinking water resources from the above activities are considered in this report. We do not address other concerns that have been raised by stakeholders about hydraulic fracturing (e.g., potential air quality impacts or induced seismicity) or other oil and gas exploration and production activities (e.g., environmental impacts from site selection and development), as these were not included in the scope of the study. Additionally, this report is not a human health risk assessment; it does not identify populations exposed to hydraulic fracturing-related chemicals, and it does not estimate the extent of exposure or estimate the incidence of human health impacts.

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[1] A base fluid is the fluid into which proppants and additives are mixed to make a hydraulic fracturing fluid; water is an example of a base fluid. Additives are chemicals or mixtures of chemicals that are added to the base fluid to change its properties.

[2] "Produced water" is defined in this report as water that flows from and through oil and gas wells to the surface as a by-product of oil and gas production.

[3] "Hydraulic fracturing wastewater" is defined in this report as produced water from hydraulically fractured oil and gas wells that is being managed using practices that include, but are not limited to, injection in Class II wells, reuse in other hydraulic fracturing operations, and various aboveground disposal practices. The term "wastewater" is being used as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes. Class II wells are used to inject wastewater associated with oil and gas production underground and are regulated under the Underground Injection Control Program of the Safe Drinking Water Act.

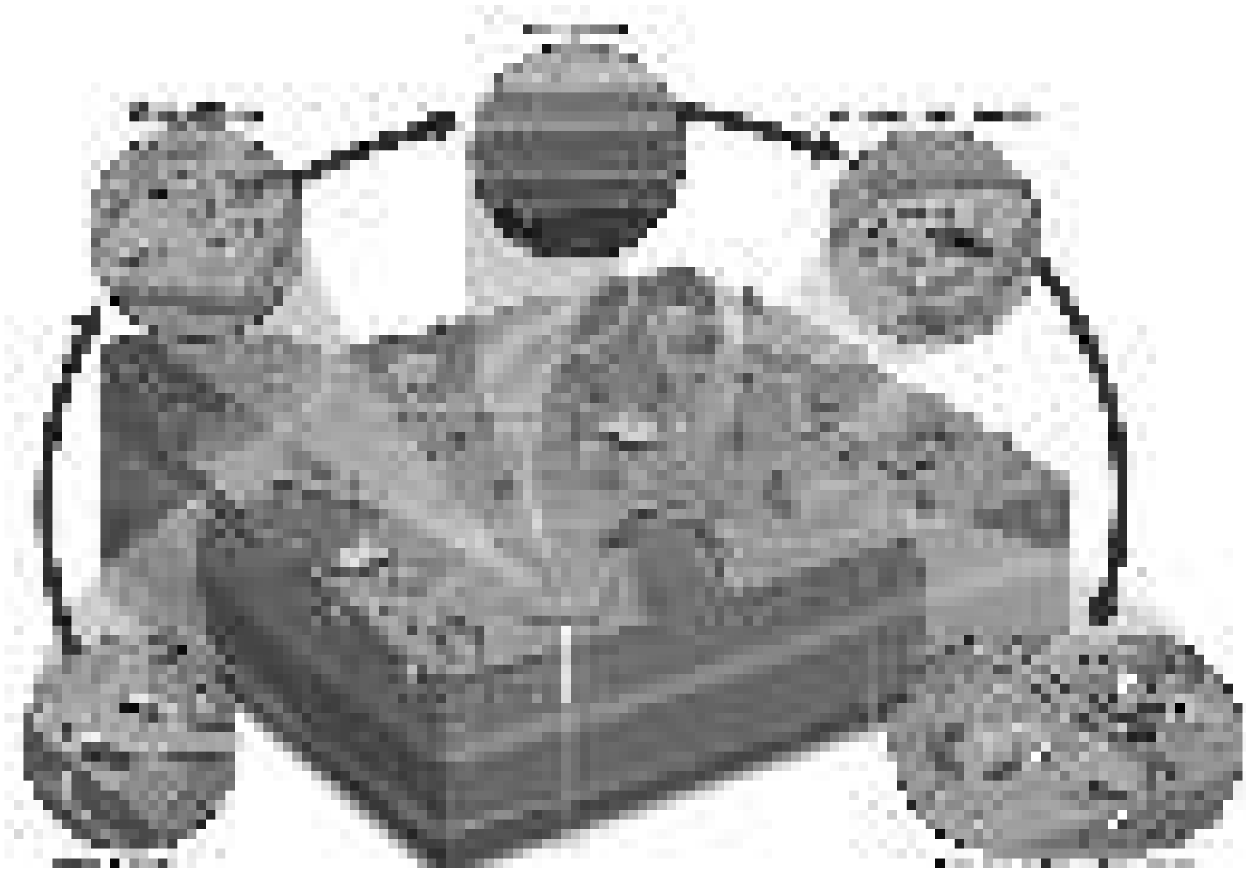

Figure ES-3. The five stages of the hydraulic fracturing water cycle.
The stages (shown in the insets) identify activities involving water that support hydraulic fracturing for oil and gas. Activities may take place in the same watershed or different watersheds and close to or far from drinking water resources. Thin arrows in the insets depict the movement of water and chemicals. Specific activities in the "Wastewater Disposal and Reuse" inset include (a) disposal of wastewater through underground injection, (b) wastewater treatment followed by reuse in other hydraulic fracturing operations or discharge to surface waters, and (c) disposal through evaporation or percolation pits.

Each stage of the hydraulic fracturing water cycle was assessed to identify (1) the potential for impacts on drinking water resources and (2) factors that affect the frequency or severity of impacts. Specific definitions used in this report are provided below:

- An **impact** is any change in the quality or quantity of drinking water resources, regardless of severity, that results from an activity in the hydraulic fracturing water cycle.
- A **factor** is a feature of hydraulic fracturing operations or an environmental condition that affects the frequency or severity of impacts.
- **Frequency** is the number of impacts per a given unit (e.g., geographic area, unit of time, number of hydraulically fractured wells, or number of water bodies).
- **Severity** is the magnitude of change in the quality or quantity of a drinking water resource as measured by a given metric (e.g., duration, spatial extent, or contaminant concentration).

Factors affecting the frequency or severity of impacts were identified because they describe conditions under which impacts are more or less likely to occur and because they could inform the development of future strategies and actions to prevent or reduce impacts. Although no attempt was made to identify or evaluate best practices, ways to reduce the frequency or severity of impacts from activities in the hydraulic fracturing water cycle are described in this report when they were reported in the scientific literature. Laws, regulations, and policies also exist to protect drinking water resources, but a comprehensive summary and broad evaluation of current or proposed regulations and policies was beyond the scope of this report.

Relevant scientific literature and data were evaluated for each stage of the hydraulic fracturing water cycle. Literature included articles published in science and engineering journals, federal and state government reports, non-governmental organization reports, and industry publications. Data sources included federal- and state-collected data sets, databases maintained by federal and state government agencies, other publicly available data, and industry data provided to the EPA.[1] The relevant literature and data complement research conducted by the EPA under its *Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (Text Box ES-3).

Text Box ES-3. The EPA's *Study of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking Water Resources*.

The EPA's study is the first national study of the potential impacts of hydraulic fracturing for oil and gas on drinking water resources. It included independent research projects conducted by EPA scientists and contractors and a state-of-the-science assessment of available data and information on the relationship between hydraulic fracturing and drinking water resources (i.e., this report).

Throughout the study, the EPA consulted with the Agency's independent Science Advisory Board (SAB) on the scope of the study and the progress made on the research projects. The SAB also conducted a peer review of both the *Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2011d; referred to as the Study Plan in this report) and a draft of this report.

Stakeholder engagement also played an important role in the development and implementation of the study. While developing the scope of the study, the EPA held public meetings to get input from stakeholders on the study scope and design. While conducting the study, the EPA requested information from the public and engaged with technical, subject-matter experts on topics relevant to the study in a series of technical workshops and roundtables. For more information on the EPA's study, including the role of the SAB and stakeholders, visit www.epa.gov/hfstudy.

[1] Industry data was provided to the EPA in response to two separate information requests to oil and gas service companies and oil and gas production well operators. Some of these data were claimed as confidential business information under the Toxic Substances Control Act and were treated as such in this report.

A draft of this report underwent peer review by the EPA's Science Advisory Board (SAB). The SAB is an independent federal advisory committee that often conducts peer reviews of high-profile scientific matters relevant to the EPA. Members of the SAB and ad hoc panels formed under the auspices of the SAB are nominated by the public and selected based on factors such as technical expertise, knowledge, experience, and absence of any real or perceived conflicts of interest. Peer review comments provided by the SAB and public comments submitted to the SAB during their peer review, including comments on major conclusions and technical content, were carefully considered in the development of this final document.

A summary of the activities in the hydraulic fracturing water cycle and their potential to impact drinking water resources is provided below, including what is known about human health hazards associated with chemicals identified across all stages of the hydraulic fracturing water cycle. Additional details are available in the full report.

Water Acquisition

Activity: The withdrawal of groundwater or surface water to make hydraulic fracturing fluids.

Relationship to Drinking Water Resources: Groundwater and surface water resources that provide water for hydraulic fracturing fluids can also provide drinking water for public or non-public water supplies.

Water is the major component of nearly all hydraulic fracturing fluids, typically making up 90–97% of the total fluid volume injected into a well. The median volume of water used, per well, for hydraulic fracturing was approximately 1.5 million gallons (5.7 million liters) between January 2011 and February 2013, as reported in FracFocus 1.0 (Text Box ES-4). There was wide variation in the water volumes reported per well, with 10^{th} and 90^{th} percentiles of 74,000 gallons (280,000 liters) and 6 million gallons (23 million liters) per well, respectively. There was also variation in water use per well within and among states (Table ES-1). This variation likely results from several factors, including the type of well, the fracture design, and the type of hydraulic fracturing fluid used. An analysis of hydraulic fracturing fluid data from Gallegos et al. (2015) indicates that water volumes used per well have increased over time as more horizontal wells have been drilled.

Water used for hydraulic fracturing is typically fresh water taken from available groundwater and/or surface water resources located near hydraulically fractured oil and gas production wells. Water sources can vary across the United States, depending on regional or local water availability; laws, regulations, and policies; and water management practices. Hydraulic fracturing operations in the humid eastern United States generally rely on surface water resources, whereas operations in the arid and semi-arid western United States generally rely on groundwater or surface water. Geographic differences in water use for hydraulic fracturing are illustrated in Figure ES-4, which shows that most of the water used for hydraulic fracturing in the Marcellus Shale region of the Susquehanna River Basin came from surface water resources between approximately 2008 and 2013. In comparison, less than half of the water used for hydraulic fracturing in the Barnett Shale region of Texas came from surface water resources between approximately 2011 and 2013.

Text Box ES-4. FracFocus Chemical Disclosure Registry.

The FracFocus Chemical Disclosure Registry is a publicly-accessible website (www.fracfocus.org) managed by the Ground Water Protection Council (GWPC) and the Interstate Oil and Gas Compact Commission (IOGCC). Oil and gas production well operators can disclose information at this website about water and chemicals used in hydraulic fracturing fluids at individual wells. In many states where oil and gas production occurs, well operators are required to disclose to FracFocus well-specific information on water and chemical use during hydraulic fracturing.

The GWPC and the IOGCC provided the EPA with over 39,000 PDF disclosures submitted by well operators to FracFocus (version 1.0) before March 1, 2013. Data in the disclosures were extracted and compiled in a project database, which was used to conduct analyses on water and chemical use for hydraulic fracturing. Analyses were conducted on over 38,000 unique disclosures for wells located in 20 states that were hydraulically fractured between January 1, 2011, and February 28, 2013.

Despite the challenge of adapting a dataset originally created for local use and single-PDF viewing to answer broader questions, the project database created by the EPA provided substantial insight into water and chemical use for hydraulic fracturing. The project database represents the data reported to FracFocus 1.0 rather than all hydraulic fracturing that occurred in the United States during the study time period. The project database is an incomplete picture of all hydraulic fracturing due to voluntary reporting in some states for certain time periods (in the absence of state reporting requirements), the omission of information on confidential chemicals from disclosures, and invalid or erroneous information in the original disclosures or created during the development of the database. The development of FracFocus 2.0, which became the exclusive reporting mechanism in June 2013, was intended to increase the quality, completeness, and consistency of the data submitted by providing dropdown menus, warning and error messages during submission, and automatic formatting of certain fields. The GWPC has announced additional changes and upgrades for FracFocus 3.0 to enhance data searchability, increase system security, provide greater data accuracy, and further increase data transparency.

Table ES-1. Water use per hydraulically fractured well between January 2011 and February 2013.
Medians and percentiles were calculated from data submitted to FracFocus 1.0 (Appendix B).

State	Number of FracFocus 1.0 disclosures	Median volume per well (gallons)	10th percentile (gallons)	90th percentile (gallons)
Arkansas	1,423	5,259,965	3,234,963	7,121,249
California	711	76,818	21,462	285,306
Colorado	4,898	463,462	147,353	3,092,024
Kansas	121	1,453,788	10,836	2,227,926
Louisiana	966	5,077,863	1,812,099	7,945,630
Montana	207	1,455,757	367,326	2,997,552
New Mexico	1,145	175,241	35,638	1,871,666
North Dakota	2,109	2,022,380	969,380	3,313,482
Ohio	146	3,887,499	2,885,568	5,571,027
Oklahoma	1,783	2,591,778	1,260,906	7,402,230
Pennsylvania	2,445	4,184,936	2,313,649	6,615,981
Texas	16,882	1,420,613	58,709	6,115,195
Utah	1,406	302,075	76,286	769,360
West Virginia	273	5,012,238	3,170,210	7,297,080
Wyoming	1,405	322,793	5,727	1,837,602

Hydraulic fracturing wastewater and other lower-quality water can also be used in hydraulic fracturing fluids to offset the need for fresh water, although the proportion of injected fluid that is reused hydraulic fracturing wastewater varies by location (Figure ES-4).[1] Overall, the proportion of

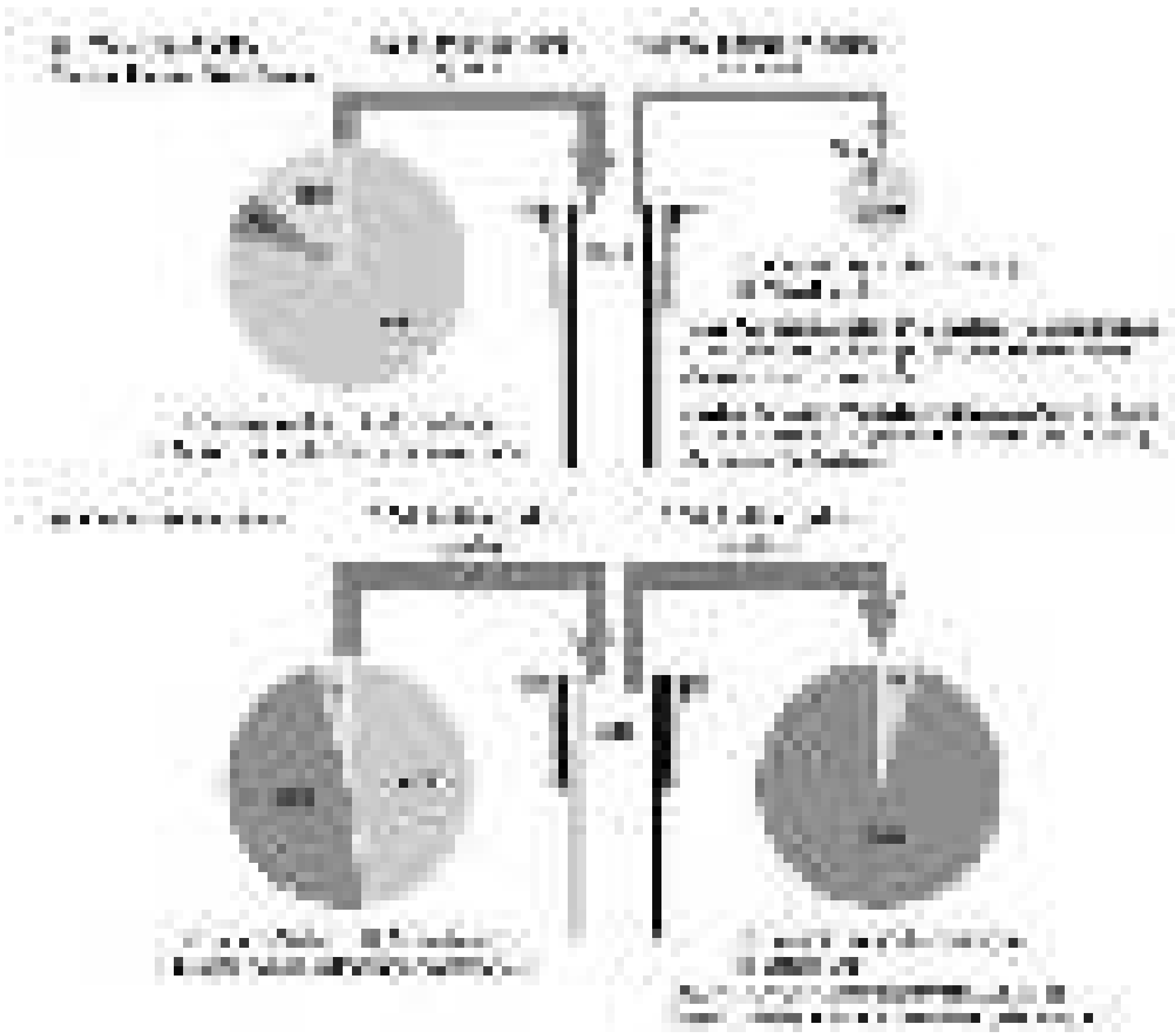

Figure ES-4. Water budgets illustrative of hydraulic fracturing water management practices in the Marcellus Shale in the Susquehanna River Basin between approximately 2008 and 2013 and the Barnett Shale in Texas between approximately 2011 and 2013.
Class II wells are used to inject wastewater associated with oil and gas production underground and are regulated under the Underground Injection Control Program of the Safe Drinking Water Act. Data sources are described in Figure 10-1 in Chapter 10.

[1] Reused hydraulic fracturing wastewater as a percentage of injected fluid differs from the percentage of produced water that is managed through reuse in other hydraulic fracturing operations. For example, in the Marcellus Shale region of the Susquehanna River Basin, approximately 14% of injected fluid was reused hydraulic fracturing wastewater, while approximately 90% of produced water was managed through reuse in other hydraulic fracturing operations (Figure ES-4a).

water used in hydraulic fracturing that comes from reused hydraulic fracturing wastewater appears to be low. In a survey of literature values from 10 states, basins, or plays, the median percentage of the injected fluid volume that came from reused hydraulic fracturing wastewater was 5% between approximately 2008 and 2014.[1] There was an increase in the reuse of hydraulic fracturing wastewater as a percentage of the injected hydraulic fracturing fluid in both Pennsylvania and West Virginia between approximately 2008 and 2014. This increase is likely due to the limited availability of Class II wells, which are commonly used to dispose of oil and gas wastewater, and the costs of trucking wastewater to Ohio, where Class II wells are more prevalent.[2] Class II wells are also prevalent in Texas, and the reuse of wastewater in hydraulic fracturing fluids in the Barnett Shale appears to be lower than in the Marcellus Shale (Figure ES-4).

Because the same water resource can be used to support hydraulic fracturing and to provide drinking water, withdrawals for hydraulic fracturing can directly impact drinking water resources by changing the quantity or quality of the remaining water. Although every water withdrawal affects water quantity, we focused on water withdrawals that have the potential to significantly impact drinking water resources by limiting the availability of drinking water or altering its quality. Water withdrawals for a single hydraulically fractured oil and gas production well are not expected to significantly impact drinking water resources, because the volume of water needed to hydraulically fracture a single well is unlikely to limit the availability of drinking water or alter its quality. If, however, multiple oil and gas production wells are located within an area, the total volume of water needed to hydraulically fracture all of the wells has the potential to be a significant portion of the water available and impacts on drinking water resources can occur.

To assess whether hydraulic fracturing operations are a relatively large or small user of water, we compared water use for hydraulic fracturing to total water use at the county level (Text Box ES-5). In most counties studied, the average annual water volumes reported in FracFocus 1.0 were generally less than 1% of total water use. This suggests that hydraulic fracturing operations represented a relatively small user of water in most counties. There were exceptions, however. Average annual water volumes reported in FracFocus 1.0 were 10% or more of total water use in 26 of the 401 counties studied, 30% or more in nine counties, and 50% or more in four counties.[3] In these counties, hydraulic fracturing operations represented a relatively large user of water.

The above results suggest that hydraulic fracturing operations can significantly increase the volume of water withdrawn in particular areas. Increased water withdrawals can result in significant impacts on drinking water resources if there is insufficient water available in the area to accommodate all users. To assess the potential for these impacts, we compared hydraulic fracturing water use to estimates of water availability at the county level.[4] In most counties studied, average

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[1] See Section 4.2 in Chapter 4.

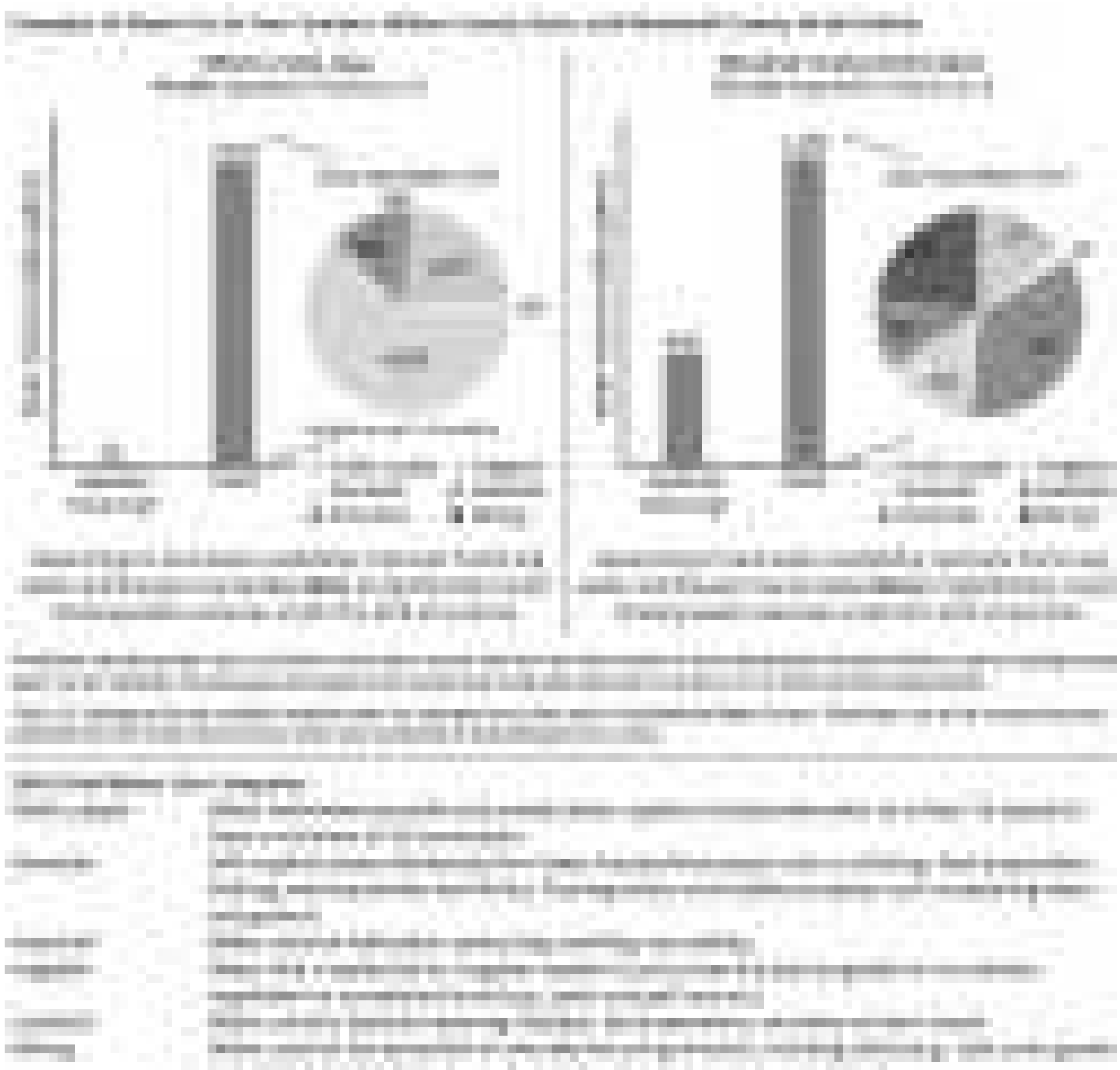
[2] See Chapter 8 for additional information on Class II wells.

[3] Hydraulic fracturing water consumption estimates followed the same general pattern as the water use estimates presented here, but with slightly larger percentages in each category (Section 4.4 in Chapter 4).

[4] County-level water availability estimates were derived from the Tidwell et al. (2013) estimates of water availability for siting new thermoelectric power plants (see Text Box 4-2 in Chapter 4 for details). The county-level water availability estimates used in this report represent the portion of water available to new users within a county.

Text Box ES-5. County-Level Water Use for Hydraulic Fracturing.

To assess whether hydraulic fracturing operations are a relatively large or small user of water, the average annual water use for hydraulic fracturing in 2011 and 2012 was compared, at the county-level, to total water use in 2010. For most counties studied, average annual water volumes reported for individual counties in FracFocus 1.0 were less than 1% of total water use in those counties. But in some counties, hydraulic fracturing operations reported in FracFocus 1.0 represented a relatively large user of water.



annual water volumes reported for hydraulic fracturing were less than 1% of the estimated annual volume of readily-available fresh water. However, average annual water volumes reported for hydraulic fracturing were greater than the estimated annual volume of readily-available fresh water in 17 counties in Texas. This analysis suggests that there was enough water available annually to support the level of hydraulic fracturing reported to FracFocus 1.0 in most, but not all, areas of the country. This observation does not preclude the possibility of local impacts in other areas of the country, nor does it indicate that local impacts have occurred or will occur in the 17 counties in Texas. To better understand whether local impacts have occurred, and the factors that affect those impacts, local-level studies, such as the ones described below, are needed.

Local impacts on drinking water quantity have occurred in areas with increased hydraulic fracturing activity. In 2011, for example, drinking water wells in an area overlying the Haynesville Shale ran out of water due to higher than normal groundwater withdrawals and drought (LA Ground Water Resources Commission, 2012). Water withdrawals for hydraulic fracturing contributed to these conditions, along with other water users and the lack of precipitation. Groundwater impacts have also been reported in Texas. In a detailed case study, Scanlon et al. (2014b) estimated that groundwater levels in approximately 6% of the area studied dropped by 100 feet (31 meters) to 200 feet (61 meters) or more after hydraulic fracturing activity increased in 2009.

In contrast, studies in the Upper Colorado and Susquehanna River basins found minimal impacts on drinking water resources from hydraulic fracturing. In the Upper Colorado River Basin, the EPA found that high-quality water produced from oil and gas wells in the Piceance tight sands provided nearly all of the water for hydraulic fracturing in the study area (U.S. EPA, 2015e). Due to this high reuse rate, the EPA did not identify any locations in the study area where hydraulic fracturing contributed to locally high water use. In the Susquehanna River Basin, multiple studies and state reports have identified the potential for hydraulic fracturing water withdrawals in the Marcellus Shale to impact surface water resources. Evidence suggests, however, that current water management strategies, including passby flows and reuse of hydraulic fracturing wastewater, help protect streams from depletion by hydraulic fracturing water withdrawals. A passby flow is a prescribed, low-streamflow threshold below which water withdrawals are not allowed.

The above examples highlight factors that can affect the frequency or severity of impacts on drinking water resources from hydraulic fracturing water withdrawals. In particular, areas of the United States that rely on declining groundwater resources are vulnerable to more frequent and more severe impacts from all water withdrawals, including withdrawals for hydraulic fracturing. Extensive groundwater withdrawals can limit the availability of belowground drinking water resources and can also change the quality of the water remaining in the resource. Because groundwater recharge rates can be low, impacts can last for many years. Seasonal or long-term drought can also make impacts more frequent and more severe for groundwater and surface water resources. Hot, dry weather reduces or prevents groundwater recharge and depletes surface water bodies, while water demand often increases simultaneously (e.g., for irrigation). This combination of factors—high hydraulic fracturing water use and relatively low water availability due to declining groundwater resources and/or frequent drought—was found to be present in southern and western Texas.

Water management strategies can also affect the frequency and severity of impacts on drinking water resources from hydraulic fracturing water withdrawals. These strategies include using hydraulic fracturing wastewater or brackish groundwater for hydraulic fracturing, transitioning from limited groundwater resources to more abundant surface water resources, and using passby flows to control water withdrawals from surface water resources. Examples of these water management strategies can be found throughout the United States. In western and southern Texas, for example, the use of brackish water is currently reducing impacts on fresh water sources, and could, if increased, reduce future impacts. Louisiana and North Dakota have encouraged well operators to withdraw water from surface water resources instead of high-quality groundwater resources. And, as described above, the Susquehanna River Basin Commission limits surface water withdrawals during periods of low stream flow.

Water Acquisition Conclusions

With notable exceptions, hydraulic fracturing uses a relatively small percentage of water when compared to total water use and availability at large geographic scales. Despite this, hydraulic fracturing water withdrawals can affect the quantity and quality of drinking water resources by changing the balance between the demand on local water resources and the availability of those resources. Changes that have the potential to limit the availability of drinking water or alter its quality are more likely to occur in areas with relatively high hydraulic fracturing water withdrawals and low water availability, particularly due to limited or declining groundwater resources. Water management strategies (e.g., encouragement of alternative water sources or water withdrawal restrictions) can reduce the frequency or severity of impacts on drinking water resources from hydraulic fracturing water withdrawals.

Chemical Mixing

Activity: The mixing of a base fluid, proppant, and additives at the well site to create hydraulic fracturing fluids.

Relationship to Drinking Water Resources: Spills of additives and hydraulic fracturing fluids can reach groundwater and surface water resources.

Hydraulic fracturing fluids are engineered to create and grow fractures in the targeted rock formation and to carry proppant through the oil and gas production well into the newly-created fractures. Hydraulic fracturing fluids are typically made up of base fluids, proppant, and additives. Base fluids make up the largest proportion of hydraulic fracturing fluids by volume. As illustrated in Text Box ES-6, base fluids can be a single substance (e.g., water in the slickwater example) or can be a mixture of substances (e.g., water and nitrogen in the energized fluid example). The EPA's analysis of hydraulic fracturing fluid data reported to FracFocus 1.0 suggests that water was the most commonly used base fluid between January 2011 and February 2013 (U.S. EPA, 2015a). Non-water substances, such as gases and hydrocarbon liquids, were reported to be used alone or blended with water to form a base fluid in fewer than 3% of wells in FracFocus 1.0.

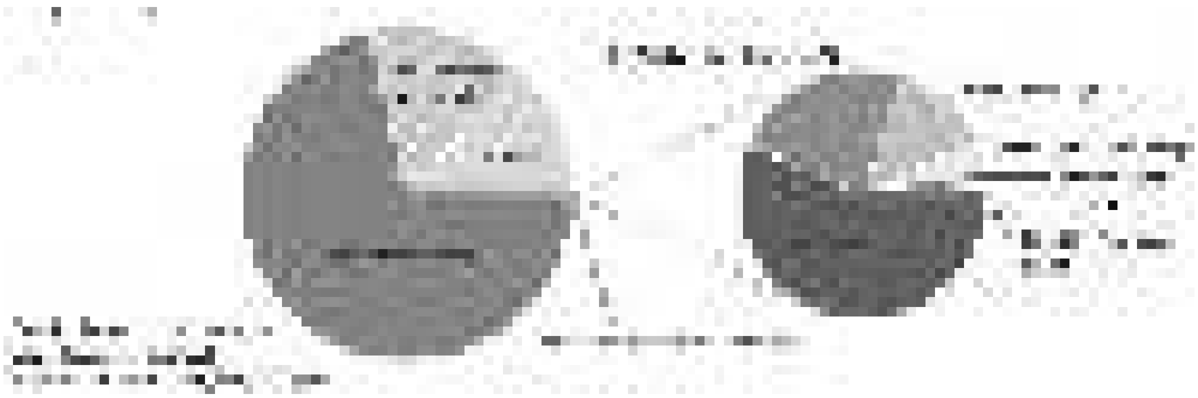
Proppant makes up the second largest proportion of hydraulic fracturing fluids (Text Box ES-6). Sand (i.e., quartz) was the most commonly reported proppant between January 2011 and February 2013, with 98% of wells in FracFocus 1.0 reporting sand as the proppant (U.S. EPA, 2015a). Other

Text Box ES-6. Examples of Hydraulic Fracturing Fluids.

Hydraulic fracturing fluids are engineered to create and extend fractures in the targeted rock formation and to carry proppant through the production well into the newly-created fractures. While there is no universal hydraulic fracturing fluid, there are general types of hydraulic fracturing fluids. Two types of hydraulic fracturing fluids are described below.

Slickwater

Slickwater hydraulic fracturing fluids are water-based fluids that generally contain a friction reducer. The friction reducer makes it easier for the fluid to be pumped down the oil and gas production well at high rates. Slickwater is commonly used to hydraulically fracture shale formations.



Energized Fluid

Energized fluids are mixtures of liquids and gases. They can be used for hydraulic fracturing in under-pressured gas formations.

proppants can include man-made or specially engineered particles, such as high-strength ceramic materials or sintered bauxite.[1]

Additives generally make up the smallest proportion of the overall composition of hydraulic fracturing fluids (Text Box ES-6), yet have the greatest potential to impact the quality of drinking water resources compared to proppant and base fluids. Additives, which can be a single chemical or a mixture of chemicals, are added to the base fluid to change its properties (e.g., adjust pH, increase fluid thickness, or limit bacterial growth). The choice of which additives to use depends on the characteristics of the targeted rock formation (e.g., rock type, temperature, and pressure), the economics and availability of desired additives, and well operator or service company preferences and experience.

The variability of additives, both in their purpose and chemical composition, suggests that a large number of different chemicals may be used in hydraulic fracturing fluids across the United States. The EPA identified 1,084 chemicals that were reported to have been used in hydraulic fracturing fluids between 2005 and 2013.[2,3] The EPA's analysis of FracFocus 1.0 data indicates that between 4 and 28 chemicals were used per well between January 2011 and February 2013 and that no single chemical was used in all wells (U.S. EPA, 2015a). Three chemicals—methanol, hydrotreated light petroleum distillates, and hydrochloric acid—were reported in 65% or more of the wells in FracFocus 1.0; 35 chemicals were reported in at least 10% of the wells (Table ES-2).

Table ES-2. Chemicals reported in 10% or more of disclosures in FracFocus 1.0.

Disclosures provided information on chemicals used at individual well sites between January 1, 2011, and February 28, 2013.

Chemical Name (CASRN)[a]	Percent of FracFocus 1.0 disclosures[b]
Methanol (67-56-1)	72
Hydrotreated light petroleum distillates (64742-47-8)	65
Hydrochloric acid (7647-01-0)	65
Water (7732-18-5)[c]	48
Isopropanol (67-63-0)	47
Ethylene glycol (107-21-1)	46
Peroxydisulfuric acid, diammonium salt (7727-54-0)	44
Sodium hydroxide (1310-73-2)	39
Guar gum (9000-30-0)	37

[1] Sintered bauxite is crushed and powdered bauxite that is fused into spherical beads at high temperatures.

[2] This list includes 1,084 unique Chemical Abstracts Service Registration Numbers (CASRNs), which can be assigned to a single chemical (e.g., hydrochloric acid) or a mixture of chemicals (e.g., hydrotreated light petroleum distillates). Throughout this report, we refer to the substances identified by unique CASRNs as "chemicals."

[3] Dayalu and Konschnik (2016) identified 995 unique CASRNs from data submitted to FracFocus between March 9, 2011, and April 13, 2015. Two hundred sixty-three of these CASRNs are not on the list of unique CASRNs identified by the EPA (Appendix H). Only one of the 263 chemicals was reported at greater than 1% of wells, which suggests that these chemicals were used at only a few sites.

Chemical Name (CASRN)[a]	Percent of FracFocus 1.0 disclosures[b]
Quartz (14808-60-7)[c]	36
Glutaraldehyde (111-30-8)	34
Propargyl alcohol (107-19-7)	33
Potassium hydroxide (1310-58-3)	29
Ethanol (64-17-5)	29
Acetic acid (64-19-7)	24
Citric acid (77-92-9)	24
2-Butoxyethanol (111-76-2)	21
Sodium chloride (7647-14-5)	21
Solvent naphtha, petroleum, heavy aromatic (64742-94-5)	21
Naphthalene (91-20-3)	19
2,2-Dibromo-3-nitrilopropionamide (10222-01-2)	16
Phenolic resin (9003-35-4)	14
Choline chloride (67-48-1)	14
Methenamine (100-97-0)	14
Carbonic acid, dipotassium salt (584-08-7)	13
1,2,4-Trimethylbenzene (95-63-6)	13
Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides (68424-85-1)	12
Poly(oxy-1,2-ethanediyl)-nonylphenyl- hydroxy (mixture) (127087-87-0)	12
Formic acid (64-18-6)	12
Sodium chlorite (7758-19-2)	11
Nonyl phenol ethoxylate (9016-45-9)	11
Tetrakis(hydroxymethyl)phosphonium sulfate (55566-30-8)	11
Polyethylene glycol (25322-68-3)	11
Ammonium chloride (12125-02-9)	10
Sodium persulfate (7775-27-1)	10

[a] "Chemical" refers to chemical substances with a single CASRN; these may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates).

[b] Analysis considered 34,675 disclosures that met selected quality assurance criteria. See Table 5-2 in Chapter 5.

[c] Quartz and water were reported as ingredients in additives, in addition to proppants and base fluids.

Concentrated additives are delivered to the well site and stored until they are mixed with the base fluid and proppant and pumped down the oil and gas production well (Text Box ES-7). While the overall concentration of additives in hydraulic fracturing fluids is generally small (typically 2% or less of the total volume of the injected fluid), the total volume of additives delivered to the well site can be large. Because over 1 million gallons (3.8 million liters) of hydraulic fracturing fluid are generally injected per well, thousands of gallons of additives can be stored on site and used during hydraulic fracturing.

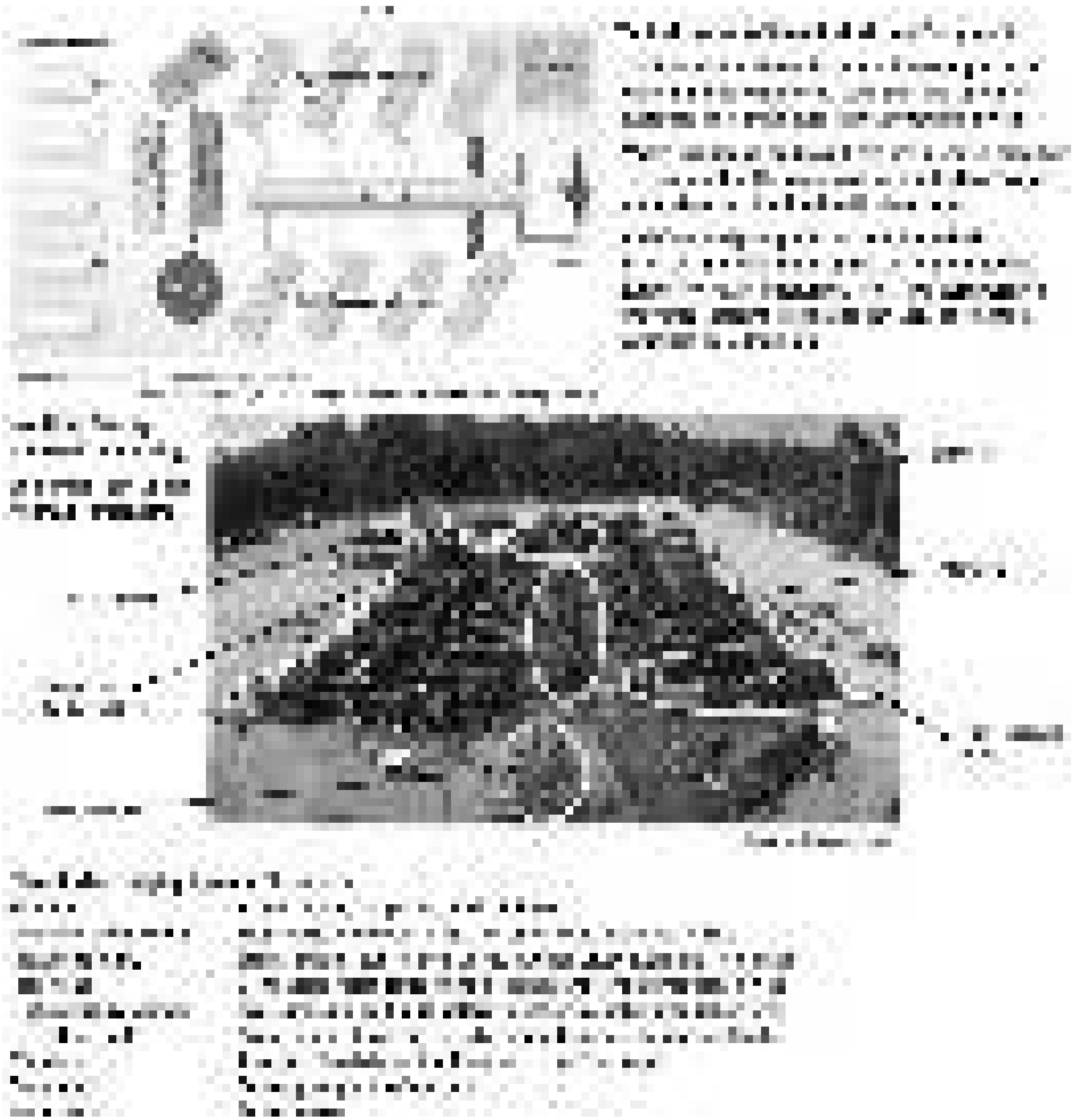
As illustrated in Text Box ES-7, additives are often stored in multiple, closed containers [typically 200 gallons (760 liters) to 375 gallons (1,420 liters) per container] and moved around the site in hoses and tubing. This equipment is designed to contain additives and blended hydraulic fracturing fluid, but spills can occur. Changes in drinking water quality can occur if spilled fluids reach groundwater or surface water resources.

Several studies have documented spills of hydraulic fracturing fluids or additives. Nearly all of these studies identified spills from state-managed spill databases. Data gathered for these studies suggest that spills of hydraulic fracturing fluids or additives were primarily caused by equipment failure or human error. For example, an EPA analysis of spill reports from nine state agencies, nine oil and gas well operators, and nine hydraulic fracturing service companies characterized 151 spills of hydraulic fracturing fluids or additives on or near well sites in 11 states between January 2006 and April 2012 (U.S. EPA, 2015m). These spills were primarily caused by equipment failure (34% of the spills) or human error (25%), and more than 30% of the spills were from fluid storage units (e.g., tanks, totes, and trailers). Similarly, a study of spills reported to the Colorado Oil and Gas Conservation Commission identified 125 spills during well stimulation (i.e., a part of the life of an oil and gas well that often, but not always, includes hydraulic fracturing) between January 2010 and August 2013 (COGCC, 2014). Of these spills, 51% were caused by human error and 46% were due to equipment failure.

Studies of spills of hydraulic fracturing fluids or additives provide insights on spill volumes, but little information on chemical-specific spill composition. Among the 151 spills characterized by the EPA, the median volume of fluid spilled was 420 gallons (1,600 liters), although the volumes spilled ranged from 5 gallons (19 liters) to 19,320 gallons (73,130 liters). Spilled fluids were often described as acids, biocides, friction reducers, crosslinkers, gels, and blended hydraulic fracturing fluid, but few specific chemicals were mentioned.[1] Considine et al. (2012) identified spills related to oil and gas development in the Marcellus Shale that occurred between January 2008 and August 2011 from Notices of Violations issued by the Pennsylvania Department of Environmental Protection. The authors identified spills greater than 400 gallons (1,500 liters) and spills less than 400 gallons (1,500 liters).

[1] A crosslinker is an additive that increases the thickness of gelled fluids by connecting polymer molecules in the gelled fluid.

Text Box ES-7. Chemical Mixing Equipment.



Spills of hydraulic fracturing fluids or additives have reached, and therefore impacted, surface water resources. Thirteen of the 151 spills characterized by the EPA were reported to have reached a surface water body (often creeks or streams). Among the 13 spills, reported spill volumes ranged from 28 gallons (105 liters) to 7,350 gallons (27,800 liters). Additionally, Brantley et al. (2014) and Considine et al. (2012) identified fewer than 10 total instances of spills of additives and/or hydraulic fracturing fluids greater than 400 gallons (1,500 liters) that reached surface waters in

Pennsylvania between January 2008 and June 2013. Reported spill volumes for these spills ranged from 3,400 gallons (13,000 liters) to 227,000 gallons (859,000 liters).

Although impacts on surface water resources have been documented, site-specific studies that could be used to describe factors that affect the frequency or severity of impacts were not available. In the absence of such studies, we relied on fundamental scientific principles to identify factors that affect how hydraulic fracturing fluids and chemicals can move through the environment to drinking water resources. Because these factors influence whether spilled fluids reach groundwater and surface water resources, they affect the frequency and severity of impacts on drinking water resources from spills during the chemical mixing stage of the hydraulic fracturing water cycle.

The potential for spilled fluids to impact groundwater or surface water resources depends on the characteristics of the spill, the environmental fate and transport of the spilled fluid, and spill response activities (Figure ES-5). Site-specific characteristics affect how spilled liquids move through soil into the subsurface or over the land surface. Generally, highly permeable soils or fractured rock can allow spilled liquids to move quickly into and through the subsurface, limiting the opportunity for spilled liquids to move over land to surface water resources. In low permeability soils, spilled liquids are less able to move into the subsurface and are more likely to move over the land surface. In either case, the volume spilled and the distance between the location of the spill and nearby water resources affects whether spilled liquids reach drinking water resources. Large-volume spills are generally more likely to reach drinking water resources because they are more likely to be able to travel the distance between the location of the spill and nearby water resources.

In general, chemical and physical properties, which depend on the identity and structure of a chemical, control whether spilled chemicals evaporate, stick to soil particles, or move with water. The EPA identified measured or estimated chemical and physical properties for 455 of the 1,084 chemicals used in hydraulic fracturing fluids between 2005 and 2013.[1] The properties of these chemicals varied widely, from chemicals that are more likely to move quickly through the environment with a spilled liquid to chemicals that are more likely to move slowly through the environment because they stick to soil particles.[2] Chemicals that move slowly through the environment may act as longer-term sources of contamination if spilled.

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[1] Chemical and physical properties were identified using EPI Suite™. EPI Suite™ is a collection of chemical and physical property and environmental fate estimation programs developed by the EPA and Syracuse Research Corporation. It can be used to estimate chemical and physical properties of individual organic compounds. Of the 1,084 hydraulic fracturing fluid chemicals identified by the EPA, 629 were not individual organic compounds, and thus EPI Suite™ could not be used to estimate their chemical and physical properties.

[2] These results describe how some hydraulic fracturing chemicals behave in infinitely dilute aqueous solutions, which is a simplified approximation of the real-world mixtures found in hydraulic fracturing fluids. The presence of other chemicals in a mixture can affect the fate and transport of a chemical.

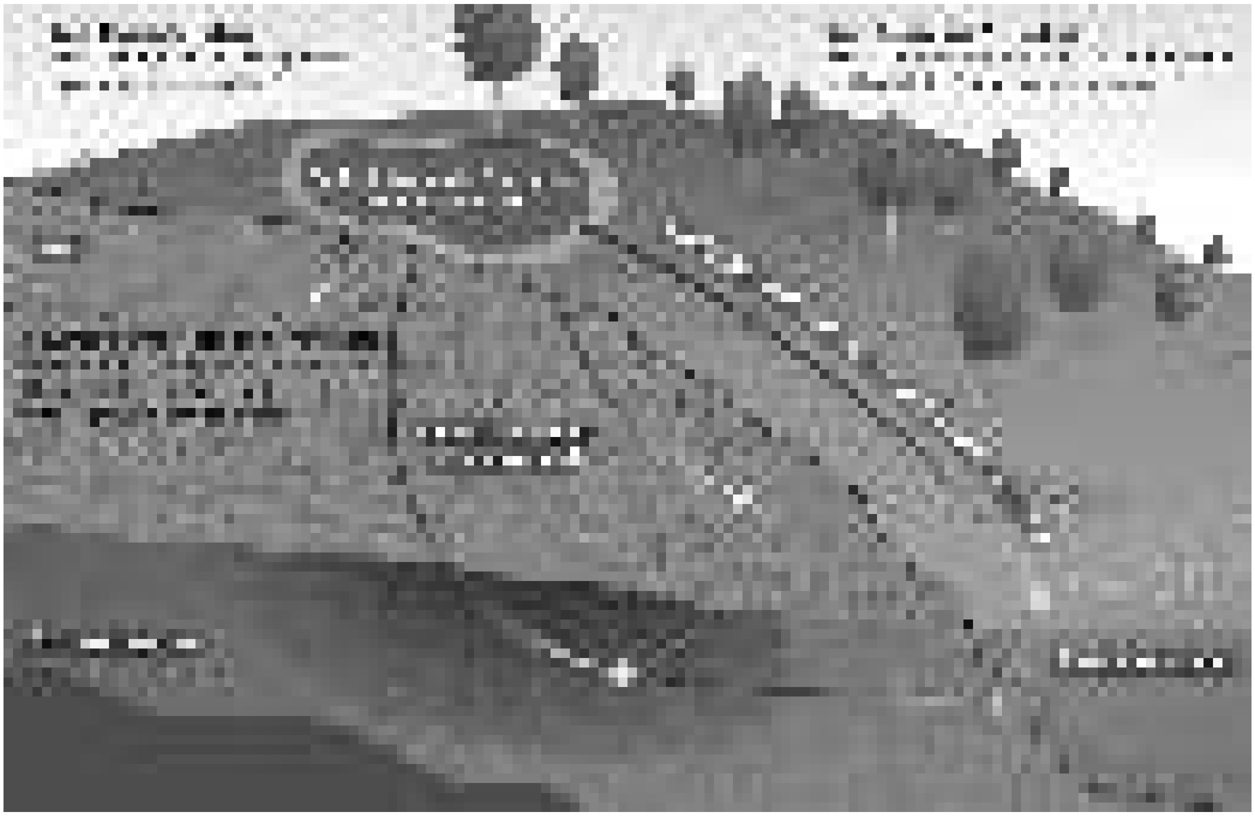

Figure ES-5. Generalized depiction of factors that influence whether spilled hydraulic fracturing fluids or additives reach drinking water resources, including spill characteristics, environmental fate and transport, and spill response activities.

Spill prevention practices and spill response activities are designed to prevent spilled fluids from reaching groundwater or surface water resources and minimize impacts from spilled fluids. Spill prevention and response activities are influenced by federal, state, and local regulations and company practices. Spill prevention practices include secondary containment systems (e.g., liners and berms), which are designed to contain spilled fluids and prevent them from reaching soil, groundwater, or surface water. Spill response activities include activities taken to stop the spill, contain spilled fluids (e.g., the deployment of emergency containment systems), and clean up spilled fluids (e.g., removal of contaminated soil). It was beyond the scope of this report to evaluate the implementation and efficacy of spill prevention practices and spill response activities.

The severity of impacts on water quality from spills of hydraulic fracturing fluids or additives depends on the identity and amount of chemicals that reach groundwater or surface water resources, the toxicity of the chemicals, and the characteristics of the receiving water resource.[1] Characteristics of the receiving groundwater or surface water resource (e.g., water resource size and flow rate) can affect the magnitude and duration of impacts by reducing the concentration of spilled chemicals in a drinking water resource. Impacts on groundwater resources have the

[1] Human health hazards associated with hydraulic fracturing fluid chemicals are discussed in Chapter 9 and summarized in the "Chemicals in the Hydraulic Fracturing Water Cycle" section below.

potential to be more severe than impacts on surface water resources because it takes longer to naturally reduce the concentration of chemicals in groundwater and because it is generally difficult to remove chemicals from groundwater resources. Due to a lack of data, particularly in terms of groundwater monitoring after spill events, little is publicly known about the severity of drinking water impacts from spills of hydraulic fracturing fluids or additives.

Chemical Mixing Conclusions

Spills of hydraulic fracturing fluids and additives during the chemical mixing stage of the hydraulic fracturing water cycle have reached surface water resources in some cases and have the potential to reach groundwater resources. Although the available data indicate that spills of various volumes can reach surface water resources, large volume spills are more likely to travel longer distances to nearby groundwater or surface water resources. Consequently, large volume spills likely increase the frequency of impacts on drinking water resources. Large volume spills, particularly of concentrated additives, are also likely to result in more severe impacts on drinking water resources than small volume spills because they can deliver a large quantity of potentially hazardous chemicals to groundwater or surface water resources. Impacts on groundwater resources are likely to be more severe than impacts on surface water resources because of the inherent characteristics of groundwater. Spill prevention and response activities are designed to prevent spilled fluids from reaching groundwater or surface water resources and minimize impacts from spilled fluids.

Well Injection

Activity: The injection and movement of hydraulic fracturing fluids through the oil and gas production well and in the targeted rock formation.

Relationship to Drinking Water Resources: Belowground pathways, including the production well itself and newly-created fractures, can allow hydraulic fracturing fluids or other fluids to reach underground drinking water resources.

Hydraulic fracturing fluids primarily move along two pathways during the well injection stage: the oil and gas production well and the newly-created fracture network. Oil and gas production wells are designed and constructed to move fluids to and from the targeted rock formation without leaking and to prevent fluid movement along the outside of the well. This is generally accomplished by installing multiple layers of casing and cement within the drilled hole (Text Box ES-2), particularly where the well intersects oil-, gas-, and/or water-bearing rock formations. Casing and cement, in addition to other well components (e.g., packers), can control hydraulic fracturing fluid movement by creating a preferred flow pathway (i.e., inside the casing) and preventing unintentional fluid movement (e.g., from the inside of the casing to the surrounding environment or vertically along the well from the targeted rock formation to shallower formations).[1] An EPA survey of oil and gas production wells hydraulically fractured between approximately September 2009 and September 2010 suggests that hydraulically fractured wells are often, but not always, constructed

[1] Packers are mechanical devices installed with casing. Once the casing is set in the drilled hole, packers swell to fill the space between the outside of the casing and the surrounding rock or casing.

with multiple casings that have varying amounts of cement surrounding each casing (U.S. EPA, 2015n). Among the wells surveyed, the most common number of casings per well was two: surface casing and production casing (Text Box ES-2). The presence of multiple cemented casings that extend from the ground surface to below the designated drinking water resource is one of the primary well construction features that protects underground drinking water resources.

During hydraulic fracturing, a well is subjected to greater pressure and temperature changes than during any other activity in the life of the well. As hydraulic fracturing fluid is injected into the well, the pressure applied to the well increases until the targeted rock formation fractures; then pressure decreases. Maximum pressures applied to wells during hydraulic fracturing have been reported to range from less than 2,000 pounds per square inch (psi) [14 megapascals (MPa)] to approximately 12,000 psi (83 MPa).[1] A well can also experience temperature changes as cooler hydraulic fracturing fluid enters the warmer well. In some cases, casing temperatures have been observed to drop from 212°F (100°C) to 64°F (18°C). A well can experience multiple pressure and temperature cycles if hydraulic fracturing is done in multiple stages or if a well is re-fractured.[2] Casing, cement, and other well components need to be able to withstand these changes in pressure and temperature, so that hydraulic fracturing fluids can flow to the targeted rock formation without leaking.

The fracture network created during hydraulic fracturing is the other primary pathway along which hydraulic fracturing fluids move. Fracture growth during hydraulic fracturing is complex and depends on the characteristics of the targeted rock formation and the characteristics of the hydraulic fracturing operation. In general, rock characteristics, particularly the natural stresses placed on the targeted rock formation due to the weight of the rock above, affect how the rock fractures, including whether newly-created fractures grow vertically (i.e., perpendicular to the ground surface) or horizontally (i.e., parallel to the ground surface) (Text Box ES-8). Because hydraulic fracturing fluids are used to create and grow fractures, fracture growth during hydraulic fracturing can be controlled by limiting the rate and volume of hydraulic fracturing fluid injected into the well.

Publicly available data on fracture growth are currently limited to microseismic and tiltmeter data collected during hydraulic fracturing operations in five shale plays in the United States. Analyses of these data by Fisher and Warpinski (2012) and Davies et al. (2012) indicate that the direction of fracture growth generally varied with depth and that upward vertical fracture growth was often on the order of tens to hundreds of feet in the shale formations studied (Text Box ES-8). One percent of the fractures had a fracture height greater than 1,148 feet (350 meters), and the maximum fracture height among all of the data reported was 1,929 feet (588 meters). These reported fracture heights suggest that some fractures can grow out of the targeted rock formation and into an overlying formation. It is unknown whether these observations apply to other hydraulically fractured rock formations because similar data from hydraulic fracturing operations in other rock formations are not currently available to the public.

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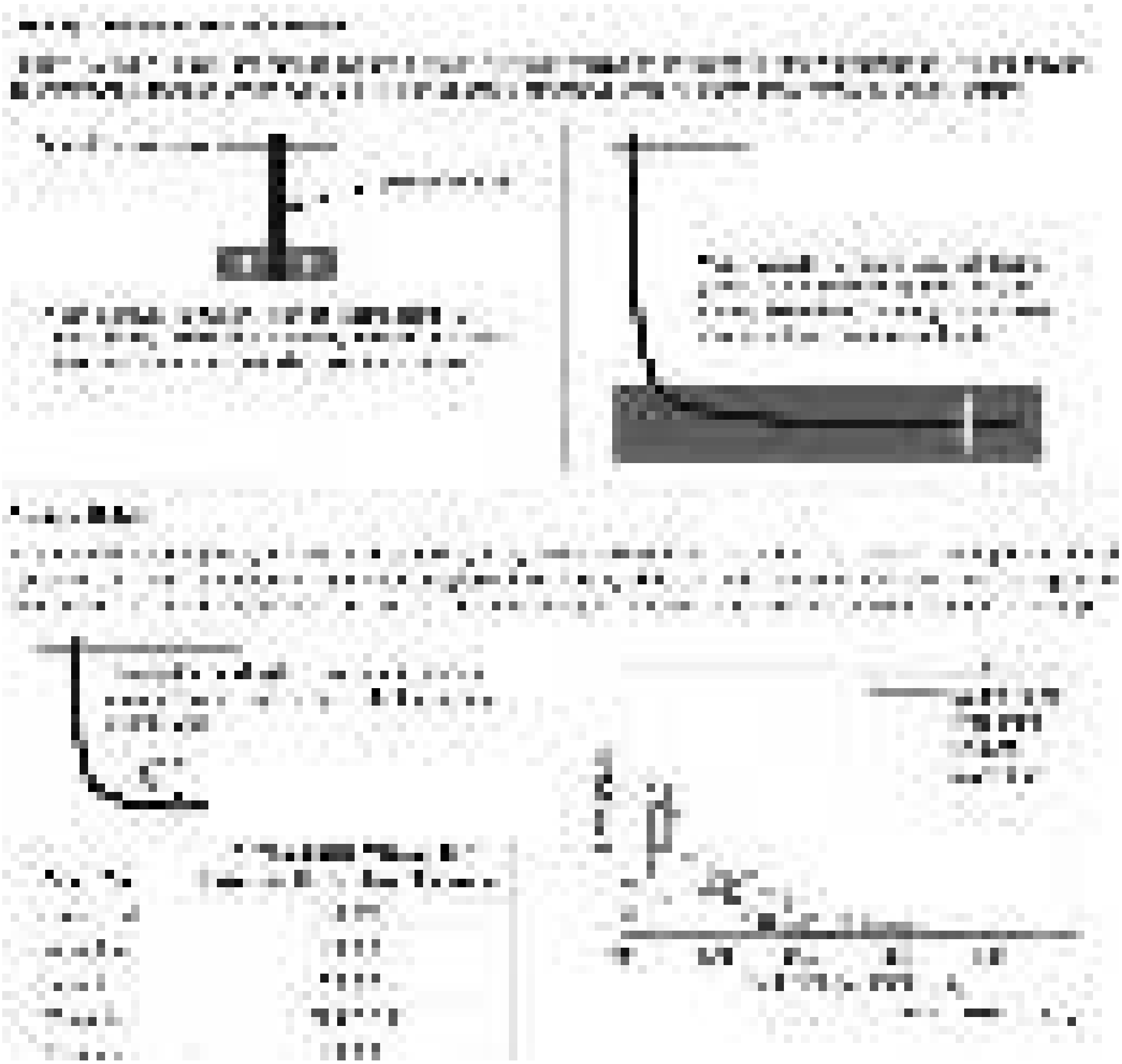
[1] For comparison, average atmospheric pressure is approximately 15 psi.

[2] In a multi-stage hydraulic fracturing operation, specific parts of the well are isolated and hydraulically fractured until the total desired length of the well has been hydraulically fractured.

Text Box ES-8. Fracture Growth.

Fracture growth during hydraulic fracturing is complex and depends on the characteristics of the targeted rock formation and the characteristics of the hydraulic fracturing operation.



The potential for hydraulic fracturing fluids to reach, and therefore impact, underground drinking water resources is related to the pathways along which hydraulic fracturing fluids primarily move during hydraulic fracturing: the oil and gas production well itself and the fracture network created during hydraulic fracturing. Because the well can be a pathway for fluid movement, the mechanical integrity of the well is an important factor that affects the frequency and severity of impacts from the well injection stage of the hydraulic fracturing water cycle.[1] A well with insufficient mechanical integrity can allow unintended fluid movement, either from the inside to the outside of the well (pathway 1 in Figure ES-6) or vertically along the outside of the well (pathways 2-5). The existence of one or more of these pathways can result in impacts on drinking water resources if hydraulic fracturing fluids reach groundwater resources. Impacts on drinking water resources can also occur if gases or liquids released from the targeted rock formation or other formations during hydraulic fracturing travel along these pathways to groundwater resources.

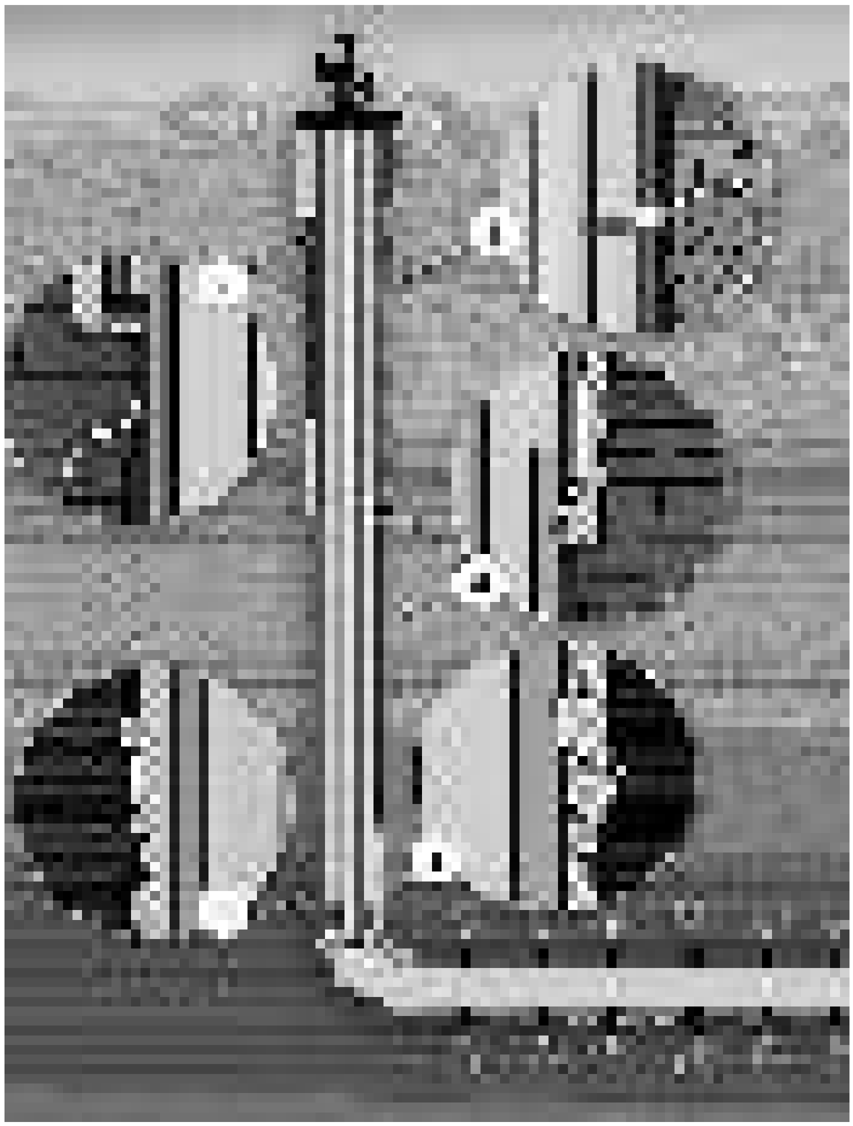


Figure ES-6. Potential pathways for fluid movement in a cemented well.
These pathways (represented by the white arrows) include: (1) a casing and tubing leak into the surrounding rock, (2) an uncemented annulus (i.e., the space behind the casing), (3) microannuli between the casing and cement, (4) gaps in cement due to poor cement quality, and (5) microannuli between the cement and the surrounding rock. This figure is intended to provide a conceptual illustration of pathways that can be present in a well and is not to scale.

[1] Mechanical integrity is the absence of significant leakage within or outside of the well components.

The pathways shown in Figure ES-6 can exist because of inadequate well design or construction (e.g., incomplete cement around the casing where the well intersects with water-, oil-, or gas-bearing formations) or can develop over the well's lifetime, including during hydraulic fracturing. In particular, casing and cement can degrade over the life of the well because of exposure to corrosive chemicals, formation stresses, and operational stresses (e.g., pressure and temperature changes during hydraulic fracturing). As a result, some hydraulically fractured oil and gas production wells may develop one or more of the pathways shown in Figure ES-6. Changes in mechanical integrity over time have implications for older wells that are hydraulically fractured because these wells may not be able to withstand the stresses applied during hydraulic fracturing. Older wells may also be hydraulically fractured at shallower depths, where cement around the casing may be inadequate or missing.

Examples of mechanical integrity problems have been documented in hydraulically fractured oil and gas production wells. In one case, hydraulic fracturing of an inadequately cemented gas well in Bainbridge Township, Ohio, contributed to the movement of methane into local drinking water resources.[1] In another case, an inner string of casing burst during hydraulic fracturing of an oil well near Killdeer, North Dakota, resulting in a release of hydraulic fracturing fluids and formation fluids that impacted a groundwater resource.

The potential for hydraulic fracturing fluids or other fluids to reach underground drinking water resources is also related to the fracture network created during hydraulic fracturing. Because fluids travel through the newly-created fractures, the location of these fractures relative to underground drinking water resources is an important factor affecting the frequency and severity of potential impacts on drinking water resources. Data on the relative location of induced fractures to underground drinking water resources are generally not available, because fracture networks are infrequently mapped and because there can be uncertainty in the depth of the bottom of the underground drinking water resource at a specific location.

Without these data, we were often unable to determine with certainty whether fractures created during hydraulic fracturing have reached underground drinking water resources. Instead, we considered the vertical separation distance between hydraulically fractured rock formations and the bottom of underground drinking water resources. Based on computer modeling studies, Birdsell et al. (2015a) concluded that it is less likely that hydraulic fracturing fluids would reach an overlying drinking water resource if (1) the vertical separation distance between the targeted rock formation and the drinking water resource is large and (2) there are no open pathways (e.g., natural faults or fractures, or leaky wells). As the vertical separation distance between the targeted rock formation and the underground drinking water resource decreases, the likelihood of upward migration of hydraulic fracturing fluids to the drinking water resource increases (Birdsell et al., 2015a).

Figure ES-7 illustrates how the vertical separation distance between the targeted rock formation and underground drinking water resources can vary across the United States. The two example

[1] Although ingestion of methane is not considered to be toxic, methane can pose a physical hazard. Methane can accumulate to explosive levels when allowed to exsolve (degas) from groundwater in closed environments.

environments depicted in panels a and b represent the range of separation distances shown in panel c. In Figure ES-7a, there are thousands of feet between the bottom of the underground drinking water resource and the hydraulically fractured rock formation. These conditions are generally reflective of deep shale formations (e.g., Haynesville Shale), where oil and gas production wells are first drilled vertically and then horizontally along the targeted rock formation. Microseismic data and modeling studies suggest that, under these conditions, fractures created during hydraulic fracturing are unlikely to grow through thousands of feet of rock into underground drinking water resources.

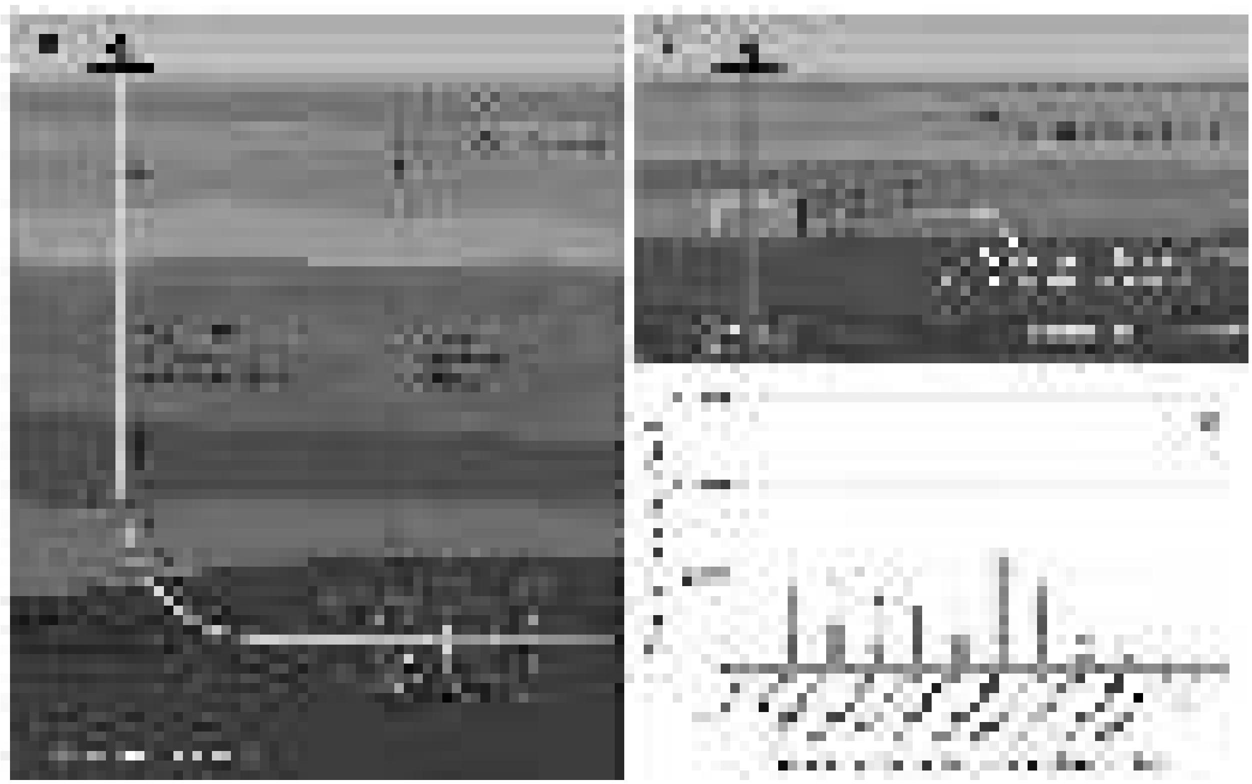


Figure ES-7. Examples of different subsurface environments in which hydraulic fracturing takes place.

In panel a, there are thousands of feet between the base of the underground drinking water resource and the part of the well that is hydraulically fractured. Panel b illustrates the co-location of groundwater and oil and gas resources. In these types of situations, there is no separation between the shallowest point of hydraulic fracturing within the well and the bottom of the underground drinking water resource. Panel c shows the estimated distribution of separation distances for approximately 23,000 oil and gas production wells hydraulically fractured by nine service companies between 2009 and 2019 (U.S. EPA, 2015n). The separation distance is the distance along the well between the point of shallowest hydraulic fracturing in the well and the base of the protected groundwater resource (illustrated in panel a). The error bars in panel c display 95% confidence intervals.

When drinking water resources are co-located with oil and gas resources and there is no vertical separation between the hydraulically fractured rock formation and the bottom of the underground drinking water resource (Figure ES-7b), the injection of hydraulic fracturing fluids impacts the quality of the drinking water resource. According to the information examined in this report, the overall occurrence of hydraulic fracturing within a drinking water resource appears to be low, with the activity generally concentrated in some areas in the western United States (e.g., the Wind River Basin near Pavillion, Wyoming, and the Powder River Basin of Montana and Wyoming).[1] Hydraulic fracturing within drinking water resources introduces hydraulic fracturing fluid into formations that may currently serve, or in the future could serve, as a drinking water source for public or private use. This is of concern in the short-term if people are currently using these formations as a drinking water supply. It is also of concern in the long-term, because drought or other conditions may necessitate the future use of these formations for drinking water.

Regardless of the vertical separation between the targeted rock formation and the underground drinking water resource, the presence of other wells near hydraulic fracturing operations can increase the potential for hydraulic fracturing fluids or other subsurface fluids to move to drinking water resources. There have been cases in which hydraulic fracturing at one well has affected a nearby oil and gas well or its fracture network, resulting in unexpected pressure increases at the nearby well, damage to the nearby well, or spills at the surface of the nearby well. These well communication events, or "frac hits," have been reported in New Mexico, Oklahoma, and other locations. Based on the available information, frac hits most commonly occur when multiple wells are drilled from the same surface location and when wells are spaced less than 1,100 feet (335 meters) apart. Frac hits have also been observed at wells up to 8,422 feet (2,567 meters) away from a well undergoing hydraulic fracturing.

Abandoned wells near a well undergoing hydraulic fracturing can provide a pathway for vertical fluid movement to drinking water resources if those wells were not properly plugged or if the plugs and cement have degraded over time. For example, an abandoned well in Pennsylvania produced a 30-foot (9-meter) geyser of brine and gas for more than a week after hydraulic fracturing of a nearby gas well. The potential for fluid movement along abandoned wells may be a significant issue in areas with historic oil and gas exploration and production. Various studies estimate the number of abandoned wells in the United States to be significant. For instance, the Interstate Oil and Gas Compact Commission estimates that over 1 million wells were drilled in the United States prior to the enactment of state oil and gas regulations (IOGCC, 2008). The location and condition of many of these wells are unknown, and some states have programs to find and plug abandoned wells.

Well Injection Conclusions

Impacts on drinking water resources associated with the well injection stage of the hydraulic fracturing water cycle have occurred in some instances. In particular, mechanical integrity failures have allowed gases or liquids to move to underground drinking water resources. Additionally, hydraulic fracturing has occurred within underground drinking water resources in parts of the United States. This practice introduces hydraulic fracturing fluids into underground drinking water resources. Consequently, the mechanical integrity of the well and the vertical separation distance between the targeted rock formation and underground drinking water resources are important factors that affect the frequency and severity of impacts on drinking water resources. The presence of multiple layers of cemented casing and thousands of feet of rock between hydraulically fractured

[1] Section 6.3.2 in Chapter 6.

rock formations and underground drinking water resources can reduce the frequency of impacts on drinking water resources during the well injection stage of the hydraulic fracturing water cycle.

Produced Water Handling

Activity: The on-site collection and handling of water that returns to the surface after hydraulic fracturing and the transportation of that water for disposal or reuse.

Relationship to Drinking Water Resources: Spills of produced water can reach groundwater and surface water resources.

After hydraulic fracturing, the injection pressure applied to the oil or gas production well is released, and the direction of fluid flow reverses, causing fluid to flow out of the well. The fluid that initially returns to the surface after hydraulic fracturing is mostly hydraulic fracturing fluid and is sometimes called "flowback" (Text Box ES-9). As time goes on, the fluid that returns to the surface contains water and economic quantities of oil and/or gas that are separated and collected. Water that returns to the surface during oil and gas production is similar in composition to the fluid naturally found in the targeted rock formation and is typically called "produced water." The term "produced water" is also used to refer to any water, including flowback, that returns to the surface through the production well as a by-product of oil and gas production. This latter definition of "produced water" is used in this report.

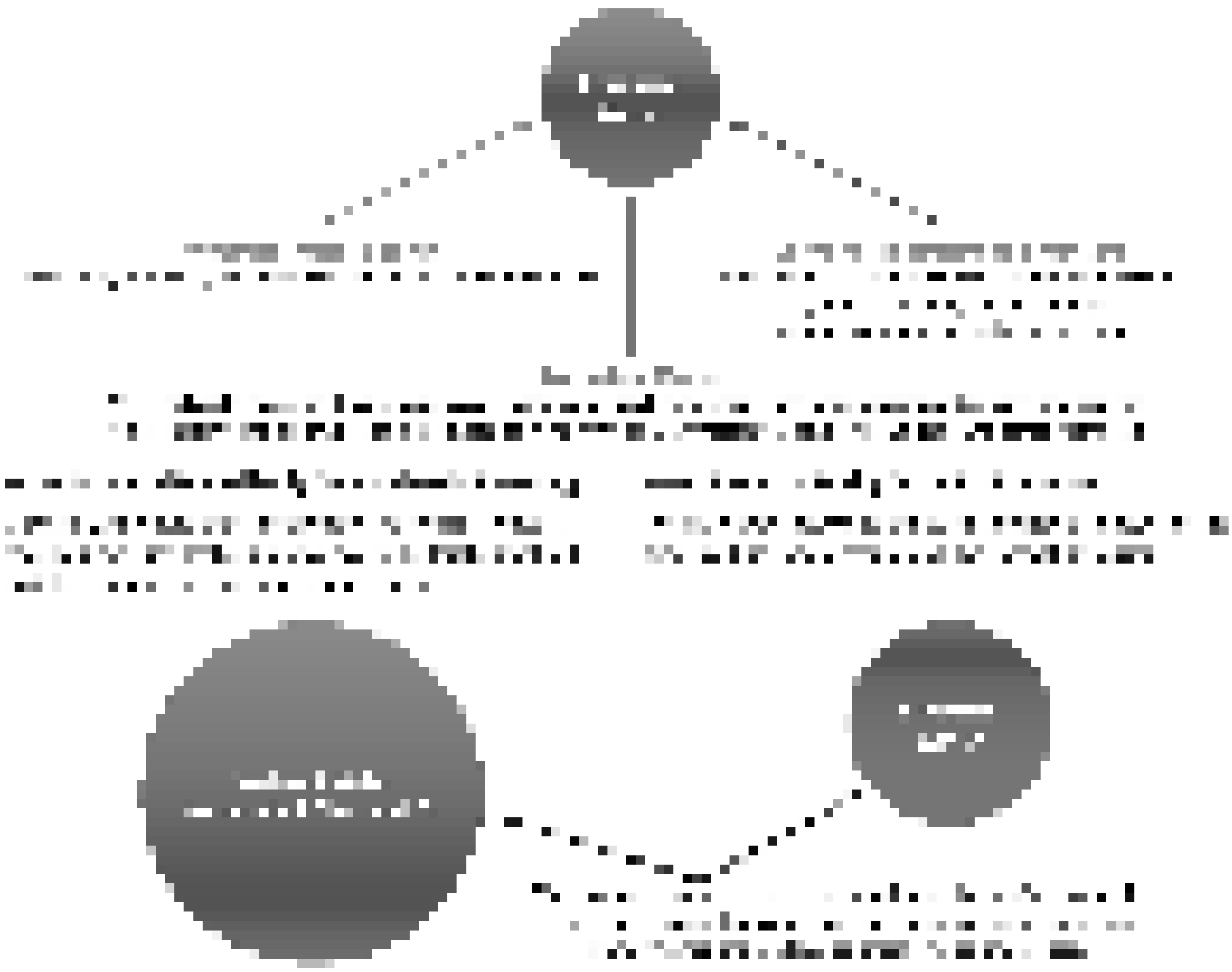
Produced water can contain many constituents, depending on the composition of the injected hydraulic fracturing fluid and the type of rock hydraulically fractured. Knowledge of the chemical composition of produced water comes from the collection and analysis of produced water samples, which often requires advanced laboratory equipment and techniques that can detect and quantify chemicals in produced water. In general, produced water has been found to contain:

- Salts, including those composed from chloride, bromide, sulfate, sodium, magnesium, and calcium;
- Metals, including barium, manganese, iron, and strontium;
- Naturally-occurring organic compounds, including benzene, toluene, ethylbenzene, xylenes (BTEX), and oil and grease;
- Radioactive materials, including radium; and
- Hydraulic fracturing chemicals and their chemical transformation products.

The amount of these constituents in produced water varies across the United States, both within and among different rock formations. Produced water from shale and tight gas formations is typically very salty compared to produced water from coalbed methane formations. For example, the salinity of produced water from the Marcellus Shale has been reported to range from less than 1,500 milligrams per liter (mg/L) of total dissolved solids to over 300,000 mg/L, while produced water from coalbed methane formations has been reported to range from 170 mg/L of total

Text Box ES-9. Produced Water from Hydraulically Fractured Oil and Gas Production Wells.

Water of varying quality is a byproduct of oil and gas production. The composition and volume of produced water varies by well, rock formation, and time after hydraulic fracturing. Produced water can contain hydraulic fracturing fluid, formation water, and chemical transformation products.



dissolved solids to nearly 43,000 mg/L.[1] Shale and sandstone formations also commonly contain radioactive materials, including uranium, thorium, and radium. As a result, radioactive materials have been detected in produced water from these formations.

Produced water volumes can vary by well, rock formation, and time after hydraulic fracturing. Volumes are often described in terms of the volume of hydraulic fracturing fluid used to fracture the well. For example, Figure ES-4 shows that wells in the Marcellus Shale typically produce 10-30% of the volume injected in the first 10 years after hydraulic fracturing. In comparison, some wells in the Barnett Shale have produced 100% of the volume injected in the first three years.

[1] For comparison, the average salinity of seawater is approximately 35,000 mg/L of total dissolved solids.

Because of the large volumes used for hydraulic fracturing [about 4 million gallons (15 million liters) per well in the Marcellus Shale and the Barnett Shale], hundreds of thousands to millions of gallons of produced water need to be collected and handled at the well site. The volume of water produced per day generally decreases with time, so the volumes handled on site immediately after hydraulic fracturing can be much larger than the volumes handled when the well is producing oil and/or gas (Text Box ES-9).

Produced water flows from the well to on-site tanks or pits through a series of pipes or flowlines (Text Box ES-10) before being transported offsite via trucks or pipelines for disposal or reuse. While produced water collection, storage, and transportation systems are designed to contain produced water, spills can occur. Changes in drinking water quality can occur if produced water spills reach groundwater or surface water resources.

Produced water spills have been reported across the United States. Median spill volumes among the datasets reviewed for this report ranged from approximately 340 gallons (1,300 liters) to 1,000 gallons (3,800 liters) per spill.[1] There were, however, a small number of large volume spills. In North Dakota, for example, there were 12 spills greater than 21,000 gallons (79,500 liters), five spills greater than 42,000 gallons (160,000 liters), and one spill of 2.9 million gallons (11 million liters) in 2015. Common causes of produced water spills included human error and equipment leaks or failures. Common sources of produced water spills included hoses or lines and storage equipment.

Spills of produced water have reached groundwater and surface water resources. In U.S. EPA (2015m), 30 of the 225 (13%) produced water spills characterized were reported to have reached surface water (e.g., creeks, ponds, or wetlands), and one was reported to have reached groundwater. Of the spills that were reported to have reached surface water, reported spill volumes ranged from less than 170 gallons (640 liters) to almost 74,000 gallons (280,000 liters). A separate assessment of produced water spills reported to the California Office of Emergency Services between January 2009 and December 2014 reported that 18% of the spills impacted waterways (CCST, 2015a).

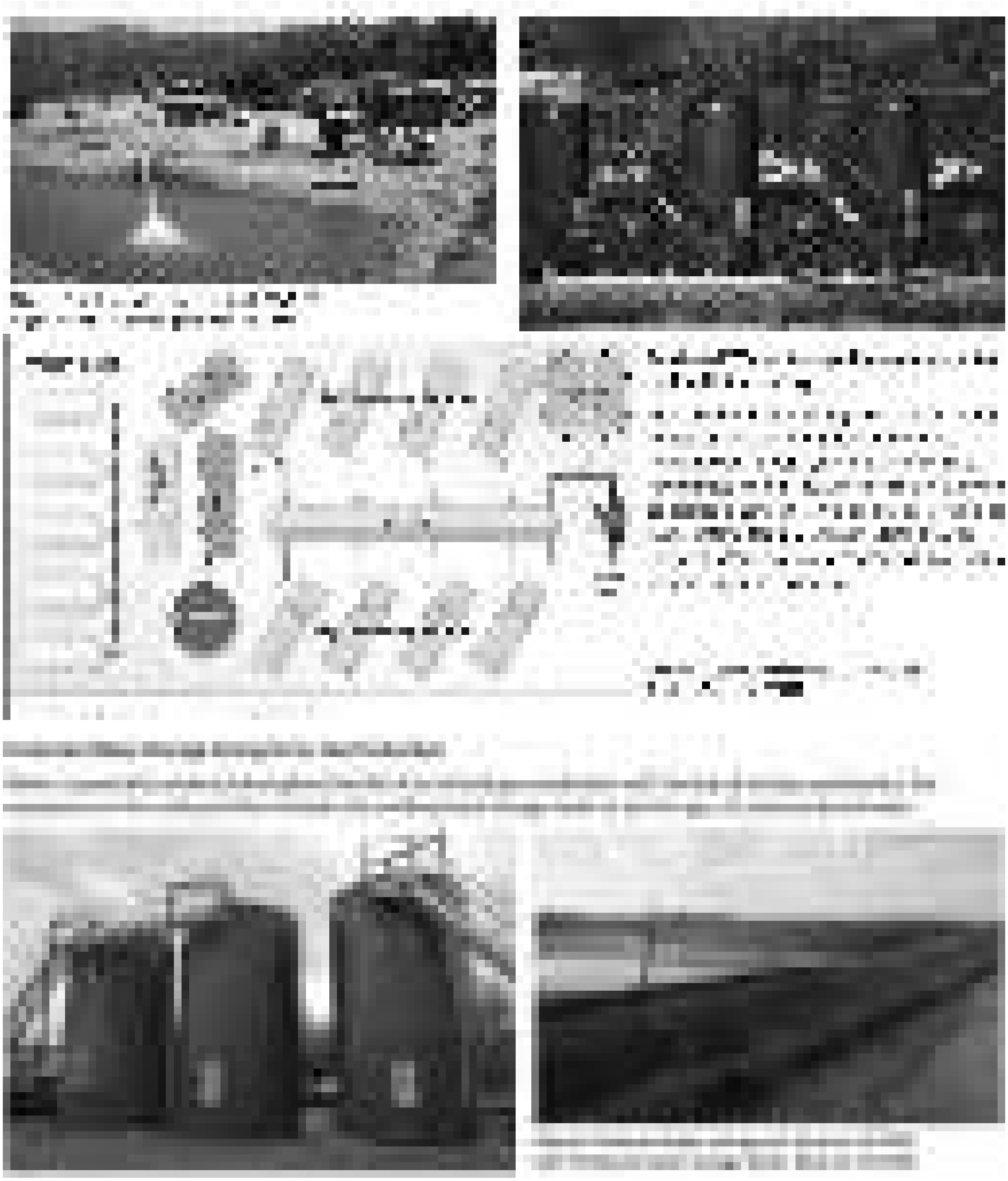
Documented cases of water resource impacts from produced water spills provide insights into the types of impacts that can occur. In most of the cases reviewed for this report, documented impacts included elevated levels of salinity in groundwater and/or surface water resources.[2] For example, the largest produced water spill reported in this report occurred in North Dakota in 2015, when approximately 2.9 million gallons (11 million liters) of produced water spilled from a broken pipeline. The spilled fluid flowed into Blacktail Creek and increased the concentration of chloride and the electrical conductivity of the creek; these observations are consistent with an increase in water salinity. Elevated levels of electrical conductivity and chloride were also found downstream in the Little Muddy River and the Missouri River. In another example, pits holding flowback fluids overflowed in Kentucky in 2007. The spilled fluid reached the Acorn Fork Creek, decreasing the pH of the creek and increasing the electrical conductivity.

[1] See Section 7.4 in Chapter 7.

[2] Groundwater impacts from produced water management practices are described in Chapter 8 and summarized in the "Wastewater Disposal and Reuse" section below.

Text Box ES-10. On-Site Storage of Produced Water.

Water that returns to the surface after hydraulic fracturing is collected and stored on site in pits or tanks.



Site-specific studies of historical produced water releases highlight the role of local geology in the movement of produced water through the environment. Whittemore (2007) described a site in Kansas where low permeability soils and rock caused produced water to primarily flow over the land surface to nearby surface water resources, reducing the amount of produced water that infiltrated soil. In contrast, Otton et al. (2007) explored the release of produced water and oil from two pits in Oklahoma. In this case, produced water from the pits flowed through thin soil and into the underlying, permeable rock. Produced water was also identified in deeper, less permeable rock. The authors suggest that produced water moved into the deeper, less permeable rock through natural fractures. Together, these studies highlight the role of preferential flow paths (i.e., paths of least resistance) in the movement of produced water through the environment.

Spill response activities likely reduce the severity of impacts on groundwater and surface water resources from produced water spills. For example, in the North Dakota example noted above, absorbent booms were placed in the affected creek and contaminated soil and oil-coated ice were removed from the site. In another example, a pipeline leak in Pennsylvania spilled approximately 11,000 gallons (42,000 liters) of produced water, which flowed into a nearby stream. In response, the pipeline was shut off, a dam was constructed to contain the spilled produced water, water was removed from the stream, and the stream was flushed with fresh water. In both examples, it was not possible to quantify how spill response activities reduced the severity of impacts on groundwater or surface water resources. However, actions taken after the spills were designed to stop produced water from entering the environment (e.g., shutting off a pipeline), remove produced water from the environment (e.g., using absorbent booms), and reduce the concentration of produced water constituents introduced into water resources (e.g., flushing a stream with fresh water).

The severity of impacts on water quality from spills of produced water depends on the identity and amount of produced water constituents that reach groundwater or surface water resources, the toxicity of those constituents, and the characteristics of the receiving water resource.[1] In particular, spills of produced water can have high levels of total dissolved solids, which affects how the spilled fluid moves through the environment. When a spilled fluid has greater levels of total dissolved solids than groundwater, the higher-density fluid can move downward through groundwater resources. Depending on the flow rate and other properties of the groundwater resource, impacts from produced water spills can last for years.

Produced Water Handling Conclusions

Spills of produced water during the produced water handling stage of the hydraulic fracturing water cycle have reached groundwater and surface water resources in some cases. Several cases of water resource impacts from produced water spills suggest that impacts are characterized by increases in the salinity of the affected groundwater or surface water resource. In the absence of direct pathways to groundwater resources (e.g., fractured rock), large volume spills are more likely to travel further from the site of the spill, potentially to groundwater or surface water resources.

[1] Human health hazards associated with chemicals detected in produced water are discussed in Chapter 9 and summarized in the "Chemicals in the Hydraulic Fracturing Water Cycle" section below.

Additionally, saline produced water can migrate downward through soil and into groundwater resources, leading to longer-term groundwater contamination. Spill prevention and response activities can prevent spilled fluids from reaching groundwater or surface water resources and minimize impacts from spilled fluids.

Wastewater Disposal and Reuse

Activity: The disposal and reuse of hydraulic fracturing wastewater.

Relationship to Drinking Water Resources: Disposal practices can release inadequately treated or untreated hydraulic fracturing wastewater to groundwater and surface water resources.

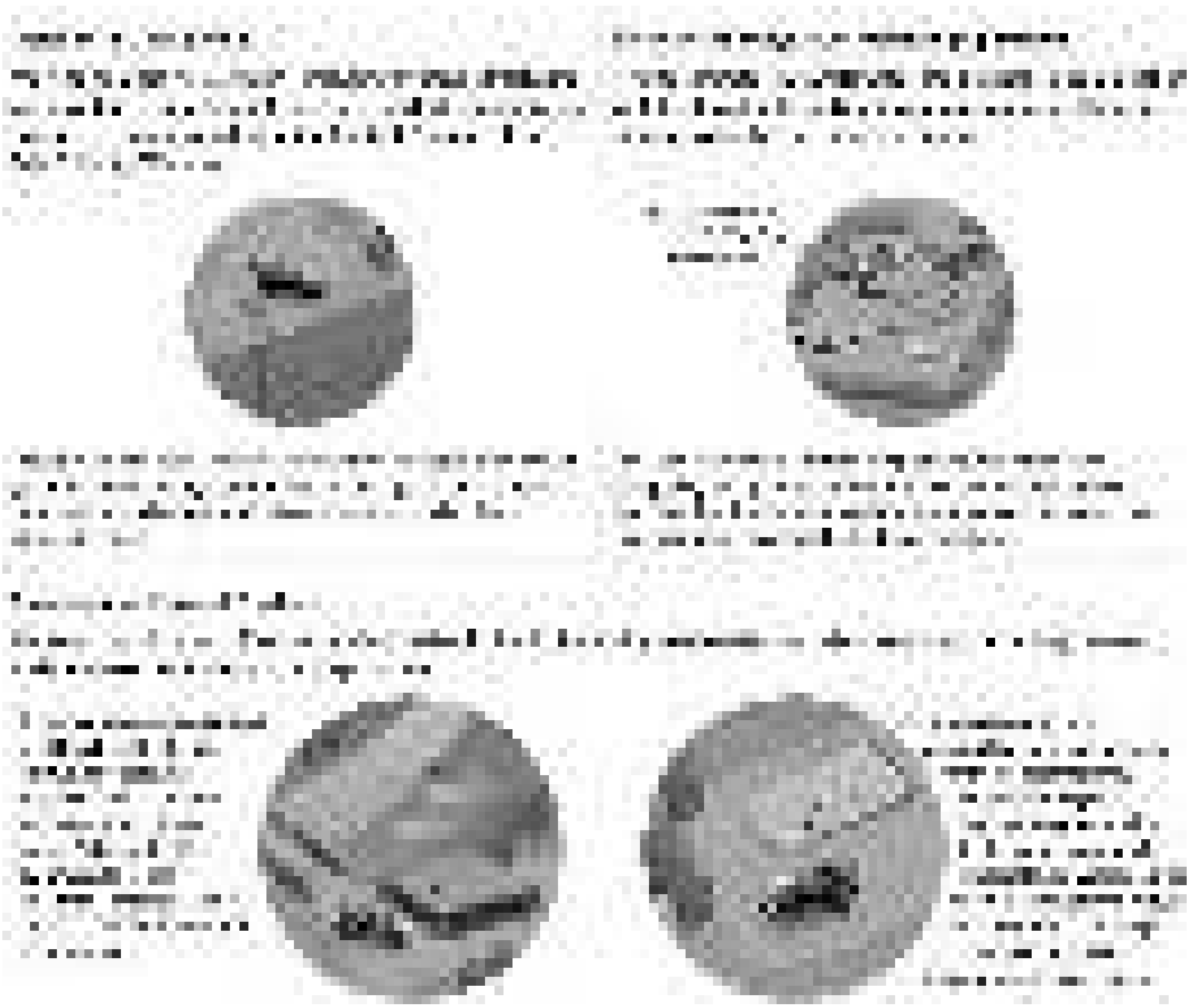
In general, produced water from hydraulically fractured oil and gas production wells is managed through injection in Class II wells, reuse in other hydraulic fracturing operations, or various aboveground disposal practices (Text Box ES-11). In this report, produced water from hydraulically fractured oil and gas wells that is being managed through one of the above management strategies is referred to as "hydraulic fracturing wastewater." Wastewater management choices are affected by cost and other factors, including: the local availability of disposal methods; the quality of produced water; the volume, duration, and flow rate of produced water; federal, state, and local regulations; and well operator preferences.

Available information suggests that hydraulic fracturing wastewater is mostly managed through injection in Class II wells. Veil (2015) estimated that 93% of produced water from the oil and gas industry was injected in Class II wells in 2012. Although this estimate included produced water from oil and gas wells in general, it is likely indicative of nationwide management practices for hydraulic fracturing wastewater. Disposal of hydraulic fracturing wastewater in Class II wells is often cost-effective, especially when a Class II disposal well is located within a reasonable distance from a hydraulically fractured oil or gas production well. In particular, large numbers of active Class II disposal wells are found in Texas (7,876), Kansas (5,516), Oklahoma (3,837), Louisiana (2,448), and Illinois (1,054) (U.S. EPA, 2016d). Disposal of hydraulic fracturing wastewater in Class II wells has been associated with earthquakes in several states, which may reduce the availability of injection in Class II wells as a wastewater disposal option in these states.

Nationwide, aboveground disposal and reuse of hydraulic fracturing wastewater are currently practiced to a much lesser extent compared to injection in Class II wells, and these management strategies appear to be concentrated in certain parts of the United States. For example, approximately 90% of hydraulic fracturing wastewater from Marcellus Shale gas wells in Pennsylvania was reused in other hydraulic fracturing operations in 2013 (Figure ES-4a). Reuse in hydraulic fracturing operations is practiced in some other areas of the United States as well, but at lower rates (approximately 5-20%). Evaporation ponds and percolation pits have historically been used in the western United States to manage produced water from the oil and gas industry and have likely been used to manage hydraulic fracturing wastewater. Percolation pits, in particular, were commonly reported to have been used to manage produced water from stimulated wells in Kern

Text Box ES-11. Hydraulic Fracturing Wastewater Management.

Produced water from hydraulically fractured oil and gas production wells is often, but not always, considered a waste product to be managed. Hydraulic fracturing wastewater (i.e., produced water from hydraulically fractured wells) is generally managed through injection in Class II wells, reuse in other hydraulic fracturing operations, and various aboveground disposal practices.



Federal and state regulations affect aboveground disposal management options. For example, existing federal regulations generally prevent the direct release of wastewater pollutants to waters of the United States from onshore oil and gas extraction facilities east of the 98th meridian. However, in the arid western portion of the continental United States (west of the 98th meridian), direct discharges of wastewater from onshore oil and gas extraction facilities to waters of the United States may be permitted if the produced water has a use in agriculture or wildlife propagation and meets established water quality criteria when discharged.

County, California, between 2011 and 2014.[1] Beneficial uses (e.g., livestock watering and irrigation) are also practiced in the western United States if the water quality is considered acceptable, although available data on the use of these practices are incomplete.

Aboveground disposal practices generally release treated or, under certain conditions, untreated wastewater directly to surface water or the land surface (e.g., wastewater treatment facilities, evaporation pits, or irrigation). If released to the land surface, treated or untreated wastewater can move through soil to groundwater resources. Because the ultimate fate of the wastewater can be groundwater or surface water resources, the aboveground disposal of hydraulic fracturing wastewater, in particular, can impact drinking water resources.

Impacts on drinking water resources from the aboveground disposal of hydraulic fracturing wastewater have been documented. For example, early wastewater management practices in the Marcellus Shale region in Pennsylvania included the use of wastewater treatment facilities that released (i.e., discharged) treated wastewater to surface waters (Figure ES-8). The wastewater treatment facilities were unable to adequately remove the high levels of total dissolved solids found in produced water from Marcellus Shale gas wells, and the discharges contributed to elevated levels of total dissolved solids (particularly bromide) in the Monongahela River Basin. In the Allegheny River Basin, elevated bromide levels were linked to increases in the concentration of hazardous disinfection byproducts in at least one downstream drinking water facility and a shift to more toxic brominated disinfection byproducts.[2] In response, the Pennsylvania Department of Environmental Protection revised existing regulations to prevent these discharges and also requested that oil and gas operators voluntarily stop bringing certain kinds of hydraulic fracturing wastewater to facilities that discharge inadequately treated wastewater to surface waters.[3]

The scientific literature and recent data from the Pennsylvania Department of Environmental Protection suggest that other produced water constituents (e.g., barium, strontium, and radium) may have been introduced to surface waters through the release of inadequately treated hydraulic fracturing wastewater. In particular, radium has been detected in stream sediments at or near wastewater treatment facilities that discharged inadequately treated hydraulic fracturing wastewater. Such sediments can migrate if they are disturbed during dredging or flood events. Additionally, residuals from the treatment of hydraulic fracturing wastewater (i.e., the solids or liquids that remain after treatment) are concentrated in the constituents removed during treatment, and these residuals can impact groundwater or surface water resources if they are not managed properly.

[1] Hydraulic fracturing was the predominant stimulation practice. Other stimulation practices included acid fracturing and matrix acidizing. California updated its regulations in 2015 to prohibit the use of percolation pits for the disposal of fluids produced from stimulated wells.

[2] Disinfection byproducts form through chemical reactions between organic material and disinfectants, which are used in drinking water treatment. Human health hazards associated with disinfection byproducts are described in Section 9.5.6 in Chapter 9.

[3] See Text Box 8-1 in Chapter 8.

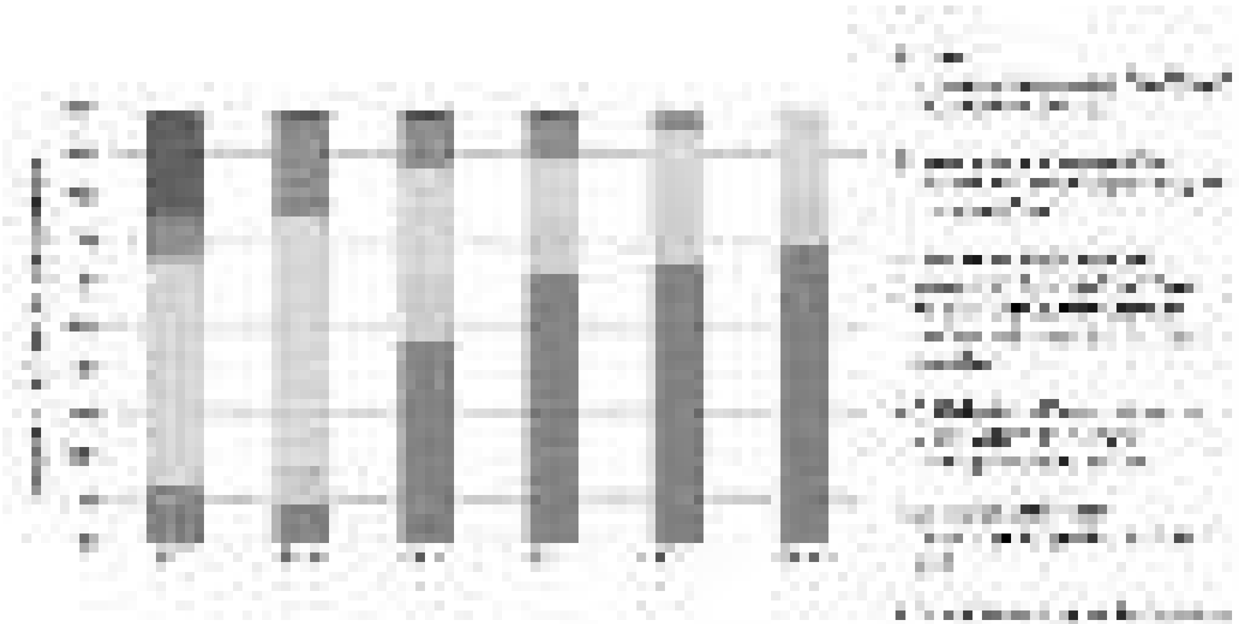


Figure ES-8. Changes in wastewater management practices over time in the Marcellus Shale area of Pennsylvania.
Data from PA DEP (2015a).

Impacts on groundwater and surface water resources from current and historic uses of lined and unlined pits, including percolation pits, in the oil and gas industry have been documented. For example, Kell (2011) reported 63 incidents of non-public water supply contamination from unlined or inadequately constructed pits in Ohio between 1983 and 2007, and 57 incidents of groundwater contamination from unlined produced water disposal pits in Texas prior to 1984. Other cases of impacts have been identified in several states, including New Mexico, Oklahoma, Pennsylvania, and Wyoming.[1] Impacts among these cases included the detection of volatile organic compounds in groundwater resources, wastewater reaching surface water resources from pit overflows, and wastewater reaching groundwater resources through liner failures. Based on documented impacts on groundwater resources from unlined pits, many states have implemented regulations that prohibit percolation pits or unlined storage pits for either hydraulic fracturing wastewater or oil and gas wastewater in general.

The severity of impacts on drinking water resources from the aboveground disposal of hydraulic fracturing wastewater depends on the volume and quality of the discharged wastewater and the characteristics of the receiving water resource. In general, large surface water resources with high flow rates can reduce the severity of impacts through dilution, although impacts may not be eliminated. In contrast, groundwater is generally slow moving, which can lead to an accumulation of hydraulic fracturing wastewater contaminants in groundwater from continuous or repeated discharges to the land surface; the resulting contamination can be long-lasting. The severity of

[1] See Section 8.4.5 in Chapter 8.

impacts on groundwater resources will also be influenced by soil and sediment properties and other factors that control the movement or degradation of wastewater constituents.

Wastewater Disposal and Reuse Conclusions

The aboveground disposal of hydraulic fracturing wastewater has impacted the quality of groundwater and surface water resources in some instances. In particular, discharges of inadequately treated hydraulic fracturing wastewater to surface water resources have contributed to elevated levels of hazardous disinfection byproducts in at least one downstream drinking water system. Additionally, the use of lined and unlined pits for the storage or disposal of oil and gas wastewater has impacted surface and groundwater resources. Unlined pits, in particular, provide a direct pathway for contaminants to reach groundwater. Wastewater management is dynamic, and recent changes in state regulations and practices have been made to limit impacts on groundwater and surface water resources from the aboveground disposal of hydraulic fracturing wastewater.

Chemicals in the Hydraulic Fracturing Water Cycle

Chemicals are present in the hydraulic fracturing water cycle. During the chemical mixing stage of the hydraulic fracturing water cycle, chemicals are intentionally added to water to alter its properties for hydraulic fracturing (Text Box ES-6). Produced water, which is collected, handled, and managed in the last two stages of the hydraulic fracturing water cycle, contains chemicals added to hydraulic fracturing fluids, naturally occurring chemicals found in hydraulically fractured rock formations, and any chemical transformation products (Text Box ES-9). By evaluating available data sources, we compiled a list of 1,606 chemicals that are associated with the hydraulic fracturing water cycle, including 1,084 chemicals reported to have been used in hydraulic fracturing fluids and 599 chemicals detected in produced water. This list represents a national analysis; an individual well would likely have a fraction of the chemicals on this list and may have other chemicals that were not included on this list.

In many stages of the hydraulic fracturing water cycle, the severity of impacts on drinking water resources depends, in part, on the identity and amount of chemicals that enter the environment. The properties of a chemical influence how it moves and transforms in the environment and how it interacts with the human body. Therefore, some chemicals in the hydraulic fracturing water cycle are of more concern than others because they are more likely to move with water (e.g., spilled hydraulic fracturing fluid) to drinking water resources, persist in the environment (e.g., chemicals that do not degrade), and/or affect human health.

Evaluating potential hazards from chemicals in the hydraulic fracturing water cycle is most useful at local and/or regional scales because chemical use for hydraulic fracturing can vary from well to well and because the characteristics of produced water are influenced by the geochemistry of hydraulically fractured rock formations. Additionally, site-specific characteristics (e.g., the local landscape, and soil and subsurface permeability) can affect whether and how chemicals enter drinking water resources, which influences how long people may be exposed to specific chemicals and at what concentrations. As a first step for informing site-specific risk assessments, the EPA compiled toxicity values for chemicals in the hydraulic fracturing water cycle from federal, state, and international sources that met the EPA's criteria for inclusion in this report.[1,2]

The EPA was able to identify chronic oral toxicity values from the selected data sources for 98 of the 1,084 chemicals that were reported to have been used in hydraulic fracturing fluids between 2005 and 2013. Potential human health hazards associated with chronic oral exposure to these chemicals include cancer, immune system effects, changes in body weight, changes in blood chemistry, cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and developmental toxicity. Of the chemicals most frequently reported to FracFocus 1.0, nine had toxicity values from the selected data sources (Table ES-3). Critical effects for these chemicals include kidney/renal toxicity, hepatotoxicity, developmental toxicity (extra cervical ribs), reproductive toxicity, and decreased terminal body weight.

Table ES-3. Available chronic oral reference values for hydraulic fracturing chemicals reported in 10% or more of disclosures in FracFocus 1.0.

Chemical name (CASRN)[a]	Chronic oral reference value (mg/kg/day)	Critical effect	Percent of FracFocus 1.0 disclosures[b]
Propargyl alcohol (107-19-7)	0.002[c]	Renal and hepatotoxicity	33
1,2,4-Trimethylbenzene (95-63-6)	0.01[c]	Decreased pain sensitivity	13
Naphthalene (91-20-3)	0.02[c]	Decreased terminal body weight	19
Sodium chlorite (7758-19-2)	0.03[c]	Neuro-developmental effects	11
2-Butoxyethanol (111-76-2)	0.1[c]	Hemosiderin deposition in the liver	23
Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides (68424-85-1)	0.44[d]	Decreased body weight and weight gain	12
Formic acid (64-18-6)	0.9[e]	Reproductive toxicity	11
Ethylene glycol (107-21-1)	2[c]	Kidney toxicity	47
Methanol (67-56-1)	2[c]	Extra cervical ribs	73

[a] "Chemical" refers to chemical substances with a single CASRN; these may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates).

[b] Analysis considered 35,957 disclosures that met selected quality assurance criteria. See Table 9-2 in Chapter 9.

[c] From the EPA Integrated Risk Information System database.

[d] From the EPA Human Health Benchmarks for Pesticides database.

[e] From the EPA Provisional Peer-Reviewed Toxicity Value database.

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[1] Specifically, the EPA compiled noncancer oral reference values and cancer oral slope factors (Chapter 9). A reference value describes the dose of a chemical that is likely to be without an appreciable risk of adverse health effects. In the context of this report, the term "reference value" generally refers to reference values for noncancer effects occurring via the oral route of exposure and for chronic durations. An oral slope factor is an upper-bound estimate on the increased cancer risk from a lifetime oral exposure to an agent.

[2] The EPA's criteria for inclusion in this report are described in Section 9.4.1 in Chapter 9. Sources of information that met these criteria are listed in Table 9-1 of Chapter 9.

Chronic oral toxicity values from the selected data sources were identified for 120 of the 599 chemicals detected in produced water. Potential human health hazards associated with chronic oral exposure to these chemicals include liver toxicity, kidney toxicity, neurotoxicity, reproductive and developmental toxicity, and carcinogenesis. Chemical-specific toxicity values are included in Chapter 9.

Chemicals in the Hydraulic Fracturing Water Cycle Conclusions

Some of the chemicals in the hydraulic fracturing water cycle are known to be hazardous to human health. Of the 1,606 chemicals identified by the EPA, 173 had chronic oral toxicity values from federal, state, and international sources that met the EPA's criteria for inclusion in this report. These data alone, however, are insufficient to determine which chemicals have the greatest potential to impact drinking water resources and human health. To understand whether specific chemicals can affect human health through their presence in drinking water, data on chemical concentrations in drinking water would be needed. In the absence of these data, relative hazard potential assessments could be conducted at local and/or regional scales using the multi-criteria decision analysis approach outlined in Chapter 9. This approach combines available chemical occurrence data with selected chemical, physical, and toxicological properties to place the severity of potential impacts (i.e., the toxicity of specific chemicals) into the context of factors that affect the likelihood of impacts (i.e., frequency of use, and chemical and physical properties relevant to environmental fate and transport).

Data Gaps and Uncertainties

The information reviewed for this report included cases of impacts on drinking water resources from activities in the hydraulic fracturing water cycle. Using these cases and other data, information, and analyses, we were able to identify factors that likely result in more frequent or more severe impacts on drinking water resources. However, there were instances in which we were unable to form conclusions about the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources and/or the factors that influence the frequency or severity of impacts. Below, we provide perspective on the data gaps and uncertainties that prevented us from drawing additional conclusions about the potential for impacts on drinking water resources and/or the factors that affect the frequency and severity of impacts.

In general, comprehensive information on the location of activities in the hydraulic fracturing water cycle is lacking, either because it is not collected, not publicly available, or prohibitively difficult to aggregate. This includes information on the:

- Above- and belowground locations of water withdrawals for hydraulic fracturing;
- Surface locations of hydraulically fractured oil and gas production wells, where the chemical mixing, well injection, and produced water handling stages of the hydraulic fracturing water cycle take place;
- Belowground locations of hydraulic fracturing, including data on fracture growth; and
- Locations of hydraulic fracturing wastewater management practices, including the disposal of treatment residuals.

There can also be uncertainty in the location of drinking water resources. In particular, depths of groundwater resources that are, or in the future could be, used for drinking water are not always known. If comprehensive data about the locations of both drinking water resources and activities in the hydraulic fracturing water cycle were available, it would have been possible to more completely identify areas in the United States in which hydraulic fracturing-related activities either directly interact with drinking water resources or have the potential to interact with drinking water resources.

In places where we know activities in the hydraulic fracturing water cycle have occurred or are occurring, data that could be used to characterize the presence, migration, or transformation of hydraulic fracturing-related chemicals in the environment before, during, and after hydraulic fracturing were scarce. Specifically, local water quality data needed to compare pre- and post-hydraulic fracturing conditions are not usually collected or readily available. The limited amount of data collected before, during, and after activities in the hydraulic fracturing water cycle reduces the ability to determine whether these activities affected drinking water resources.

Site-specific cases of alleged impacts on underground drinking water resources during the well injection stage of the hydraulic fracturing water cycle are particularly challenging to understand (e.g., methane migration in Dimock, Pennsylvania; the Raton Basin of Colorado; and Parker County, Texas[1]). This is because the subsurface environment is complex and belowground fluid movement is not directly observable. In cases of alleged impacts, activities in the hydraulic fracturing water cycle may be one of several causes of impacts, including other oil and gas activities, other industries, and natural processes. Thorough scientific investigations are often necessary to narrow down the list of potential causes to a single source at site-specific cases of alleged impacts.

Additionally, information on chemicals in the hydraulic fracturing water cycle (e.g., chemical identity; frequency of use or occurrence; and physical, chemical, and toxicological properties) is not complete. Well operators claimed at least one chemical as confidential at more than 70% of wells reported to FracFocus 1.0 (U.S. EPA, 2015a).[2] The identity and concentration of these chemicals, their transformation products, and chemicals in produced water would be needed to characterize how chemicals associated with hydraulic fracturing activities move through the environment and interact with the human body. Identifying chemicals in the hydraulic fracturing water cycle also informs decisions about which chemicals would be appropriate to test for when establishing pre-hydraulic fracturing baseline conditions and in the event of a suspected drinking water impact.

Of the 1,606 chemicals identified by the EPA in hydraulic fracturing fluid and/or produced water, 173 had toxicity values from sources that met the EPA's criteria for inclusion in this report. Toxicity values from these selected data sources were not available for 1,433 (89%) of the chemicals, although many of these chemicals have toxicity data available from other data sources.[3] Given the

[1] See Text Boxes 6-2 (Dimock, Pennsylvania), 6-3 (Raton Basin), and 6-4 (Parker County, Texas) in Chapter 6.

[2] Chemical withholding rates in FracFocus have increased over time. Konschnik and Dayalu (2016) reported that 92% of wells reported in FracFocus 2.0 between approximately March 2011 and April 2015 used at least one chemical that was claimed as confidential.

[3] Chapter 9 describes the availability of data in other data sources. The quality of these data sources was not evaluated as part of this report.

large number of chemicals identified in the hydraulic fracturing water cycle, this missing information represents a significant data gap that makes it difficult to fully understand the severity of potential impacts on drinking water resources.

Because of the significant data gaps and uncertainties in the available data, it was not possible to fully characterize the severity of impacts, nor was it possible to calculate or estimate the national frequency of impacts on drinking water resources from activities in the hydraulic fracturing water cycle. We were, however, able to estimate impact frequencies in some, limited cases (i.e., spills of hydraulic fracturing fluids or produced water and mechanical integrity failures).[1] The data used to develop these estimates were often limited in geographic scope or otherwise incomplete. Consequently, national estimates of impact frequencies for any stage of the hydraulic fracturing water cycle have a high degree of uncertainty. Our inability to quantitatively determine a national impact frequency or to characterize the severity of impacts, however, did not prevent us from qualitatively describing factors that affect the frequency or severity of impacts at the local level.

Report Conclusions

This report describes how activities in the hydraulic fracturing water cycle can impact—and have impacted—drinking water resources and the factors that influence the frequency and severity of those impacts. It also describes data gaps and uncertainties that limited our ability to draw additional conclusions about impacts on drinking water resources from activities in the hydraulic fracturing water cycle. Both types of information—what we know and what we do not know—provide stakeholders with scientific information to support future efforts.

The uncertainties and data gaps identified throughout this report can be used to identify future efforts to further our understanding of the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources and the factors that affect the frequency and severity of those impacts. Future efforts could include, for example, groundwater and surface water monitoring in areas with hydraulically fractured oil and gas production wells or targeted research programs to better characterize the environmental fate and transport and human health hazards associated with chemicals in the hydraulic fracturing water cycle. Future efforts could identify additional vulnerabilities or other factors that affect the frequency and/or severity of impacts.

In the near term, decision-makers could focus their attention on the combinations of hydraulic fracturing water cycle activities and local- or regional-scale factors that are more likely than others to result in more frequent or more severe impacts. These include:

- Water withdrawals for hydraulic fracturing in times or areas of low water availability, particularly in areas with limited or declining groundwater resources;
- Spills during the management of hydraulic fracturing fluids and chemicals or produced water that result in large volumes or high concentrations of chemicals reaching groundwater resources;

[1] See Chapter 10.

- Injection of hydraulic fracturing fluids into wells with inadequate mechanical integrity, allowing gases or liquids to move to groundwater resources;
- Injection of hydraulic fracturing fluids directly into groundwater resources;
- Discharge of inadequately treated hydraulic fracturing wastewater to surface water resources; and
- Disposal or storage of hydraulic fracturing wastewater in unlined pits, resulting in contamination of groundwater resources.

The above combinations of activities and factors highlight, in particular, the vulnerability of groundwater resources to activities in the hydraulic fracturing water cycle. By focusing attention on the situations described above, impacts on drinking water resources from activities in the hydraulic fracturing water cycle could be prevented or reduced.

Overall, hydraulic fracturing for oil and gas is a practice that continues to evolve. Evaluating the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources will need to keep pace with emerging technologies and new scientific studies. This report provides a foundation for these efforts, while helping to reduce current vulnerabilities to drinking water resources.

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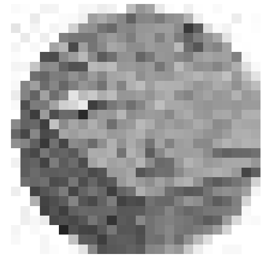

Chapter 1. Introduction

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1. Introduction

1.1 Background

People rely on clean and plentiful water resources to meet their basic needs. In the early 2000s, members of the public began to raise concerns about the use of hydraulic fracturing for oil and gas production and its potential impacts on drinking water resources. Hydraulic fracturing involves the injection of fluids into a well under pressures great enough to fracture oil- and gas-bearing formations. While hydraulic fracturing has been used to enhance oil and gas production from conventional rock formations, the combination of hydraulic fracturing and directional drilling has made it economical to produce oil and gas from previously unused unconventional rock formations.[1] This has led to increases in oil and gas production and expanded activity throughout the United States.

Concerns about the impacts of hydraulic fracturing activities on both the quality and quantity of drinking water resources have been raised by the public. Some residents living close to oil and gas production wells report changes in the quality of groundwater resources used for drinking water and assert that hydraulic fracturing is responsible for these changes. Other concerns include impacts on water availability due to water use in hydraulic fracturing, especially in areas of the country experiencing drought, and impacts on water quality from the disposal of wastewater generated after hydraulic fracturing.

In response to public concerns, the U.S. Congress urged the U.S. Environmental Protection Agency (EPA) to study the relationship between hydraulic fracturing and drinking water (H.R. Rep. 111-316, 2009). In 2011, the EPA published its *Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2011d; hereafter Study Plan), which described the research the Agency would be conducting on activities involving water that support hydraulic fracturing (referred to as the "hydraulic fracturing water cycle"). The research described in the Study Plan began the same year. In 2012, the EPA issued *Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources: Progress Report* (U.S. EPA, 2012h; hereafter Progress Report) to update the public on the status of EPA's research. Since its initiation, the EPA's hydraulic fracturing study has directly resulted in the publication of 27 separate government reports and scientific journal articles. This assessment integrates results from those reports and scientific journal articles with publicly available data and information. It represents the culmination of the EPA's hydraulic fracturing study focused on characterizing the relationship between hydraulic fracturing and drinking water.

[1] Conventional oil- and gas-bearing rock formations are often described as "permeable" and tend to have many large, well-connected pore spaces that allow fluids to move within the rock formation. Unconventional oil- and gas-bearing rock formations do not exhibit these characteristics. See Chapter 3 for more information on uses of the terms conventional and unconventional.

1.2 Goals

The goals of this assessment are to assess the potential for activities in the hydraulic fracturing water cycle to impact the quality or quantity of drinking water resources and to identify factors that affect the frequency or severity of those impacts.

1.3 Scope

The hydraulic fracturing water cycle defines the activities that are within the scope of this assessment. This cycle encompasses activities involving water that support hydraulic fracturing and consists of five stages:

1. **Water Acquisition:** the withdrawal of groundwater or surface water to make hydraulic fracturing fluids;
2. **Chemical Mixing:** the mixing of a base fluid (typically water), proppant, and additives at the well site to create hydraulic fracturing fluids;[1]
3. **Well Injection:** the injection and movement of hydraulic fracturing fluids through the oil and gas production well and in the targeted rock formation;
4. **Produced Water Handling:** the on-site collection and handling of water that returns to the surface after hydraulic fracturing and the transportation of that water for disposal or reuse; and [2]
5. **Wastewater Disposal and Reuse:** the disposal and reuse of hydraulic fracturing wastewater.[3]

The hydraulic fracturing water cycle, and thus the scope of this assessment, was developed with input from stakeholders (i.e., federal, state, and tribal partners; industry and non-governmental organizations; and the general public) and the EPA's Science Advisory Board (SAB) (U.S. EPA, 2011d). The hydraulic fracturing water cycle and our assessment scope reflect interest from stakeholders in understanding impacts from the act of hydraulic fracturing itself as well as the activities involving water that support it, without examining impacts from oil and gas production development broadly.

[1] A base fluid is the fluid into which proppants and additives are mixed to make a hydraulic fracturing fluid; water is an example of a base fluid. Additives are chemicals or mixtures of chemicals that are added to the base fluid to change its properties.

[2] "Produced water" is defined in this report as water that flows from and through oil and gas wells to the surface as a by-product of oil and gas production.

[3] "Hydraulic fracturing wastewater" is defined in this report as produced water from hydraulically fractured oil and gas wells that is being managed using practices that include, but are not limited to, injection in Class II wells, reuse in other hydraulic fracturing operations, and various aboveground disposal practices. The term "wastewater" is being used as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes. Class II wells are used to inject wastewater associated with oil and gas production underground and are regulated under the Underground Injection Control Program of the Safe Drinking Water Act.

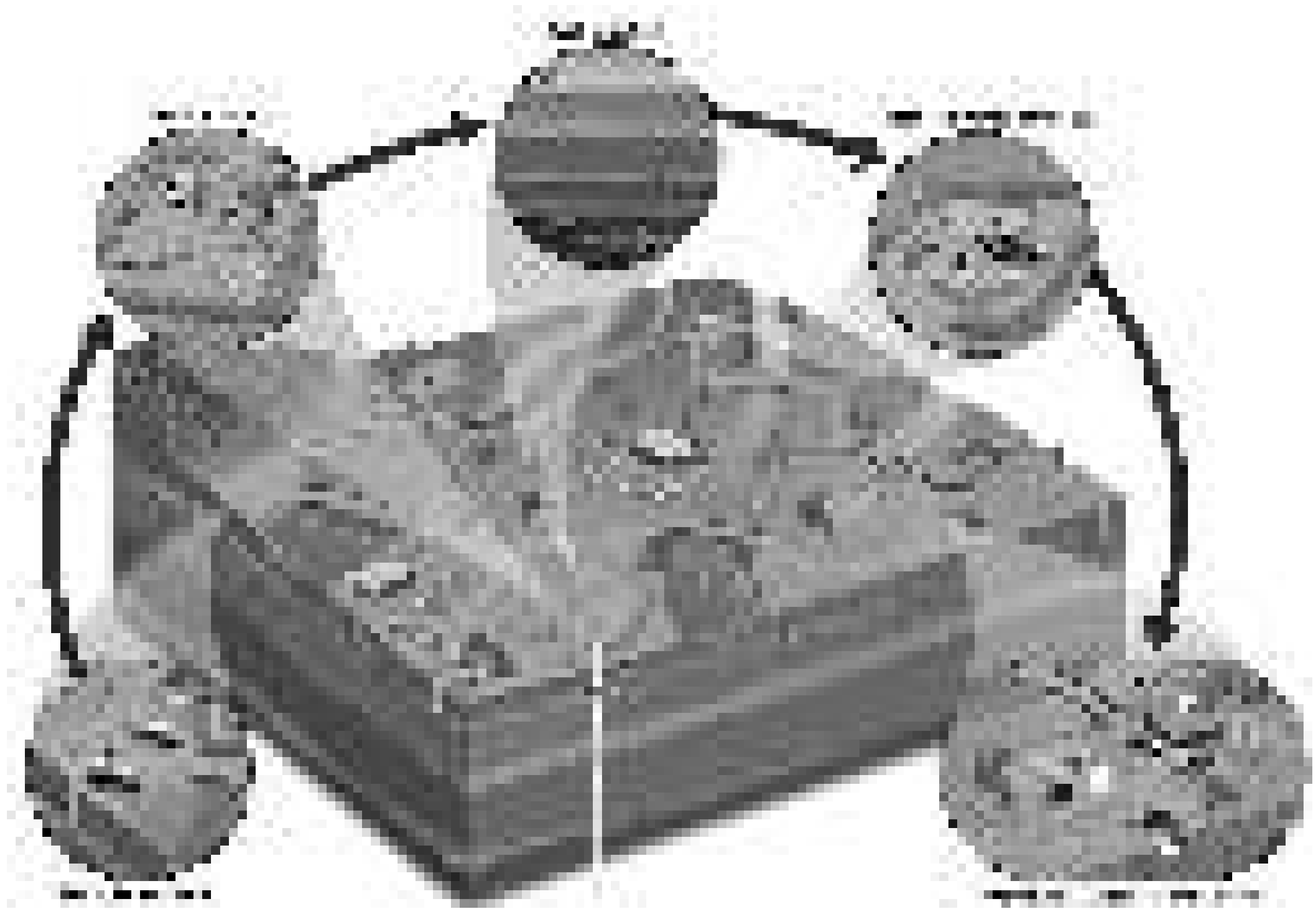

Figure 1-1. Conceptualized view of the stages of the hydraulic fracturing water cycle.
Shown here is a generalized landscape depicting simplified activities of the hydraulic fracturing water cycle, their relationship to each other, and their relationship to drinking water resources. Activities may take place in the same watershed or different watersheds and close to or far from drinking water resources. Drinking water resources are any groundwater or surface water that now serves, or in the future could serve, as a source of drinking water for public or private use. Arrows depict the movement of water and chemicals. Specific activities in the “Wastewater Disposal and Reuse” inset are (a) disposal via injection well, (b) wastewater treatment with reuse or discharge, and (c) evaporation or percolation pit disposal. Note: Figure not to scale.

This assessment focuses on hydraulic fracturing in onshore oil and gas wells in the contiguous United States; limited available information on hydraulic fracturing in Alaska is included. To the extent possible, this assessment addresses hydraulic fracturing in all types of oil- and gas-bearing formations in which it is conducted, including shale, so-called 'tight' formations (e.g., certain sandstones, siltstones, and carbonates), coalbeds, and conventional rock formations. The assessment tends to focus on hydraulic fracturing in shale, reflecting the abundance and availability of literature and data on hydraulic fracturing in this type of rock formation.

In this assessment, we consider how activities in the hydraulic fracturing water cycle interact with drinking water resources. Consistent with the Study Plan (U.S. EPA, 2011d), drinking water resources are defined within this assessment as any groundwater or surface water that now serves, or in the future could serve, as a source of drinking water for public or private use. This definition is broader than most regulatory definitions of "drinking water" to include both fresh and non-fresh bodies of water that are and could be used now or could be used in the future as sources of drinking water (Chapter 2). We note that drinking water resources provide not only water that individuals actually drink but also water used for many additional purposes such as cooking and bathing.

As part of the assessment, we evaluated immediate, near-term, and delayed effects on drinking water resources from normal operations and accidents. For example, we considered how surface spills of hydraulic fracturing fluids may have immediate or near-term impacts on neighboring surface water and shallow groundwater quality (Chapters 5 and 7). We also considered how the potential release of hydraulic fracturing fluids in the subsurface may take years to impact groundwater resources, because liquids and gas often move slowly in the subsurface (Chapter 6). Additionally, impacts may be transient or long-term, often depending on the characteristics of the affected drinking water resource. Finally, impacts may be detected near the hydraulic fracturing water cycle activity or some distance away. For instance, we considered that, depending on the constituents of treated hydraulic fracturing wastewater discharged to a stream and the flow in that stream, drinking water resource quality could be affected a significant distance downstream (Chapter 8).

This assessment focuses predominantly on activities supporting a single well or multiple wells at one site, accompanied by a more limited discussion of cumulative activities and the impacts that could result from having many wells on a landscape. Studies of cumulative effects are generally lacking, but we use the scientific literature to address this topic where possible.[1]

We examine *impacts* of hydraulic fracturing for oil and gas on drinking water resources and address *factors* that affect the *frequency* or *severity* of impacts. Specific definitions used in this assessment are provided below:

- An **impact** is any change in the quality or quantity of drinking water resources, regardless of severity, that results from an activity in the hydraulic fracturing water cycle.

[1] Cumulative effects refer to combined changes in the environment that can take place as a result of multiple activities over time and/or space.

- A **factor** is a feature of hydraulic fracturing operations or an environmental condition that affects the frequency or severity of impacts.
- **Frequency** is the number of impacts per a given unit (e.g., per geographic area, per unit time, per number of hydraulically fractured wells, per number of water bodies). Reflecting the scientific literature, the most common representation of frequency in this assessment is number of impacts per hydraulically fractured well.
- **Severity** is the magnitude of change in the quality or quantity of a drinking water resource as measured by a given metric (e.g., duration, spatial extent, contaminant concentration).

We identify and discuss factors affecting the frequency or severity of impacts to avoid a simple inventory of all specific situations in which hydraulic fracturing might alter drinking water quality or quantity. This allows knowledge about the conditions under which impacts are likely or unlikely to occur to be applied to new circumstances (e.g., a new area of oil or gas development where hydraulic fracturing is expected to be used) and could inform the development of strategies to prevent impacts. Although no attempt has been made in this assessment to identify or evaluate comprehensive best practices for states, tribes, or the industry, we describe ways to avoid or reduce the frequency or severity of impacts from hydraulic fracturing activities as they have been reported in the scientific literature. Laws, regulations, and policies also exist to protect drinking water resources (Text Box 1-1), but a comprehensive summary and evaluation of current or proposed regulations and policies is beyond the scope of this assessment.

Text Box 1-1. Regulatory Protection for Drinking Water Resources.

The quality and quantity of drinking water resources are protected in the United States by a collection of federal, state, tribal, and local laws, regulations, and polices. They differ with respect to how water resources are defined (Chapter 2) and thus which resources qualify for protection. Some policies protect water resources from oil and gas industry activities as part of a larger set of regulated industries, or from oil and gas industry activities only, or from hydraulic fracturing-related activities, specifically. Multiple federal and state agencies, departments, or divisions are responsible for implementing these laws, regulations, and policies. An exhaustive summary of current and emerging laws, regulations, and policies, those responsible for implementing them, and enforcement or effectiveness is not in the scope of this assessment. The following information is designed to give the reader a general understanding of how the U.S. government and states protect drinking water resources from the potential impacts of activities in the hydraulic fracturing water cycle.

On the federal level, the U.S. government regulates some activities in the hydraulic fracturing water cycle to protect drinking water resources. For example, under the Clean Water Act, the National Pollution Discharge Elimination System (NPDES) program regulates surface discharge of wastewater from the oil and gas sector (in addition to many other industries). Issuance and enforcement of NPDES discharge permits is primarily the responsibility of the states that have received NPDES program authorization from the EPA. In addition, the Safe Drinking Water Act's (SDWA) Underground Injection Control program regulates the underground disposal of hydraulic fracturing wastewater (and wastewater generated in other industries) and, like the NPDES program, allows states to seek program authorization from the EPA. The federal government does not have the authority to regulate hydraulic fracturing as an injection activity under the SDWA except when it

(Text Box 1-1 is continued on the following page.)

Text Box 1-1 (continued). Regulatory Protection for Drinking Water Resources.

(1) involves diesel fuel, a result of legislation passed in 2005, or (2) causes an imminent and substantial endangerment to the health of persons. Additionally, produced water is exempted from regulation as a hazardous waste under the Resource Conservation and Recovery Act Subtitle C. In 2015, the U.S. Department of the Interior published a set of regulations for conducting hydraulic fracturing operations on federal public and tribal lands. It includes requirements to help protect groundwater by updating standards for well mechanical integrity, wastewater disposal, and public disclosure of chemicals. As of late 2016, a federal district court judge has set aside these regulations as outside the scope of the U.S. Department of the Interior's authority, and this decision is being appealed.

States generally have the primary responsibility for protecting drinking water resources from the impacts of hydraulic fracturing activities (Guralnick, 2016; Zirogannis et al., 2016). Some states have put in place broad restrictions or moratoria on hydraulic fracturing activities due in part to concerns about potential risks to drinking water resources. Many other states allow hydraulic fracturing activities, and several sources of information track and/or summarize their laws, regulations, and policies. An online database of statutes and regulations applicable to the oil and gas industry and related to water quality, water quantity, and air quality in 17 states is maintained by LawAtlas (www. lawatlas.org/oilandgas).

State approaches vary widely, from comprehensive laws addressing all aspects of hydraulic fracturing activities to regulations addressing specific activities (Guralnick, 2016). In 2009 and 2014, the Ground Water Protection Council (GWPC) summarized regulations that are designed to protect water resources and applicable to the oil and gas industry in 27 states; they did not investigate compliance (GWPC, 2014, 2009). The summaries revealed that regulations are carried out by either oil and gas agencies, environmental agencies, or both, depending on the state. They also identified general categories of existing regulations that could control impacts on drinking water resources from activities in the hydraulic fracturing water cycle, including permitting, well design and integrity, injection activities, and surface management of fluids. Categories were comprised of regulatory "elements." Certain elements had been adopted across 90% or more of states included in the summaries that allowed hydraulic fracturing as of July 2013: surface casing generally must be set below the deepest protected groundwater zone; protected groundwater depth is determined on a well-specific basis or by rule; and surface casing must be cemented from bottom to top. All other elements were adopted at lower and widely varying rates. For example, as of July 2013, a requirement for water well testing and monitoring adjacent to hydraulic fracturing operations existed in five states. Other states, including California, have added this requirement since then.

State laws, regulations, and policies are continually changing. Changes may be initiated by state legislatures or regulatory agencies (sometimes in response to legal decisions) and generally apply to new wells or future hydraulic fracturing operations and not existing wells or wells that have been hydraulically fractured in the past. Third-party groups, like the State Review of Oil and Natural Gas Environmental Regulations (STRONGER) organization, offer multi-stakeholder reviews of state oil and gas regulatory programs and recommendations to improve those programs according to guidelines developed by their workgroups. Interstate organizations of state agency representatives also have initiatives to develop oil and gas resources while protecting water and other environmental resources, initiatives like the GWPC and Interstate Oil and Gas Compact Commission's States First. In combination with changing policies, new technologies (such as those that make it possible to reuse hydraulic fracturing wastewater in subsequent hydraulic fracturing operations) have the potential to further reduce impacts on drinking water resources.

We identify and evaluate potential human health hazards of hydraulic fracturing-related chemicals (Chapter 9), but this assessment is not a human health risk assessment. It does not identify populations that are exposed to chemicals or other stressors in the environment, estimate the extent of exposure, or estimate the incidence of human health impacts. Relatedly, we did not conduct site-specific predictive modeling to quantitatively estimate contaminant concentrations in drinking water resources, although modeling studies conducted by others are described.

This assessment focuses on the potential for impacts from activities in the hydraulic fracturing water cycle on drinking water resources. It does not address all concerns that have been raised about hydraulic fracturing nor about oil and gas exploration and production more generally. Activities that are not considered in this assessment include acquisition and transport of constituents of hydraulic fracturing fluids besides water (e.g., sand mining and chemical production); site selection and development; other infrastructure development (e.g., roads, pipelines, compressor stations); site reclamation; and well closure. We consider these activities to be outside the scope of the hydraulic fracturing water cycle and, therefore, their impacts are not addressed in this assessment. Disposal of hydraulic fracturing wastewater in underground injection control wells is described and characterized, but consistent with the Study Plan, potential for impacts of this practice on drinking water resources is not included. Additionally, this report does not discuss the potential impacts of hydraulic fracturing on other water uses (e.g., agriculture or industry), other aspects of the environment (e.g., air quality, induced seismicity, or ecosystems), worker health and safety, or communities. Finally, this assessment focuses on the available science and does not review, consider, or recommend policy options.

1.4 Approach

This assessment relies on scientific literature and data that address topics within the scope of the hydraulic fracturing water cycle. Scientific journal articles and peer-reviewed EPA reports containing results from the EPA's hydraulic fracturing study comprise one set of applicable literature. Other literature evaluated includes articles published in science and engineering journals, federal and state government reports, non-governmental organization (NGO) reports, and oil and gas industry publications. Data sources examined include federal- and state-collected data sets, databases curated by federal and state government agencies, other publicly available data and information, and data submitted by industry to the EPA.[1] In total, we cite approximately 1,200 sources of scientific data and information in this assessment.

1.4.1 EPA Hydraulic Fracturing Study Publications

The research topic areas and projects described in the Study Plan were developed with substantial expert and public input and were designed to meet the data and information needs of this assessment. As such, peer-reviewed results of research that the EPA conducted under the Study Plan, published separately as EPA reports or as journal articles, are incorporated and cited

[1] Confidential and non-confidential business information was provided to the EPA by nine hydraulic fracturing service companies in response to a September 2010 information request and by nine oil and gas well operators in response to an August 2011 information request.

frequently throughout this assessment. As is customary in assessments that synthesize a large body of literature and data, the results of EPA research are contextualized and interpreted in combination with the other literature and data described in Section 1.4.2. The journal articles and EPA reports that give complete and detailed project results can be found on the EPA's hydraulic fracturing study website (www.epa.gov/hfstudy). For ease of reference, a description of the individual projects, the type of research activity they represent (i.e., analysis of existing data, scenario evaluation, laboratory study, or case study), and the corresponding citations of published journal articles and EPA reports that are referenced in this assessment can be found in Appendix A.

1.4.2 Literature and Data Search Strategy

We used a broad search strategy to identify approximately 4,100 sources of scientific information applicable to this assessment. This strategy included requesting input from scientists, stakeholders, and the public about relevant data and information, and thorough searches of published information and applicable data.[1]

Over 1,600 articles, reports, data, and other sources of information were obtained through outreach to the public, stakeholders, and scientific experts. The EPA requested material through many venues, as follows. We received recommended literature from the SAB, the EPA's independent federal scientific advisory committee, from its review of the EPA's draft Study Plan; from its consultation on the EPA's Progress Report; during an SAB briefing on new and emerging information related to hydraulic fracturing in fall 2013; and from its peer review of the external review draft of this assessment. Subject matter experts and stakeholders also recommended literature through a series of technical workshops and roundtables organized by the EPA between 2011 and 2013. In addition, the public submitted literature recommendations to the SAB during the SAB review of the draft Study Plan, consultation on the Progress Report, briefing on emerging information, and review of the external review draft of this assessment, as well as in response to a formal request for data and information posted in the *Federal Register* (EPA-HQ-ORD-2010-0674) in November 2012. The submission deadline was extended from April to November 2013 to provide the public with additional opportunity to provide information to the EPA.

Approximately 2,500 additional sources were identified by conducting searches via online scientific databases and federal, state, and stakeholder websites. We searched these databases and websites in particular for (1) materials addressing topics not covered by the documents submitted by experts, stakeholders, and the public as noted above, and (2) newly emerging scientific studies. Multiple targeted and iterative searches on topics determined to be within the scope of the assessment were conducted until June 1, 2016. After that time, we included newer literature as it was recommended to us during our internal technical reviews or as it came to our attention and was determined to be important for filling a gap in information.

[1] This study did not review information contained in state and federal enforcement actions concerning alleged contamination of drinking water resources.

1.4.3 Literature and Data Evaluation Strategy

We evaluated the literature and data identified in the search strategy using the five assessment factors outlined by the EPA Science Policy Council in *A Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information* (U.S. EPA, 2003c). The factors are (1) applicability and utility, (2) evaluation and review, (3) soundness, (4) clarity and completeness, and (5) uncertainty and variability. Table 1-1 lists these factors along with the specific criteria developed for this assessment. We first evaluated all materials for applicability. If we determined that the material was "applicable" under the criteria, the reference was evaluated on the basis of the other four factors.

Our objective was to consider and then cite literature in the assessment that fully conforms to all criteria defining each assessment factor. However, in some cases, literature on a topic did not fully conform to an aspect of the outlined criteria. For instance, the preponderance of literature in some technical areas is published as white papers and reports for which independent peer review is not standard practice or is not well documented. To address these areas in which peer-reviewed literature was limited, we cited literature that may not have been peer-reviewed. These references often provided useful background information or corroborated conclusions in the peer-reviewed literature.

Table 1-1. The five factors and accompanying criteria used to evaluate literature and data cited in this assessment.
Criteria are consistent with those outlined by the EPA's Science Policy Council (U.S. EPA, 2003c). Criteria are incorporated into the Quality Assurance Project Plans for this assessment (U.S. EPA, 2014d, 2013d).

Factor	Criteria
Applicability	Document provides information useful for assessing the potential pathways for hydraulic fracturing activities to change the quality or quantity of drinking water resources, identifies factors that affect the frequency and severity of impacts, or suggests ways that potential impacts may be avoided or reduced.
Review	Document has been peer-reviewed.
Soundness	Document relies on sound scientific theory and approaches, and conclusions are consistent with data presented.
Clarity/completeness	Document provides underlying data, assumptions, procedures, and model parameters, as applicable, as well as information about sponsorship and author affiliations.
Uncertainty/variability	Document identifies uncertainties, variability, sources of error, and/or bias and properly reflects them in any conclusions drawn.

1.4.4 Quality Assurance and Peer Review

The use of quality assurance (QA) and peer review helps ensure that the EPA conducts high-quality science that can be used to inform policymakers, industry, and the public. Quality assurance activities performed by the EPA ensure that the agency's environmental data are of sufficient quantity and quality to support the data's intended use. The EPA prepared a programmatic Quality

Management Plan (U.S. EPA, 2014e) for all of the research conducted under the EPA's Study Plan, including the review and synthesis of the scientific literature in this assessment. The hydraulic fracturing Quality Management Plan describes the QA program's organizational structure; defines and assigns QA and quality control (QC) responsibilities; and describes the processes and procedures used to plan, implement, and assess the effectiveness of the quality system. The broad plan is then supported by more detailed QA Project Plans (QAPPs). The QAPPs developed for this assessment provide the technical approach and associated QA/QC procedures for our data and literature search and evaluation strategies introduced in Section 1.4.2 and 1.4.3 (U.S. EPA, 2014d, 2013d). A QA audit was conducted by the QA Manager during the preparation of this assessment to verify that the appropriate QA procedures, criteria, reviews, and data verification were adequately performed and documented. Identifying uncertainties is another aspect of QA; uncertainty, including data gaps and data limitations, is discussed throughout this assessment.

This report is classified as a Highly Influential Scientific Assessment (HISA), which is defined by the Office of Management and Budget (OMB) as a scientific assessment that (1) could have a potential impact of more than $500 million in any year or (2) is novel, controversial, or precedent-setting or has significant interagency interest (OMB, 2004). The OMB describes specific peer review requirements for HISAs. To meet these requirements, the EPA often engages the SAB as an independent federal advisory committee to conduct peer reviews of high-profile scientific matters relevant to the agency. Members of an ad hoc panel, the same panel that was convened under the auspices of the SAB to provide comment on the Progress Report, also provided comment on an external review draft of this assessment.[1] Panel members were nominated by the public and chosen to create a balanced review panel based on factors such as technical expertise, knowledge, experience, and absence of any real or perceived conflicts of interest. Both peer review comments provided by the SAB panel (SAB, 2016) and public comments submitted to the panel during their deliberations about the external review draft of this assessment were carefully considered in the development of this final document.

1.5 Organization

This assessment begins with an Executive Summary that summarizes our overall content and conclusions. The Executive Summary is written to be accessible to all members of the public.[2]

This introductory chapter establishes the goals, scope, and approach for the rest of the assessment. Following is a characterization of drinking water resources in the contiguous United States (Chapter 2). Next, we present a general description of hydraulic fracturing activities and the role of hydraulic fracturing in the oil and gas industry in the United States (Chapter 3). Chapter 1 is written

[1] Information about this process is available online at http://yosemite.epa.gov/sab/sabproduct.nsf/02ad90b136fc21ef85256eba00436459/b436304ba804e3f885257a5b00521b3b!OpenDocument.

[2] The terminology used in the data and literature cited in this assessment can be very technical in nature and sometimes inconsistent. An attempt has been made throughout this document to provide definitions of technical terms and to use terminology in a consistent way that enhances understanding of the topics presented for the audiences targeted by each part of the assessment.

to be accessible to all members of the public. Chapters 2 and 3 are written to be accessible to an audience with general science knowledge.

Chapters 4 through 8 are organized around the stages of the hydraulic fracturing water cycle (Figure 1-1) and address the potential for activities conducted during those stages to change the quality or quantity of drinking water resources. Each stage is covered by a separate chapter. There is also a chapter devoted to an examination of the properties of the chemicals and constituents in hydraulic fracturing-related fluids (Chapter 9). These chapters are written to be accessible to an audience with a moderate amount of technical training and expertise in the respective topic areas.

The final chapter provides a synthesis of the information in the assessment (Chapter 10). This chapter is written to be accessible to an audience with general science knowledge.

The appendices supply information that support the chapters of the assessment. This includes an appendix with a table of all individual products published under the EPA's hydraulic fracturing study and cited in this assessment, as well as answers to the research questions posed in the Study Plan (Appendix A). These answers were informed by the products of the study and the data and literature reviewed in this assessment.

1.6 Intended Use

This state-of-the-science assessment will contribute to the understanding of the potential impacts of activities in the hydraulic fracturing water cycle on drinking water resources and the factors that influence those impacts. The data and findings can be used by federal, tribal, state, and local officials; industry; and the public to better understand and address vulnerabilities of drinking water resources to hydraulic fracturing activities.

We expect this report will be used to help facilitate and inform dialogue among interested stakeholders, including Congress, other federal agencies, states, tribal governments, the international community, industry, NGOs, academia, and the general public. Additionally, the identification of knowledge gaps will promote greater attention to these areas by researchers.

This report may support future assessment efforts. We anticipate that it could contribute context to site-specific exposure or risk assessments of hydraulic fracturing, to regional public health assessments, or to assessments of cumulative impacts of hydraulic fracturing on drinking water resources over time or over defined geographic areas of interest.

Finally, and most importantly, this assessment presents the science to inform decisions by federal, state, tribal, and local officials; industry; and the public on how best to protect drinking water resources now and in the future.

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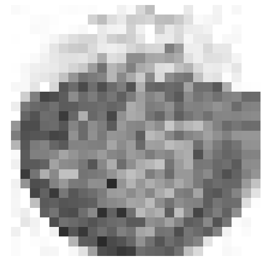
Chapter 2. Drinking Water Resources in the United States



Abstract

In this assessment, drinking water resources are defined as any body of groundwater or surface water that now serves, or in the future could serve, as a source of drinking water for public or private use. An estimated 86% of the United States population derives its household drinking water from public water systems (PWSs), which mostly use surface water sources, while nearly all of the remaining 14% of people self-supply their drinking water from groundwater.

Future access to high-quality drinking water in the United States will likely be affected by changes in climate and water use. The existing distribution and abundance of the drinking water resources may not be sufficient in some locations to meet future demand. Since 2000, about 30% of the total area of the contiguous United States has experienced moderate drought conditions and about 20% has experienced severe drought conditions, which often correlates with diminishment of drinking water supplies. As a result, non-fresh water resources, such as wastewater from sewage treatment plants, brackish surface water and groundwater, and seawater are increasingly treated and used to meet the demand for drinking water.

Hydraulically fractured oil and gas production wells can be located near drinking water sources. Between 2000 and 2013, approximately 3,900 PWSs had between one and 144 wells hydraulically fractured within 1 mile of their water source; these PWSs served more than 8.6 million people year-round in 2013. An additional 740,000 people self-supply their drinking water in counties where at least 30% of the population relies on groundwater and where there were at least 400 hydraulically fractured wells. Belowground, hydraulic fracturing can occur in close vertical proximity to drinking water resources. Available data show that depths to hydraulically fractured rock formations containing oil and gas resources can range from less than 1,000 feet (300 meters) to more than 10,000 feet (3,000 meters), while drinking water resources may be found between a few tens of feet to as much as 8,000 feet (2,000 meters) below the surface. The EPA found that, along individual wellbores, where data were available, the distance between these two resources ranged from no separation to more than 10,000 feet (3,000 meters). There is considerable uncertainty in this range of values, however. In many cases, the lack of accessible information about the depth to the base of formations containing groundwater resources in need of current and future protection prevents calculation of a vertical separation distance.

The locations of drinking water resources relative to hydraulically fractured oil and gas production wells influence the potential for activities in the hydraulic fracturing water cycle to impact drinking water resources. With increased proximity, activities in the hydraulic fracturing water cycle have more potential to affect aboveground and belowground drinking water resources.

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2. Drinking Water Resources in the United States

2.1 Introduction

Drinking water resources provide the water humans consume, cook with, bathe in, and need for other purposes. In this assessment, drinking water resources are considered to be any groundwater or surface water that now serves, or in the future could serve, as a source of drinking water for public or private use. [1] This chapter provides information about drinking water resources in the United States, including current sources and indications of future trends for drinking water resources. Assessment of whether and where activities in the hydraulic fracturing water cycle may impact drinking water resources requires consideration, in part, of the locations of water and oil and gas resources and what physically separates them. More information about oil and gas resources and the areas of the United States where hydraulic fracturing occurs is described in Chapter 3, however this chapter focuses on the lateral (horizontal) and vertical distances between hydraulic fracturing operations and drinking water resources.

2.2 Ground and Surface Water Resources

All drinking water derives from the finite amount of water found on or below the earth's surface. Fresh water serves as the source for most drinking water.[2] To get an idea of the fresh water fraction of all water, this section presents an estimate of the earth's water abundance. Shiklomanov (1993) estimates the amounts of all water on earth, and here these amounts are expressed as the percent of the earth's total water volume:

- Oceans account for about 96.5%.
- Saline groundwater, saline lakes, and water in the form of ice or vapor account for 2.7%.
- Fresh groundwater, swamps, lakes, and rivers account for the remaining 0.8%, of which about 99% is groundwater.

Hydrologic Cycle. The process describing the movement of the earth's water through the atmosphere, land, and oceans is referred to as the hydrologic cycle. Text Box 2-1 describes the hydrologic cycle, including the manner in which the finite amount of water on the earth moves through different locations during the stages of the cycle. On land, surface water and groundwater interact, shown in the text box as surface water infiltrating into the ground, and separately as an example of groundwater flowing into the river. Water consumption (for example when used for agriculture, incorporated into a product, or for drinking purposes), temporarily removes water

[1] In this chapter, a "drinking water *source*" means the body of water is now supplying, or is known to be capable of supplying drinking water.

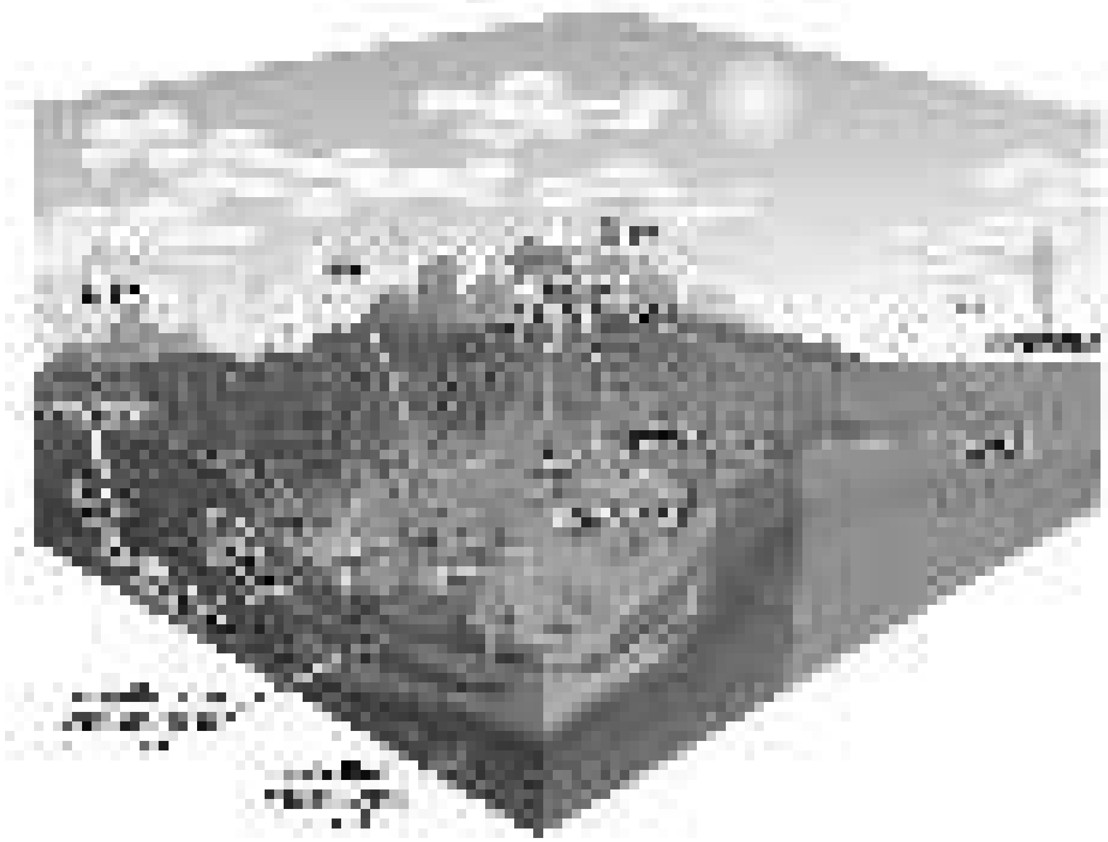
[2] Published estimates of worldwide water supplies, such as by Shiklomanov, do not use a salinity threshold value to define "fresh" or "saline" water. "Fresh" water is characterized in these published estimates as serving as a source for domestic, agricultural, and industrial uses. As described further in Section 2.2.1.1, the term "fresh" in this chapter refers to water having total dissolved solids content up to 3,000 milligrams per liter.

from one local place in the hydrologic cycle, but it may be returned to a different point in the hydrologic cycle. See Chapter 4 for additional discussion of water consumption.

Text Box 2-1. The Hydrologic Cycle.

The finite amount of water and its movement on earth is often called the hydrologic cycle, depicted below. The three basic, and repeating, stages of this cycle include:

1. Rainfall transfers water from the atmosphere into oceans or onto land,
2. Water on land moves among surface water bodies and groundwater, and
3. Evaporation from land and the oceans returns water to the atmosphere.



Rainwater and melted snow collect into rivers, lakes or other water bodies to become surface water, or infiltrates into the ground to become groundwater. Humans drink fresh surface and groundwater, and in some locations, ocean water treated by desalination. Water resides on land or in the ground for varying amounts of time before moving into another of stage of the hydrologic cycle. Residence times for water found in different land locations can range from days to millions of years, depending on the path water takes. Residence time affects water quality on land or in the ground because water dissolves natural earth salts when in contact with those materials. When water on or under land reaches the ocean, its salt content ultimately stays in the ocean because evaporation leaves behind dissolved salt creating freshwater vapor. Evaporation from land and the ocean contribute fresh water to the atmosphere where it can precipitate once again, thus completing a hydrologic cycle. As drawn in this depiction, evaporation includes the release of water vapor from plant leaves that originally entered plant root systems in a process known as transpiration.

2.2.1 Groundwater Resources

Groundwater can be found in the subsurface nearly everywhere, but it varies in quality and quantity. Groundwater exists in that part of the hydrologic cycle where surface water infiltrates through soil into subsurface cracks and voids in rock, creating and sustaining aquifers, a natural process known as groundwater recharge.[1] The opposite natural process from recharge is discharge, where groundwater flows to the surface at springs or through the bottoms of lakes and rivers. Groundwater also includes water trapped in the pores of sedimentary rocks as they were deposited.

The scale of groundwater flow systems can be local, regional, or something in between. Local groundwater flows may be small enough to be measured in the tens of feet while regional groundwater flows may be large enough to be measured in hundreds of miles (Alley et al., 1999). Groundwater movement is related to the rate of groundwater recharge, gravity's effect on the groundwater, and the permeability of the rock through which groundwater flows. Localized groundwater flow tends to occur along shallower flow paths with shorter overall residence times, whereas regional groundwater flow tends to occur along deeper flow paths with longer residence times (Winter et al., 1998). Text Box 2-1 depicts differences between local and regional flow regimes.

The U.S. Geological Survey (USGS) has mapped and described more than 60 principal aquifers in the United States, although these aquifers are not the only occurrences of groundwater (USGS, 2009).[2] Although the depth to the water table can vary from ground surface to a few tens of feet below ground surface, the depth to the base of groundwater can be tens of thousands of feet below ground.[3] The depth to the base of individual principal aquifers can be a relatively uniform or may vary by thousands of feet across the aquifer's areal extent due to sloping geologic formations and/or changes in topography.

2.2.1.1 Groundwater Quality

The quality of groundwater often correlates with its age, which ranges from days to millions of years (Alley et al., 1999; Freeze and Cherry, 1979a; Chebotarev, 1955).[4] As explained in Text Box 2-1, groundwater salinity tends to increase with increasing residence time due to gradual dissolution of contacted earth materials. Some groundwater can become very saline. These waters can result from exposure to soluble sedimentary rocks and/or concentration of salt content due to evaporation of liquid water in the subsurface (Zolfaghari et al., 2016; Levorsen, 1965). It is also possible that sea water was trapped in sediments during deposition in ancient oceans, which were subsequently buried over geologic time. There are instances where groundwater is found at great

[1] An aquifer is a water-bearing geologic formation, group of formations, or part of a formation. Groundwater is the water in an aquifer.

[2] Principal aquifers are defined as a regionally extensive aquifer or aquifer system that has the potential to be used to supply potable water. Principal aquifers in Puerto Rico and the U.S. Virgin Islands are included.

[3] The water table refers to the top, or uppermost surface, of groundwater. Below the water table, the ground is saturated with water.

[4] Groundwater age used here refers to how long the water has been in the ground.

depths but is relatively fresh. This can be caused by groundwater moving from the surface to deep locations relatively quickly with little time to pick up dissolved solids and become saline. This phenomenon is more pronounced in mountains where rainwater or melted snow in upland areas supply groundwater that moves downward through steeply dipping, permeable sedimentary rock layers to reach great depths. Chemicals occurring naturally in groundwater include both inorganic (e.g., salts, metals) and organic (carbon-based) types.

Salinity variation. Salinity is often the principal characteristic used to describe the overall quality of groundwater. The term “fresh” groundwater often means groundwater containing no more than 1,000 milligrams per liter of total dissolved solids (mg/L TDS) but it is sometimes used to refer to groundwater containing no more than 3,000 mg/L TDS (Maupin et al., 2014; U.S. EPA, 2012e; Freeze and Cherry, 1979a). When characterizing groundwater quality, scientists generally consider the relative abundance of sodium, calcium, potassium, magnesium, chloride, bicarbonate, and sulfate to account for the bulk of dissolved constituents (Freeze and Cherry, 1979a). Natural salinity ranges from less than 100 mg/L to over 300,000 mg/L TDS (Lauer et al., 2016; Clark and Veil, 2009). Higher salinity groundwater can contribute to palatability problems, and in the very high salinity ranges, causes water to be unhealthful for human consumption (Ellis, 1997). People have a range of reactions to drinking water salinity. Some people object to the taste of drinking water having comparatively lower salinity levels while other people reach this objection threshold at higher salinity levels (Burlingame and Waer, 2002). Desalinating water containing salinity values of 10,000 mg/L TDS to render it potable is technically and economically feasible (Esser et al., 2015).[1] As a result, groundwater with salinity values up to 10,000 mg/L TDS is often defined as a protected groundwater resource under several laws, including the regulations implementing the federal Safe Drinking Water Act (SDWA) and the U.S. Bureau of Land Management (BLM) Onshore Order #2. The complete basis and standards for defining a protected groundwater in all locations within the United States is beyond the scope of this report. Additional information about protections given to groundwater is described in Chapter 1 in Text Box 1-1.

Groundwater suitable for drinking is found within a large range of depths around the United States. The groundwater quality profile with depth varies around the United States. Feth (1965) described patterns in the relationship of depth to groundwater containing salinity ranging from 1,000 to 3,000 mg/L TDS.[2] The patterns include: (1) large portions of the Southeast and middle Midwest have at least 1,000 ft (300 m) of separation between the land surface and groundwater containing 1,000-3,000 mg/L TDS, and (2) significant portions of the Northeast, northern Midwest, and parts of the West have less than 500 ft (200 m) separating the land surface from groundwater containing 1,000-3,000 mg/L TDS. The report does not contain information about the base or thickness of groundwater having certain quality. As a result, these depths represent minimum distances between the land surface and bottom depth of groundwater having this salinity range.

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[1] For instance, desalination of sea water (approximately 35,000 mg/L TDS) now occurs in Florida, California, and Texas.

[2] Salinity and total dissolved solids are frequently interchangeable terms. The vast majority of dissolved constituents in natural water are inorganic salts, although a minor fraction of dissolved constituents can be organic matter. Feth (1965) maps groundwater found at ranges of depth with spans of salinity. Singular depth and salinity values are not present on the map.

Methane in groundwater. Methane can be found naturally at detectable levels in groundwater (Kappel and Nystrom, 2012; Eltschlager et al., 2001; Coleman et al., 1988). There are different origins of methane in groundwater. Biogenic methane is produced at comparatively low temperature and pressure from biologic decay of carbon-bearing matter, while thermogenic methane is formed over geologic time when carbon-bearing matter is exposed to elevated pressure and temperature conditions typically associated with deep burial (Baldassare et al., 2014). Given the buoyancy of natural gas, if a pathway exists or enough time is available, it can move upward and accumulate at shallower depths. Natural gas found in small, uneconomic quantities in shallow zones may have originated in place or may have migrated upward, and is often referred to as stray gas. For more discussion about the issue of stray gas, see Text Box 6-3 in Chapter 6. When consumed in drinking water, methane does not generally have human health effects,[1] however, it is an explosive gas if it comprises between 5% and 15% of a volume of air (Astle and Weast, 1984). If methane from well water enters the atmosphere within a confined space under conditions that allow it to concentrate, it can pose an explosive threat if it reaches this threshold.

2.2.1.2 Groundwater Quantity

Groundwater quantity can be characterized as the total subsurface water available, although a practical limiting property is the rate at which groundwater can be withdrawn from the subsurface, a property known as yield (Freeze and Cherry, 1979a). If rock formations in the subsurface contain water within exceedingly small or poorly connected pore spaces, then the low yield may preclude its practical use as a source of drinking water.

When recharge and discharge are in balance, the volume of groundwater existing in the subsurface remains the same. Recharge and discharge also occur in connection with human-caused activity. Groundwater recharge increases due to irrigation, underground injection wells, surface impoundments, and dammed reservoirs, while groundwater discharge increases through well withdrawals for irrigation, household use, etc. (Winter et al., 1998). These activities can locally affect the natural balance between groundwater recharge and discharge. Climatic variation that changes precipitation rates also affects groundwater recharge rates, which in turn leads to changes in subsurface groundwater volume (Winter et al., 1998).

When an aquifer consistently yields water at rates suitable for human use, and the water is of good enough quality to drink or be treated for drinking, it can serve as a source of drinking water.

2.2.2 Surface Water Resources

Surface water is that part of the hydrologic cycle that occurs on land surface and includes water in the ocean as well as rainwater or meltwater. Surface water collects into depressions or along channels in sufficient volume to create standing or running water all or much of the time. Non-ocean surface water has often had little time to become saline, because much of it is not in direct contact with anything other than more water in the surrounding surface water body. Non-ocean surface water can quickly move into the next phase of the hydrological cycle, either evaporating

[1] There is no enforceable drinking water standard established for dissolved methane in drinking water.

into the atmosphere or infiltrating the subsurface. Because surface water is open to the atmosphere and is generally located at the lowest points on a landscape, it is susceptible to contamination. Contamination sources include atmospheric deposition, and run-off from urban land areas or lands used for agricultural or industrial activities (Winter et al., 1998). Many non-ocean surface water bodies in the United States have a set of water quality standards based on their designated use, which can include recreation, drinking water, supportive of aquatic life, fishery, industrial supply, and other uses. In turn, National Discharge Pollution Elimination (NPDES) permits governing point source discharge into the surface water bodies are issued under the Clean Water Act and contain limits on pollutants designed to achieve these water quality standards.[1] When taken together, these permits are meant to ensure that the surface water achieves a water quality consistent with the designated use.

2.2.2.1 Surface Water Quality

Studies conducted in connection with the National Water Quality Assessment Program show the presence of human-made chemicals at low concentrations in the streams surveyed (Kingsbury et al., 2008).[2] Based on dissolved solids alone, sampled streams range from less than 100 mg/L TDS to more than 500 mg/L TDS (Anning and Flynn, 2014). Large lakes can range in salinity from less than 500 mg/L TDS to more than 200,000 mg/L. By comparison, ocean water has a salinity of about 35,000 mg/L TDS. Considering the vast array of possible chemical, biological, and radiological content in surface water, it is beyond the scope of this report to describe in detail the surface water qualities that exist in the United States.

2.2.2.2 Surface Water Quantity

About 7% of the surface area of the United States is covered by surface water, but it is not uniformly distributed. The portion of the United States located east of the Mississippi River comprises about 25% of the total area, yet it contains about 42% of the total land area covered by surface water (USGS, 2016; U.S. Census Bureau, 2012). The Great Lakes alone, located in the eastern half of the United States, contain about one-fifth of the world's surface fresh water (Government of Canada and U.S. EPA, 1995).[3] In contrast, the western part of the United States has a lower proportion of land covered by surface water with streams that tend to be more intermittent in nature.[4] For instance, 81 percent of the streams in Arizona, New Mexico, Nevada, Utah, Colorado, and California are not permanent streams (Levick et al., 2008). Certain parts of the western U.S. are presently experiencing less surface water availability as indicated by declining water reservoir levels with some reservoirs in the southwest currently below 50% of their capacity.[5] For example, according to

[1] Title 40, United States Code of Federal Regulations, Part 131, as of May 25, 2016.

[2] See USGS (2012) for more information about this program.

[3] Including the portion of the Great Lakes lying within Canada.

[4] Not all western states follow this trend. Hawaii and Alaska, for instance, have a significantly higher percentage of land mass covered by surface water (41% and 14%, respectively) than the national average.

[5] See for instance U.S. DOI (2016b), California Department of Water Resources (2016), and SRP (2016).

the U.S. Department of the Interior (DOI), the largest capacity reservoir in the United States, Lake Mead, holds about 37% of its volume capacity as of the fall of 2016 (U.S. DOI, 2016a).

2.3 Current Drinking Water Sources

Drinking water is supplied to households and businesses by either public water systems (PWSs) or non-public systems (non-PWSs).[1] In 2010, approximately 270 million people (86% of the population) in the United States relied on PWSs to supply their homes with drinking water (Maupin et al., 2014; U.S. EPA, 2013b). These PWSs provided households with nearly 24 billion gal (91 billion L) of water per day (Maupin et al., 2014).[2] In areas without service by PWSs, approximately 45 million people (14% of the population) obtain drinking water from non-PWSs, using mostly water wells. Non-PWSs account for about 3.6 billion gal (14 billion L) of daily water withdrawals (Maupin et al., 2014).[3]

Both groundwater and surface water serve as drinking water sources in the United States. Surface water accounts for about 58% of all drinking water withdrawals and groundwater supplies the remaining 42%. Table 2-1 portrays the relative abundance of surface water and groundwater as sources for both publicly and non-publicly supplied drinking water.

Of the population receiving water supplied by PWSs, the relative importance of surface and groundwater sources for supplying drinking water varies geographically (Figure 2-1). Most larger PWSs rely on surface water and are located in urban areas (U.S. EPA, 2011c), whereas most smaller PWSs rely on groundwater and are located in rural areas (U.S. EPA, 2014h, 2013b). More than 95% of households in rural areas obtain their drinking water from groundwater (U.S. EPA, 2011c).

PWSs are subject to routine monitoring and testing requirements required under the National Primary Drinking Water Standards regulations, whereas no such monitoring or testing is required for non-PWSs.[4] The required monitoring and testing at PWSs ensures that the public has information regarding the extent to which delivered water meets drinking water standards, whereas users of non-PWSs (e.g., private water wells) make individual, voluntary decisions about how often they monitor and test their water. Lack of monitoring may make non-PWS users more vulnerable to contamination, if present, than PWS users.

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[1] PWSs provide water for human consumption from surface water or groundwater through pipes or other infrastructure to at least 15 service connections or serve an average of at least 25 people for at least 60 days a year (U.S. EPA, 2012g). The EPA categorizes PWSs as either community water systems, which supply water to the same population year-round, or non-community water systems, which supply water to at least 25 of the same people at least six months per year, but not year-round. Non-public water systems (non-PWSs) have fewer than 15 service connections and serve fewer than 25 individuals (U.S. EPA, 1991). Non-PWSs are often private water wells supplying drinking water to a singular residence.

[2] The USGS compiles data in cooperation with local, state, and federal environmental agencies to produce water-use information aggregated at the county, state, and national levels. Every five years, data at the county level are compiled into a national water use census and state-level data are published. The most recent USGS water use report was released in 2014, and contains water use estimates from 2010. Water withdrawals are distinguished from and are greater than water deliveries due to water loss during the process of delivering finished water (Maupin et al., 2014; USGS, 2014b).

[3] A withdrawal means the volume of water taken from its source regardless of how much of that volume is either returned to the local hydrologic cycle or is consumed without being returned to the local hydrologic cycle.

[4] See Title 40 of the Code of Federal Regulations, Part 141, promulgated pursuant to the SDWA.

Table 2-1. Summary of drinking water sources in the United States in 2010.
The volume and percentages of daily domestic water withdrawals in the United States are shown by public and non-public water systems, total withdrawal, and whether the source is surface water or groundwater. Volume is in billions of gallons per day (Bgal/day) and percentages are of either water supply type or total volume withdrawn, as indicated in italics. Some figures shown are rounded values. Source of data: Maupin et al. (2014).

Drinking water source	Public water supply	Non-public water supply	Total volume withdrawn
Surface Water			
Daily volume withdrawn (billion gallons)	26.3	0.1	26.4
Percent of water supply type	63	2	58
Groundwater			
Daily volume withdrawn (billion gallons)	15.7	3.5	19.2
Percent of water supply type	37	98	42
Total			
Daily volume withdrawn (billion gallons)	42.0	3.6	45.6
Percent of water supply type	92	8	100

Figure 2-1. Geographic variability in drinking water sources for public water systems.
The relative importance of surface and groundwater as sources for public water systems varies by state. The public water system sources used in this analysis include infiltration galleries, intakes, reservoirs, springs, and wells. Sources: ESRI (2010), U.S. Census Bureau (2013), and U.S. EPA (2013b).

2.3.1 Factors Affecting How Water Becomes a Drinking Water Source

The most common source of drinking water in the world, including in the United States, is fresh water (see Section 2.2.1.1). There can be exceptions to the use of fresh water as a drinking water source. For instance, projects in California, Florida, Arizona and Texas desalinate sea water or brackish groundwater to produce drinking water.[1] The principle of supply and demand that affects availability of commercial products in the marketplace is also applicable to drinking water resources. Water not considered a practical drinking water source under one demand condition may become desirable as a drinking water source under different demand conditions. Text Box 2-2 presents El Paso, Texas as such an example.

Text Box 2-2. El Paso's Use of Higher Salinity Water for Drinking Water.

The El Paso Water Utility (EPWU) provides drinking water to over 600,000 people in the City of El Paso, Texas and surrounding communities. Historically, the EPWU has withdrawn surface water from the Rio Grande River and groundwater to meet water needs. Salinity from the freshwater aquifers typically ranges between 300 and 1,000 mg/L TDS. With increases in population and periodic drought conditions stressing the water supply, the EPWU instituted a number of different measures to diversify its water supply portfolio. Components of the EPWU water supply portfolio include water conservation, surface water, groundwater and, more recently, desalinating saline groundwater. Continued long-term pumping of fresh groundwater allowed higher salinity groundwater to enter into one of EPWU's well fields from more saline parts of the aquifer. This well field is now used as the source for the Kay Bailey Desalination Plant, which began operation in 2007 and desalinates groundwater with salinity ranging from 1,000 and 5,000 mg/L TDS (El Paso Water Utilities, 2016). The plant uses reverse osmosis technology to remove the high salt content thereby creating additional fresh water supplies. Use of this higher salinity water supply has added approximately 25% more water availability, decreasing the stress on the original fresh water supplies available to the EPWU and highlights the potential value of groundwater that had not formerly been considered a drinking water source.

2.3.1.1 General Considerations Applicable to All Water as Source of Drinking Water

Factors to consider when assessing a possible source of drinking water include availability, contaminants in the water, and the cost to obtain and treat water. Surface water in streams, lakes, or reservoirs is almost always considered to be a source for drinking water, because they contain fresh, readily accessible water. Groundwater is a critically important drinking water source in many parts of the United States, especially where surface water is less abundant. Challenges for use as drinking water exist for both surface and groundwater. Surface water may not suffice as a drinking water source when it exists only temporarily or cannot supply the volume demand. Both surface water and groundwater may have contaminant levels that require expensive treatment technology. For instance, in an extensive report, the USGS describes how human activities cause unnaturally fast and deep groundwater movement, which degrades water quality over long periods in the

[1] Brackish water is often a general term used for water having a salinity content intermediate between fresh water and sea water, although it may also have a more specific definition, such as the 1,000 – 10,000 mg/L TDS value used in some USGS publications.

nation's principal aquifers (DeSimone et al., 2014). Despite these challenges, changes in the demand for water affect the consideration of sources of water for drinking purposes.

2.3.1.2 Considerations Applicable to Groundwater as a Drinking Water Source

Determining what groundwater is eligible for use as a drinking water source can include additional challenges. Groundwater may be located at significant depth or within low-yield aquifers, requiring additional engineering solutions to make them practical and/or cost effective as a drinking water source. Aquifers, or parts of aquifers, not in use today for drinking water purposes may nonetheless eventually be considered a drinking water source. The future viability of currently unused aquifers depends on the definition of what constitutes a drinking water resource and knowledge of the physical and chemical characteristics of the aquifers. The extent of knowledge about what exists in the subsurface depends on extrapolation from limited subsurface data (e.g., water samples collected from wells in, or passing through, aquifers). Although salinity is a common criterion for designating an aquifer as a drinking water resource (see Section 2.2.1.4), there is not a uniform threshold value for making that determination. The Groundwater Protection Council (GWPC) notes:

> There is a great deal of variation between states with respect to defining protected groundwater. The reasons for these variations relate to factors such as the quality of water, the depth of Underground Sources of Drinking Water, the availability of groundwater, and the actual use of groundwater (GWPC, 2009).[1]

In addition to variation in applicable water quality criteria, the availability of information regarding groundwater that meets an applicable criterion (if one exists) is also variable. For instance, the bottom depth of aquifers or parts of aquifers that may be defined as a drinking water resource are not always readily publicly available. In some locations, such as the State of Texas, estimates of the bottom depth of groundwater meeting certain regulatory threshold criteria are made public on a website.[2] In other parts of the United States the depth of identified protected subsurface drinking water resources may not be publicly available. No centralized compilation of groundwater depth and quality exists for all locations in the United States, nor does such a reference exist for depths to protected groundwater resources. The depths to protected groundwater resources can vary. In one example, the EPA described the reported bottom depths of protected groundwater resources as ranging from just below ground surface to 8,000 ft (2,000 m) (U.S. EPA, 2015n).[3]

Even in regions where the bottom depth of protected groundwater resources are generally known, there can remain uncertainty regarding precise depths at specific locations. Examples include the states of Indiana and Michigan according to the EPA Region 5 Underground Injection Control (UIC)

[1] An underground source of drinking water (USDW) is defined in the federal regulations that implement the UIC program. A USDW is generally considered to be any aquifer, or its portion, that currently serves as a source for a public water system; or which contains enough groundwater to supply a public drinking water system, and either now supplies water for human consumption, or contains fewer than 10,000 mg/L TDS. See Title 40 of the Code of Federal Regulation, Section 144.3.

[2] See http://www.beg.utexas.edu/sce/index.html.

[3] This reference provided 1,000-foot (305 meters) depth resolution for the reported base of protected groundwater.

program, the State of Utah according to the Utah Geological Survey, and the State of California according to the California State Water Resources Board (Esser et al., 2015; Anderson et al., 2012; U.S. EPA, 2012e). In these examples, the depth to groundwater meeting the salinity threshold necessary for decision-making is stated not to be known with precision, and collection of additional groundwater quality information is advised.[1]

2.4 Future Drinking Water Sources

The future availability of fresh drinking water sources in the United States (Section 2.2.1.1) will likely be affected by changes in climate and water use (Georgakakos et al., 2014). Since 2000, about 30% of the total area of the contiguous United States has experienced moderate drought conditions and about 20% has experienced severe drought conditions (National Drought Mitigation Center, 2015; U.S. EPA, 2015p). Declines in surface water resources have already led to increased withdrawals and cumulative net depletions of groundwater in some areas (Castle et al., 2014; Georgakakos et al., 2014; Konikow, 2013; Famiglietti et al., 2011). Loss of approximately 240 mi^3 (1,000 km^3) of groundwater between 1900 and 2008 has been documented by the USGS. USGS reports that about 20% of that loss occurred in the final eight years of that timeframe and that depletion is greater in the arid and semi-arid western states than in the more humid eastern states (Konikow, 2013). Other sources of water that might not be considered fresh, such as wastewater from sewage treatment plants, brackish and saline surface and groundwater, as well as sea water, are also increasingly being used to meet water demand. Through treatment or desalination, these water sources can reduce the use of high-quality, potable fresh water for industrial processes, irrigation, recreation, and toilet flushing (i.e., non-potable uses). In addition, in 2010, approximately 355 million gal per day (1.3 billion L per day) of treated wastewater was reclaimed through potable reuse projects (NRC, 2012). Such projects use reclaimed wastewater to augment surface drinking water sources or to recharge aquifers that supply drinking water to PWSs (NRC, 2012; Sheng, 2005). In 2007, among approximately 13,000 desalination plants worldwide, there existed the capacity to produce about 14.7 billion gal (55.6 billion L) of fresh water each day. In 2005, the United States had approximately 11 % of that volume capacity (Gleick, 2008; Cooley et al., 2006).

An increasing number of states are developing new water supplies to augment existing drinking water sources through reuse of reclaimed water, recycling of storm water, and desalination (U.S. GAO, 2014). Most desalination programs currently use brackish water as a source, although plans are underway to expand the use of sea water. States with the highest installed capacity for desalination include Florida, California, Arizona, and Texas (Cooley et al., 2006). It is likely that various water treatment technologies will continue to expand drinking water sources beyond those that are currently being considered. In addition to treatment technologies, there are efforts by public water systems to alleviate demand on drinking water supplies such as encouraging more modest consumer water usage and repairing leaks in water infrastructure.

[1] Decisions dependent on knowledge of threshold salinity values in groundwater can include permitting injection wells and oil and gas production well construction design approvals.

2.5 Proximity of Drinking Water Resources to Hydraulic Fracturing Operations

Hydraulic fracturing in oil and gas production wells necessarily takes place where oil and gas resources are located. The relative locations of drinking water resources influences the degree to which they may be affected by activities in the hydraulic fracturing water cycle. With increased proximity, hydraulic fracturing activities have a greater potential to affect surface and subsurface sources of current and future drinking water (Vengosh et al., 2014; Entrekin et al., 2011). To estimate potentially vulnerability populations that use drinking water resources, the EPA performed an analysis of the number of hydraulically fractured production wells that are located within 1 mi (1.6 km) of a PWS source. The EPA also presents subsurface separation distances between the depths of drinking water resources and hydraulic fracturing in production wells.

2.5.1 Lateral Distance between Public Water System Sources and Hydraulic Fracturing

The EPA analyzed the locations of the approximately 275,000 oil and gas wells that were assumed to be hydraulically fractured in 25 states between 2000 and 2013 (Chapter 3) to determine the number of fractured wells within a 1-mile radius of facilities that withdraw water for a PWS.[1,2,3] Based on 2000–2013 DrillingInfo data, the lateral distance from the nearest facility that withdraws water for PWS to a hydraulically fractured well ranged from 0.01 to 41 mi (0.02 to 66 km), with an average distance of 6.2 mi (10.0 km) and a median distance of 4.8 mi (7.7 km) (DrillingInfo, 2014a; U.S. EPA, 2014h). Of the approximately 275,000 wells that were estimated to have been hydraulically fractured in 25 states between 2000 and 2013, an estimated 21,900 (8%) were within 1 mile of at least one PWS groundwater well or surface water intake. Most of these approximately 6,800 individual facilities that withdraw water for a PWS were located in Colorado, Louisiana, Michigan, North Dakota, Ohio, Oklahoma, Pennsylvania, Texas, and Wyoming (Figure 2-2). These facilities that withdraw water for a PWS had an average of seven hydraulically fractured production wells and a maximum of 144 such production wells within a 1-mile radius. These water sources supplied water to 3,924 PWSs—1,609 of which are community water systems—that served more than 8.6 million people year-round in 2013 (U.S. EPA, 2014h; U.S. Census Bureau, 2013; U.S. EPA, 2013b).[4]

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[1] The EPA estimated the number of oil and gas production wells hydraulically fractured between 2000 and 2013. To do this, EPA assumed that all horizontal wells were hydraulically fractured in the year they started producing and assumed that all wells within a shale, coalbed, or low-permeability formation, regardless of well orientation, were hydraulically fractured in the year they started producing. More details are provided in U.S. EPA (2013c). Not all coalbed methane wells are hydraulically fractured, but coalbed methane wells represent production wells that sometimes uses hydraulic fracturing. Given that there were 15% of coalbed methane wells relative to all hydraulically fractured wells and the lack of data that distinguishes whether or not coalbed wells are hydraulically fractured, EPA included coalbed wells into all counts of wells that are hydraulically fractured.

[2] The selected 1-mile distance used in this analysis provides a consistent approach. Local topographic conditions could support the use of a different analysis at any specific site.

[3] A facility that withdraws water for a PWS includes water intakes, water wells, springs, infiltration galleries, and reservoirs. It is common for a PWS to operate multiple individual facilities to withdraw the cumulative water supplied by the PWS.

[4] All PWS types were included in the locational analyses performed. However, only community water systems were used to calculate the number of customers obtaining water from a PWS with at least one source within 1 mile of a hydraulically

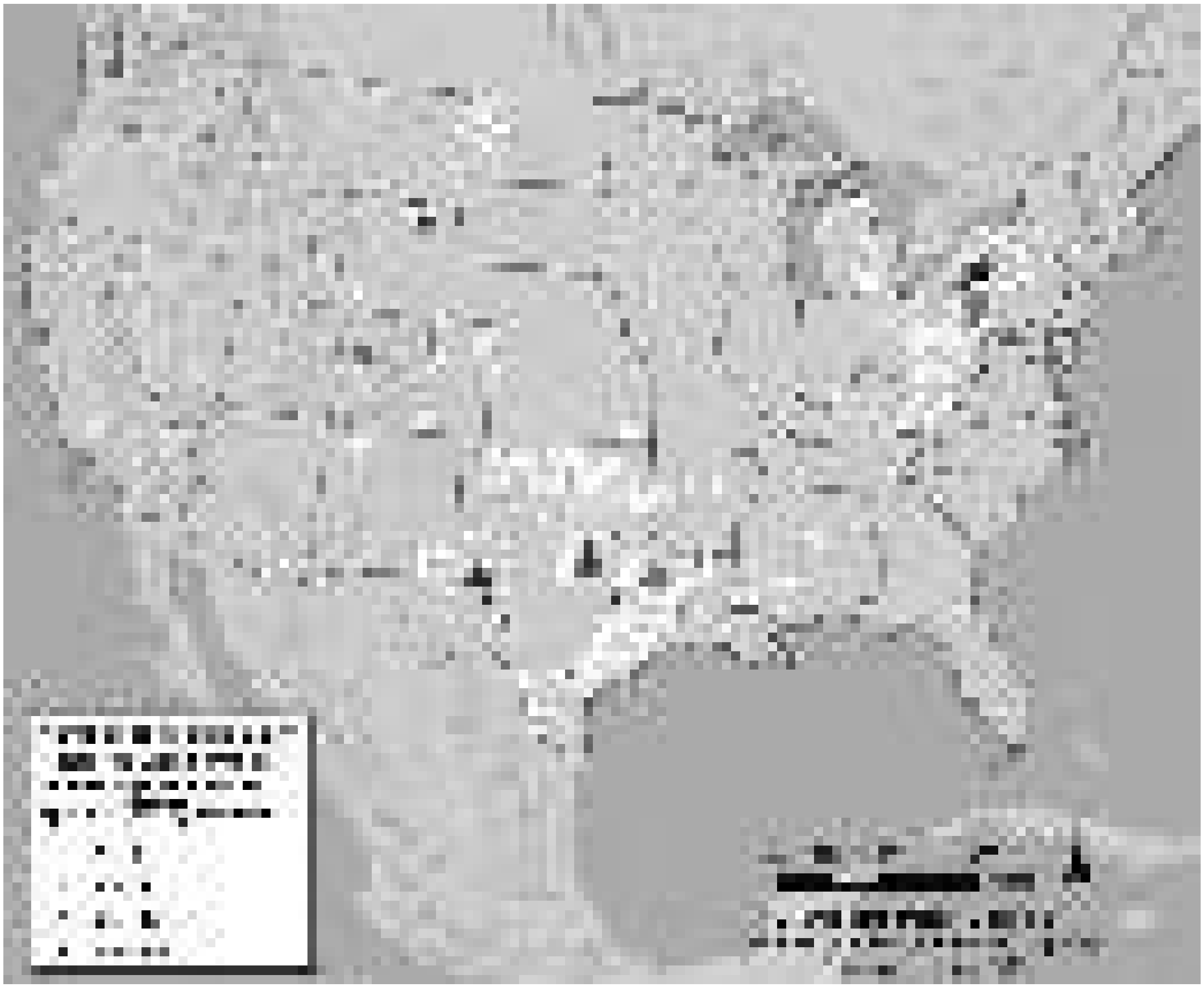

Figure 2-2. The location of public water system sources having hydraulically fractured wells within 1 mile.

Points indicate the location of public water system (PWS) sources; point color indicates the number of hydraulically fractured wells within 1 mile of each PWS source. The estimates of wells hydraulically fractured from 2000 to 2013 developed from the DrillingInfo data were based on assumptions described in Chapter 3. Sources: DrillingInfo (2014), U.S. EPA (2013b), and ESRI (2010).

The EPA also analyzed the location of hydraulically fractured wells relative to populations where a high proportion (≥30%, or at least twice the national average) obtain drinking water from non-PWSs (mostly private groundwater wells).[1] Based on DrillingInfo well location data and USGS drinking water data, between 2000 and 2013, approximately 3.6 million people live in counties

fractured well. If non-community water systems are included, the estimated number of customers increases by 533,000 people (U.S. EPA, 2012g). A community water system is a PWS which serves at least 15 service connections used by year-round residents or regularly serves at least 25 year-round residents.

[1] There is no national data set of non-PWSs. In Maupin et al. (2014), the USGS estimates the proportion of the population reliant on non-PWSs, referred to as the "self-supplied population," by county, based on estimates of the population without connections to a public water system. The USGS estimates were used for this analysis.

with at least one hydraulically fractured well and where at least 30% of the population relies on non-PWSs for drinking water (DrillingInfo, 2014a; USGS, 2014b). The population changes to approximately 740,000 people living in counties with more than 400 hydraulically fractured wells and at least 30% of the population relies on non-PWSs for drinking water (DrillingInfo, 2014a; USGS, 2014b).[1] The counties having more than 400 hydraulically fractured wells and at least 30% of the population relying on non-PWSs for drinking water were located in Colorado, Kentucky, Michigan, Montana, New Mexico, New York, Oklahoma, Pennsylvania, Texas, and Wyoming.

As described in Chapter 1, this assessment defines five stages in the hydraulic fracturing water cycle. The lateral distance analysis presented here relates to the wellhead locations of hydraulically fractured production wells, and therefore addresses three stages that take place near production wellheads, evaluated in Chapters 5, 6, and 7, respectively (chemical mixing, well injection, and produced water handling).[2] A lateral distance analysis was not possible for the other two stages (water acquisition, wastewater disposal and reuse) because there is a lack information about where water is acquired for hydraulic fracturing and where the wastewater from any given hydraulically fractured well is disposed or treated.

2.5.2 Vertical Distance between Drinking Water Resources and Hydraulic Fracturing

The depth at which hydraulic fracturing takes place varies depending on the depth to the targeted production zone. For instance, in a study of wells representing approximately 23,000 production wells hydraulically fractured by nine service companies during 2009 and 2010, the EPA found that, when measured vertically from the surface to total depth, well depths ranged from less than 2,000 ft (600 m) to more than 11,000 ft (3,000 m) (U.S. EPA, 2015n). Similarly, based on more than 38,000 hydraulic fracturing disclosures to the FracFocus registry website, the middle 90% of these well disclosures had vertical depths between 2,900 and 13,000 ft (880 and 4,000 m) with a median value of about 8,100 ft (2,500 m) (U.S. EPA, 2015a). Hydraulic fracturing can occur at or near the bottom of a production well or it may take place at different intermediate depths depending on the location of economically producible oil and gas, and thus the total vertical depth of a production well does not necessarily correlate to the depth at which hydraulic fracturing occurs (Chapter 6). Hydraulic fracturing has been conducted at depths ranging from less than 1,000 ft (300 m) to greater than 10,000 ft (3,000 m) depth (U.S. EPA, 2015n; NETL, 2013). The distance from the base of the drinking water resource to the shallowest hydraulic fracturing initiation point in a production well serves as a separation distance.[3] The EPA reports separation distances in depth measured along the well ranging from no separation distance (where hydraulic fracturing took

[1] Approximately 14% of the U.S. population is self-supplied by non-PWSs (Maupin et al., 2014). This analysis considers only counties in which more than double the national average—that is, at least 30% of the county's population—was supplied by non-PWSs.

[2] Chapter 7 (Produced Water Handling) examines potential effects on drinking water resources at hydraulically fractured wellhead locations, as well as away from wellhead locations.

[3] If measured vertically from the shallowest hydraulic fracturing initiation point to the bottom of the drinking water resource, this is referred to as a vertical separation distance. If measured along a borehole from the shallowest hydraulic fracturing initiation point to the bottom of the drinking water resource, this is referred to as a separation distance in measured depth.

place at depths shallower than the reported base of the drinking water resource) to more than 10,000 ft (3,000 m) (U.S. EPA, 2015n).

In a given setting, it is the geologic and hydrologic history that determines the depths to potential oil and gas and/or subsurface drinking water resources. In some settings, rock formations bearing economic quantities of oil or gas also contain groundwater that, based on salinity value alone, qualifies it as a drinking water resource. Large distances vertically separate these two resources in other settings. Figure 2-3 depicts two different types of these settings.

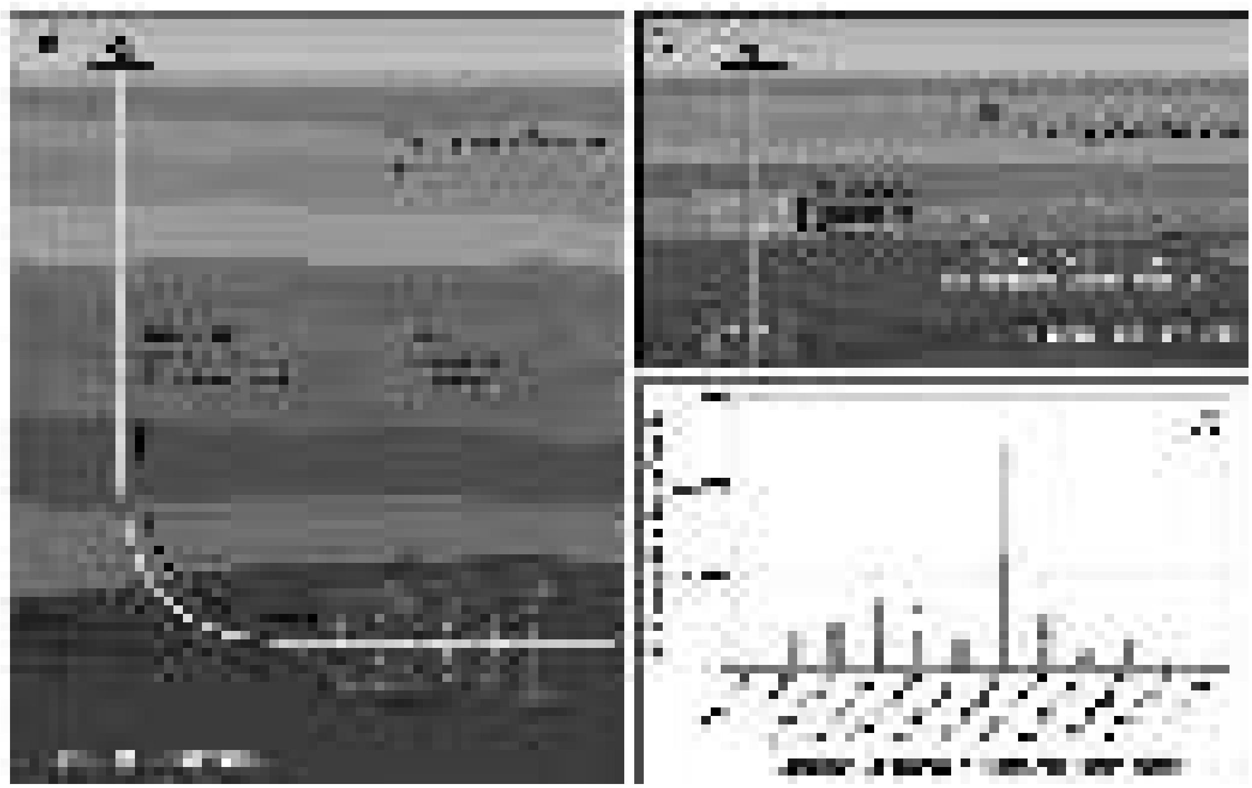

Figure 2-3. Separation distance between drinking water resources and hydraulically fractured intervals in wells

Schematic examples showing a relatively large separation distance (panel a) and the absence of any separation distance (panel b) between the shallowest fracture initiation depth in a well to the base of the protected drinking water resource. Distances may be presented as vertical or as a measured distance along a non-vertical well. Panel c shows result from wells studied representing approximately 23,000 production wells hydraulically fractured between 2009 and 2010 (U.S. EPA, 2015n). Error bars in panel c display 95% confidence intervals.

In Figure 2-3, panel (a), the hydraulically fractured oil- and gas-bearing zone is much deeper than drinking water resources, therefore separation distance is large. In panel (b), the hydraulically fractured oil- and gas-bearing zone is at the same depth as drinking water resources and there is no separation. The lack of separation distance can be due to the oil- and gas-bearing zone being shallow and/or the drinking water resource being deep. Panel (c) illustrates the distribution of separation distances in measured depth for study wells representing approximately 23,000 oil and gas production wells hydraulically fractured by nine service companies between 2009 and 2010, as reported in U.S. EPA (2015n). The calculation of 95% confidence intervals shown in panel (c) is described in the EPA report and was affected by the number of companies in the study and the well file selection methods.

2.6 Conclusions

Drinking water resources provide the water humans consume, cook with, bathe in, and need for other purposes. An estimated 86% of the United States population derives its household drinking water from PWSs that serve at least 25 people. The remaining 14% self-supply their homes with drinking water from non-PWSs, which are largely private water wells. Publicly supplied drinking water is subject to monitoring and testing to determine compliance with drinking water standards while no such monitoring and testing is required at non-PWSs. Surface water is the source for an estimated 58% of the volume needed to supply drinking water and groundwater is the source for the remaining 42%.

The existing distribution and abundance of the drinking water resources in the United States may not be sufficient in some locations to meet future demand. The future availability of sources of drinking water that are considered fresh will likely be affected by changes in climate and water use. Since at least 2000, many areas of the United States have experienced significant drought, which often correlate with diminishment of ground and surface water supplies in these areas. Locally, measures are now being implemented to prolong use of current drinking water sources such as encouraging more modest drinking water use and using treated wastewater or other non-potable water sources to help meet demand.

Between 2000 and 2013, the EPA estimates there were approximately 275,000 oil and gas production wells hydraulically fractured in 25 states. To produce a consistent measure of proximity between these hydraulically fractured oil and gas production wells and drinking water resources during this time frame, the EPA counted the number hydraulically fractured oil and gas production wells located within 1 mile of public drinking water sources, and performed a count of the counties with a relatively high reliance on self-supplied drinking water that also contain one or more of these hydraulically fractured production wells. Between 2000 and 2013, approximately 3,900 public water systems had between one and 144 wells hydraulically fractured within 1 mile of their water source; these public water systems served more than 8.6 million people year-round in 2013. An additional 740,000 people between 2000 and 2013 self-supplied their drinking water in counties where at least 30% of the population relies on groundwater and having at least 400 hydraulically fractured wells.

Depending on the nature of the geologic setting, hydraulically fractured oil and gas production wells can be located near where people get their drinking water. Depths to hydraulically fractured oil and gas resources can range from less than 1,000 ft (300 f) to more than 10,000 ft (3,000 m) while drinking water resources may be found between a few tens of feet to as much as 8,000 ft (2,000 m) below the surface. There is limited publicly available information to determine the vertical distance separating the shallowest hydraulic fracturing initiation point in a production well from the deepest drinking water resource. The EPA found, among 323 wells studied statistically representing more than 23,000 production wells hydraulically fractured by nine service companies between 2009 and 2010, the distance along the wells between these two resources ranged from none to more than 10,000 ft (3,000 m).

Chapter 3. Hydraulic Fracturing for Oil and Gas in the United States

Abstract

This chapter provides a general description of the practice of hydraulic fracturing, where it is conducted, how prevalent it is, and how hydraulic fracturing-based oil and gas production fits into the context of energy production in the United States. Some of the information in this chapter also serves as an introduction to the more in-depth technical chapters in the assessment.

Hydraulic fracturing is a technique used to increase oil and gas production from underground oil- and/or gas-bearing rock formations (reservoirs). The technique involves the injection of hydraulic fracturing fluids through the production well and into the reservoir under pressures great enough to fracture the reservoir rock. Hydraulic fracturing fluids typically consist mainly of water, a “proppant” (typically sand) that props open the created fractures, and additives (usually chemicals) that modify the properties of the fluid for fracturing. Fractures created during hydraulic fracturing enable better flow of oil and gas from the reservoir into the production well. Water that naturally occurs in the oil and gas reservoirs also typically flows into and through the production well to the surface as a byproduct of the oil and gas production process.

Since the mid-2000s, the combination of modern hydraulic fracturing and directional drilling has become widespread and significantly contributed to a surge in oil and gas production in the United States. Slightly more than 50% of oil production and nearly 70% of gas production in 2015 is estimated to have occurred using hydraulic fracturing. Hydraulic fracturing is widely used in unconventional (low permeability) oil and gas reservoirs that include shales, so-called tight oil and tight gas formations, and coalbeds, but it is also used in conventional reservoirs.

There is no comprehensive national database of wells that are hydraulically fractured in the United States. Using data from several commercial and public sources, the EPA estimates that 25,000 to 30,000 new wells were drilled and hydraulically fractured in the United States annually between 2011 and 2014. These hydraulic fracturing wells are geographically concentrated; in 2011 and 2012 almost half of hydraulic fracturing wells were located in Texas, and a little more than a quarter were located in the four states of Colorado, Pennsylvania, North Dakota, and Oklahoma.

New drilling activity for hydraulic fracturing wells is generally linked with oil and gas prices, and those peaked in the United States between 2005 and 2008 for gas and between 2011 and 2014 for oil. Following price declines, the number of new hydraulically fractured wells in 2015 decreased to about 20,000. Despite recent declines in prices and new drilling, U.S. gas and oil production continues at levels above those in recent decades, and production for both is predicted to continue growing in the long term, led by hydraulic fracture-based production from unconventional reservoirs.

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3. Hydraulic Fracturing for Oil and Gas in the United States

3.1 Introduction

This chapter provides general background information on hydraulic fracturing and will help the reader understand the in-depth technical chapters that follow. We describe the purpose and process of hydraulic fracturing and the situations and settings in which it is used (Section 3.1). Then we provide a general description of activities at a hydraulic fracturing well site including assessing and preparing the well site, well drilling and construction, the hydraulic fracturing event, the oil and gas production phase, and eventual site closure (Section 3.3). A characterization of the prevalence of hydraulic fracturing in the United States is then presented (Section 3.4), followed by a review of its current and future importance in the oil and gas industry and its role in the U.S. energy sector (Section 3.5), and a brief conclusion (Section 3.6).

3.2 What is Hydraulic Fracturing?

Hydraulic fracturing is a technique used to increase oil and gas production from underground oil- or gas-bearing rock formations (reservoirs).[1] The technique involves the injection of hydraulic fracturing fluids through the production well and into the reservoir under pressures great enough to fracture the reservoir rock. The injected hydraulic fracturing fluid carries "proppant" (typically sand) into the fractures so that they remain propped open after the pressurized injection is stopped. In addition to water, which typically makes up most of the injected fracturing fluid, the fluid also contains chemical additives (additives) that serve a variety of purposes. These additives, for example, can increase the fluid viscosity (how "thick" the fluid is) so that it carries the proppant into the fractures more effectively, can help control well corrosion, can help minimize microbial growth in the well, and so on (King and Durham, 2015; Gupta and Valkó, 2007). The resulting fractures enable better flow of oil and gas from the reservoir into the production well. Water that naturally occurs in the reservoirs also typically flows into and through the production well to the surface as a byproduct of the production process.

Although hydraulic fracturing is not new, how and where it is employed has changed (Text Box 3-1). For about a half-century after its introduction in the late 1940s, it was used to increase production from vertical wells in conventional oil and gas reservoirs. Conventional reservoirs develop over geologic time (many millions of years) when naturally buoyant oil and gas very slowly migrate upward from the shale rock formations in which they formed until they are trapped by geologic formations or structures and accumulate under a confining layer (Figure 3-1). As the oil and gas accumulate, the pressure may increase. If the reservoir is under enough pressure and has

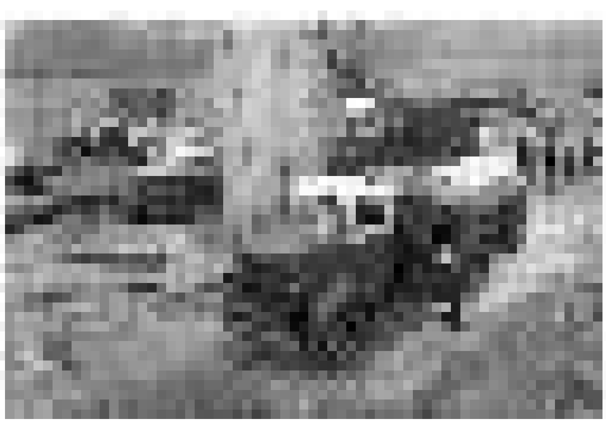
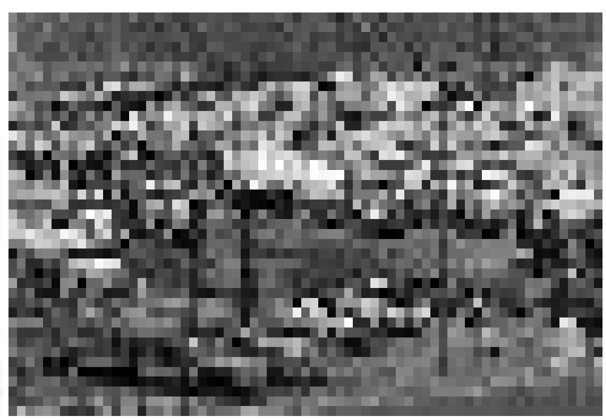
[1] A version of hydraulic fracturing, sometimes called hydrofracturing or hydrofracking, can be used to increase water yields from water wells and is typically done by injecting only water under pressure. This application of hydraulic fracturing is out of the scope of this assessment.

Text Box 3-1. Hydraulic Fracturing: Not New, but Different and Still Changing.

From the mid-1800s to the 1940s, operators of oil and gas wells occasionally tried to increase production by pumping fluids or sometimes dropping explosives into wells. In the late 1940s, a fracturing technique to increase production was patented by the Stanolind Oil and Gas Company and licensed to the Halliburton Oil Well Cementing Company (Montgomery and Smith, 2010). Close to 1 million wells were hydraulically fractured from the late 1940s to about 2000 (IOGCC, 2002). The typical well design and hydraulic fracturing operations during most of that time, though, were very different from today's modern hydraulic fracturing operations.

The groundwork for the transformation to modern hydraulic fracturing was laid in the 1970s and early 1980s. Public-private research and development (R&D) partnerships that included industry, the Department of Energy, and the Gas Research Institute were established because large amounts of natural gas were known to occur in some shale rock formations yet traditional production well technology was not able to recover much of the gas (Avila, 1976). These R&D programs played a key role in advancing technologies such as deep horizontal drilling and fracturing with higher water volumes that ultimately enabled production from shales and other unconventional sources of gas and oil (DOE, 2015; NRC: Committee on Benefits of DOE R&D on Energy Efficiency and Fossil Energy, 2001). During this period, the U.S. Congress began offering tax incentives for producers to use the developing technologies in the field (Wang and Krupnick, 2013; EIA, 2011a; Yergin, 2011). Advances in directional drilling technologies led to the first horizontal wells being drilled in the mid-1980s in the Austin Chalk oil-bearing rock formation in Texas (Pearson, 2011; Haymond, 1991). Directional drilling and other technologies matured in the late 1990s. In 2001, the Mitchell Energy company developed a cost-effective technique to fracture the Barnett Shale in Texas. The company was bought by Devon Energy, a company with advanced experience in directional and horizontal drilling, that, in 2002, drilled seven wells and developed in the Barnett Shale using the combination of horizontal drilling and hydraulic fracturing; fifty-five more wells were completed in 2003 (Yergin, 2011). The techniques were rapidly adopted and further developed by others (DOE, 2011b; Montgomery and Smith, 2010). By 2005, the techniques were being used in unconventional (low-permeability) oil and gas reservoirs outside of Texas. Modern hydraulic fracturing quickly became the industry standard, driving a surge in U.S. production of oil and natural gas.

Hydraulic fracturing techniques and technologies continue to evolve. Wells are being drilled with longer horizontal sections and are more closely spaced. Multiple, horizontal sections extending from a single vertical well enable production from larger subsurface areas from a single well pad on the land's surface. These historic and continuing technological developments enable production from previously unused oil and gas-bearing geologic formations, altering and expanding the geographic range of oil and gas production activities.

Left: Early hydraulic fracturing site, late 1940s (source: Halliburton, used with permission). Right: Contemporary hydraulic fracturing operation, late 2000s (source: NYSDEC (2015), used with permission).

adequate natural permeability, the economic extraction of oil and/or gas may only require using a drilled well to bring the oil or gas to the surface.[1]

If the natural pressure is not high enough for the oil and gas to readily flow to the surface, various pumping and "lift" techniques can be used to help the oil and gas move up the well to the surface (Hyne, 2012). In other situations, operators may pump water or a mix of water and carbon dioxide (or other similar mixtures) into the reservoir through injection wells to help move and enhance the extraction of oil and gas through nearby production wells. These techniques address pressure and fluid characteristics in the reservoir, are not designed to fracture the reservoir rock, and therefore are production-increasing techniques that are distinct from hydraulic fracturing. The discussions in the remainder of this chapter focus on hydraulic fracturing in unconventional reservoirs.

Hydraulic fracturing is now combined with directional drilling technologies to access oil and gas in unconventional reservoirs (although hydraulic fracturing is still used in conventional reservoirs, too).[2] Unconventional reservoirs have a very low natural permeability, which prevents oil and gas from flowing through the rock into wells in economic amounts. Production from unconventional reservoirs becomes economically feasible when wells, typically horizontal or deviated, are drilled and hydraulically fractured through long portions of the production zone (the targeted oil- and gas-bearing zones within a reservoir). See Figure 3-1 for a diagram of horizontal and other well types and the reservoir types from which they can produce. Text Box 3-2 provides a brief discussion on the use of the terms conventional and unconventional.

More details about the geologic formations that can be unconventional reservoirs are presented below:

- ***Shales.*** Some organic-rich black shales serve as the source of oil and gas found in conventional resources when, over geologic time, the lighter and more buoyant oil and gas migrate upward from these shales and become trapped under impermeable confining layers (Figure 3-1). Shales have very low permeability and the oil and gas are contained in poorly connected pore space in the shale rock. With hydraulic fracturing and directional drilling now enabling oil and gas production from very low permeability formations, some of these shale source rocks are now unconventional reservoirs in addition to being sources. Some shales produce predominantly gas and others predominantly oil; often there will be some co-production of gas from oil wells and co-production of liquid oil from gas wells (USGS, 2013a; EIA, 2011a).
- ***Tight formations.*** Some oil- and gas-bearing sandstone, siltstone, and carbonate formations can be referred to as "tight" formations (for example, "tight sands") because of their relatively low permeability and the fact that oil and gas are contained in small, poorly connected pore spaces. Given a range of permeabilities, some tight formations require

[1] Permeability in rocks is the ability of fluids, including oil and gas, to flow through well-connected pores or small openings in the rock.

[2] Directional drilling is the practice of controlling the direction and deviation (angle) of a borehole during drilling to extend the borehole in a predetermined orientation and to a targeted area in the subsurface. Directional drilling is required for drilling a deviated or horizontal well and is common in unconventional reservoirs. The terms deviated wells and directional wells are often used interchangeably.

hydraulic fracturing for economic production and some do not. In the literature, "tight gas" generally refers to gas in tight sands and is distinguished from "shale gas." Oil resources from shale and other tight formations, in contrast, are frequently referred together under the label "shale oil" or "tight oil" (Schlumberger, 2014; USGS, 2014a).

- ***Coalbeds.*** Organic-rich coal, found in coalbeds, can be a source of methane (natural gas). The gas primarily adheres to the coal surface rather than being contained in pore space or structurally trapped in the formation. A range of techniques can be used to extract methane from coalbeds and these techniques sometimes, but not always, employ hydraulic fracturing. A key component of all coalbed methane production is the need to "dewater" the coalbeds (pumping out naturally occurring or injected water) to reduce the pressure in the coal allowing the methane to be released and flow from the coal into the production well (Palmer, 2010; Al-Jubori et al., 2009; USGS, 2000).

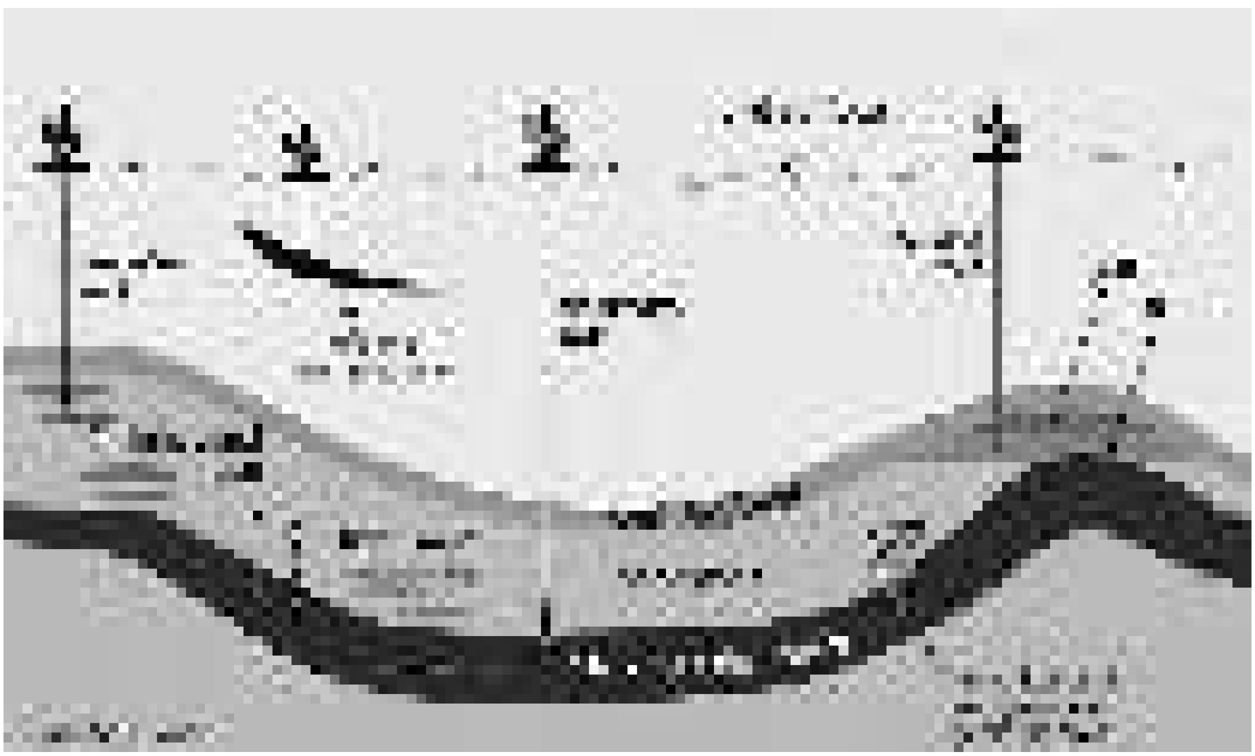


Figure 3-1. Conceptual illustration of the types of oil and gas reservoirs and production wells used in hydraulic fracturing.

A vertical well is producing from a conventional oil and gas reservoir (right). The impermeable gray confining layer (sometimes called a cap rock) traps the lighter and more buoyant gas (red) and oil (green) as it migrates up from the deeper oil- or gas-rich shale source rock. Also shown are wells producing from unconventional reservoirs: a horizontal well producing from a deep shale (center); a vertical well producing methane (gas) from coalbeds (second from left); and a deviated well producing from a tight sand reservoir (left). Multiple deviated or horizontal wells can be constructed and operated from a single well site. Note that the oil- or gas-rich shale serves as both a source and a reservoir. Modified from Schenk and Pollastro (2002) and Newell (2011).

Text Box 3-2. "Conventional" Versus "Unconventional."

The terms "conventional" and "unconventional" are widely used in articles and reports to distinguish types of oil and gas reservoirs, wells, production techniques, and more. In this report, the terms are mainly used to distinguish different types of oil and gas reservoirs: "conventional" reservoirs are those that can support the economically feasible production of oil and gas using long-established technologies, and "unconventional" reservoirs are those in which production has become economical only with the advances that have occurred in hydraulic fracturing (often combined with directional drilling) in recent years.

Note that as hydraulic fracturing has increasingly become a standard industry technique, the word "unconventional" is less apt than it once was to describe these oil and gas reservoirs. In a sense, "the unconventional has become the new conventional" (NETL, 2013).

The following three maps show the locations of major shale gas and oil resources, tight gas resources, and coalbed methane resources, respectively, in the contiguous United States (Figure 3-2, Figure 3-3, and Figure 3-4). To explain the terminology used in the maps: a group of known or possible oil and gas accumulations in the same region and with similar geologic characteristics can be referred to as a *play* (Schlumberger, 2014). Plays can sometimes be geologically layered atop one another (or "stacked") and are located in broad depressions filled with sedimentary rock formations in the earth's continental crust known as *basins*. A group of similar coalbed methane (gas) reservoirs can be referred to as coalbed methane *fields* (rather than plays) and are also found in basins. The plays and fields in the maps below represent unconventional reservoirs that are being exploited now or could be exploited in the future using hydraulic fracturing.

There is a wide range of depths at which hydraulic fracturing occurs across the country. For example, approximate average depths for some of the largest gas-producing reservoirs are as deep as 6,000 ft (2,000 m) in the Marcellus Shale in Pennsylvania and West Virginia, 7,500 ft (2,300 m) in the Barnett Shale in Texas, and 12,000 ft (3,700 m) for the Haynesville-Bossier Shale in Louisiana and Texas (NETL, 2013).[1] A few other, smaller plays are shallower, with depths less than 2,000 ft (600 m) in parts of the Antrim (Michigan), Fayetteville (Arkansas), and New Albany (Indiana and Kentucky) shale plays (NETL, 2013; GWPC and ALL Consulting, 2009). Coal seams that can be drilled to produce gas (coalbed methane) range in depth from less than 600 ft (200 m) to more than 6,000 ft (2,000 m) with production often occurring at depths between 1,000 and 3,000 ft (300 and 900 m) (U.S. EPA, 2006; ALL Consulting, 2004). Coalbed methane production occurs in the San Juan Basin in New Mexico, the Powder River Basin in Wyoming and Montana, and the Black Warrior Basin in Alabama and Mississippi. See Chapter 6 for more information on the general locations and depths of formations being hydraulically fractured.

[1] These are approximate average depths; hydraulic fracturing occurs in shallower and deeper zones in all these plays.

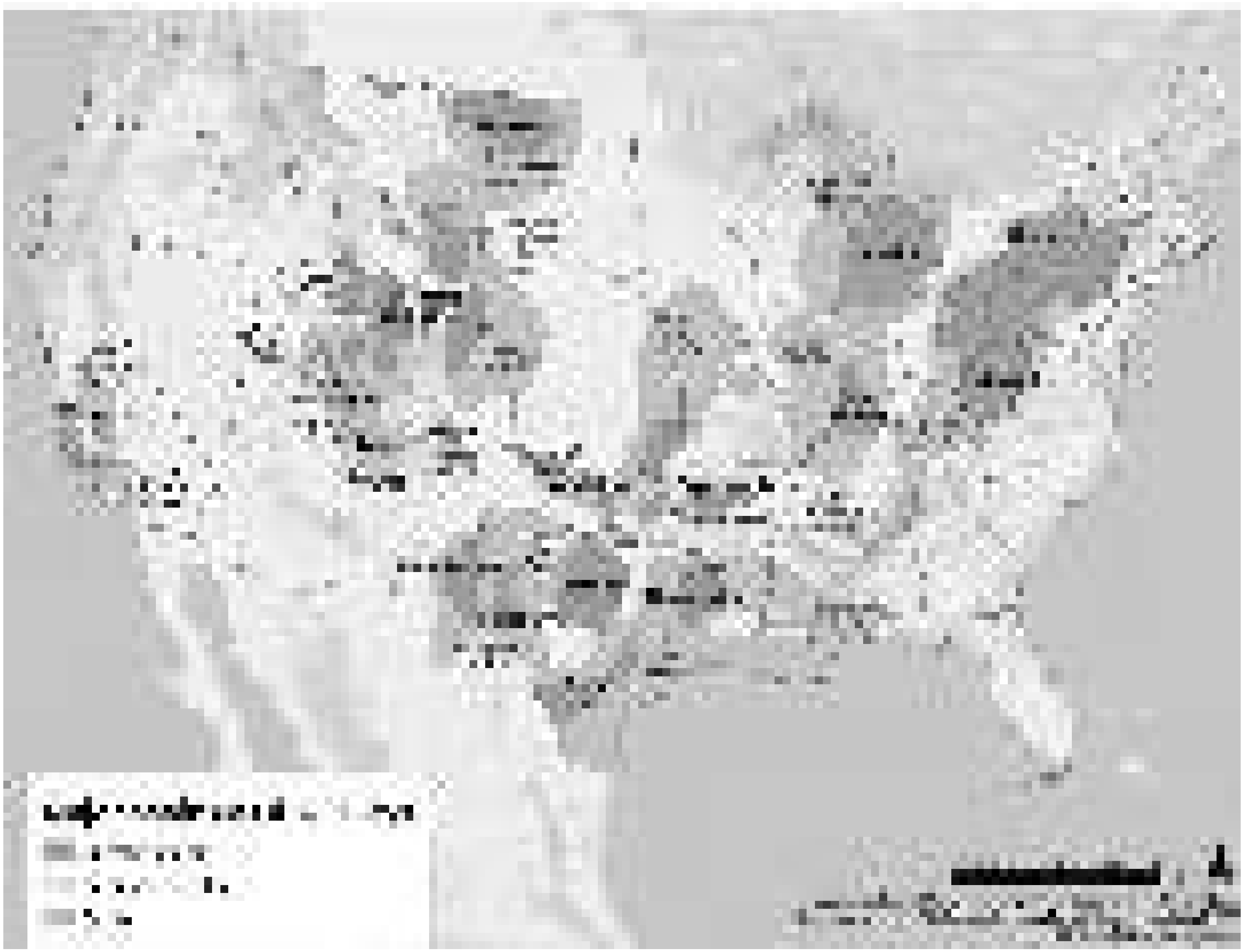


Figure 3-2. Major shale gas and oil plays in the contiguous United States.
The plays represent geologically similar accumulations of oil and gas that are or could be developed. Adapted from EIA (2015).

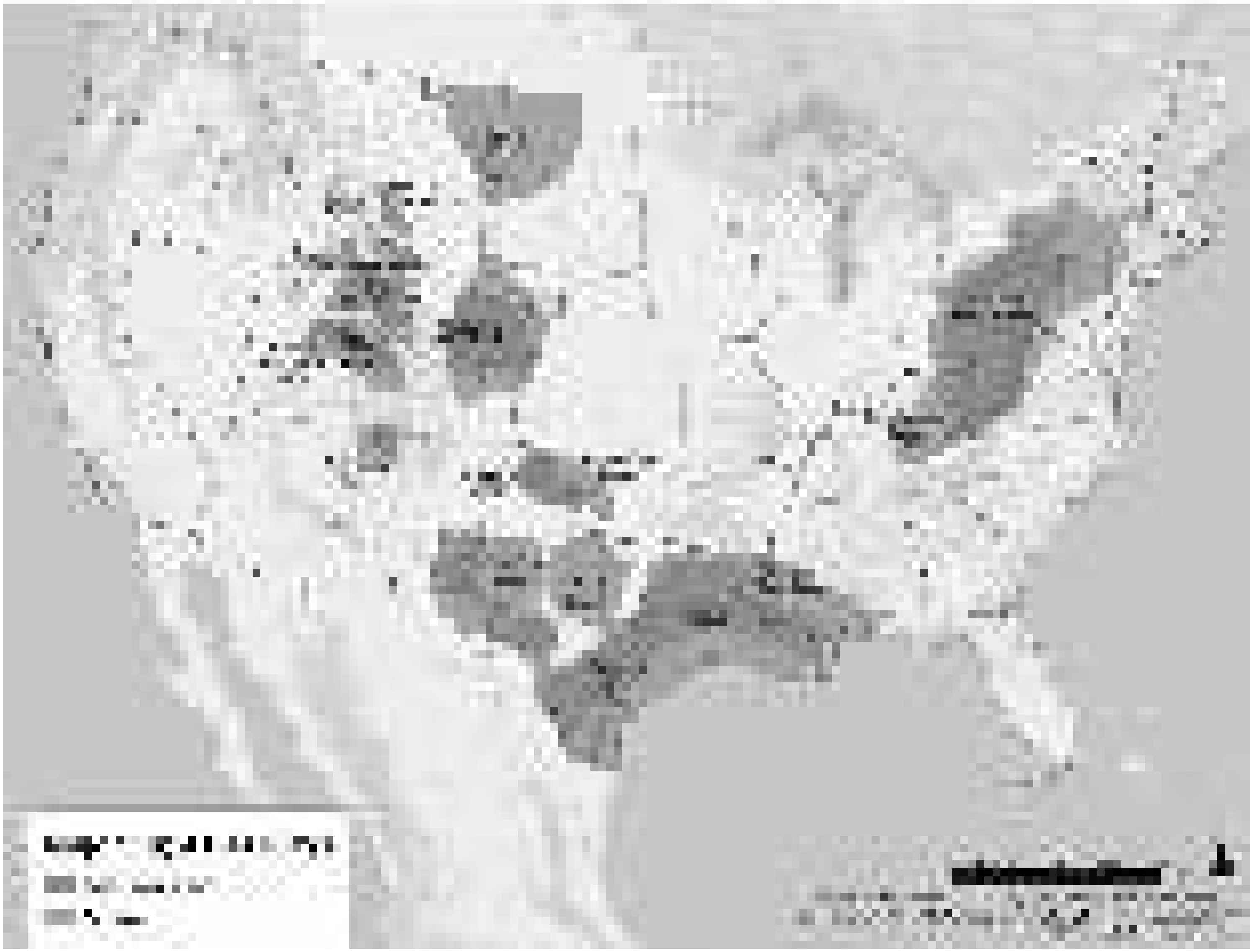


Figure 3-3. Major tight gas plays in the contiguous United States.
The plays represent geologically similar accumulations of gas that are or could be developed. Adapted from EIA (2011b).

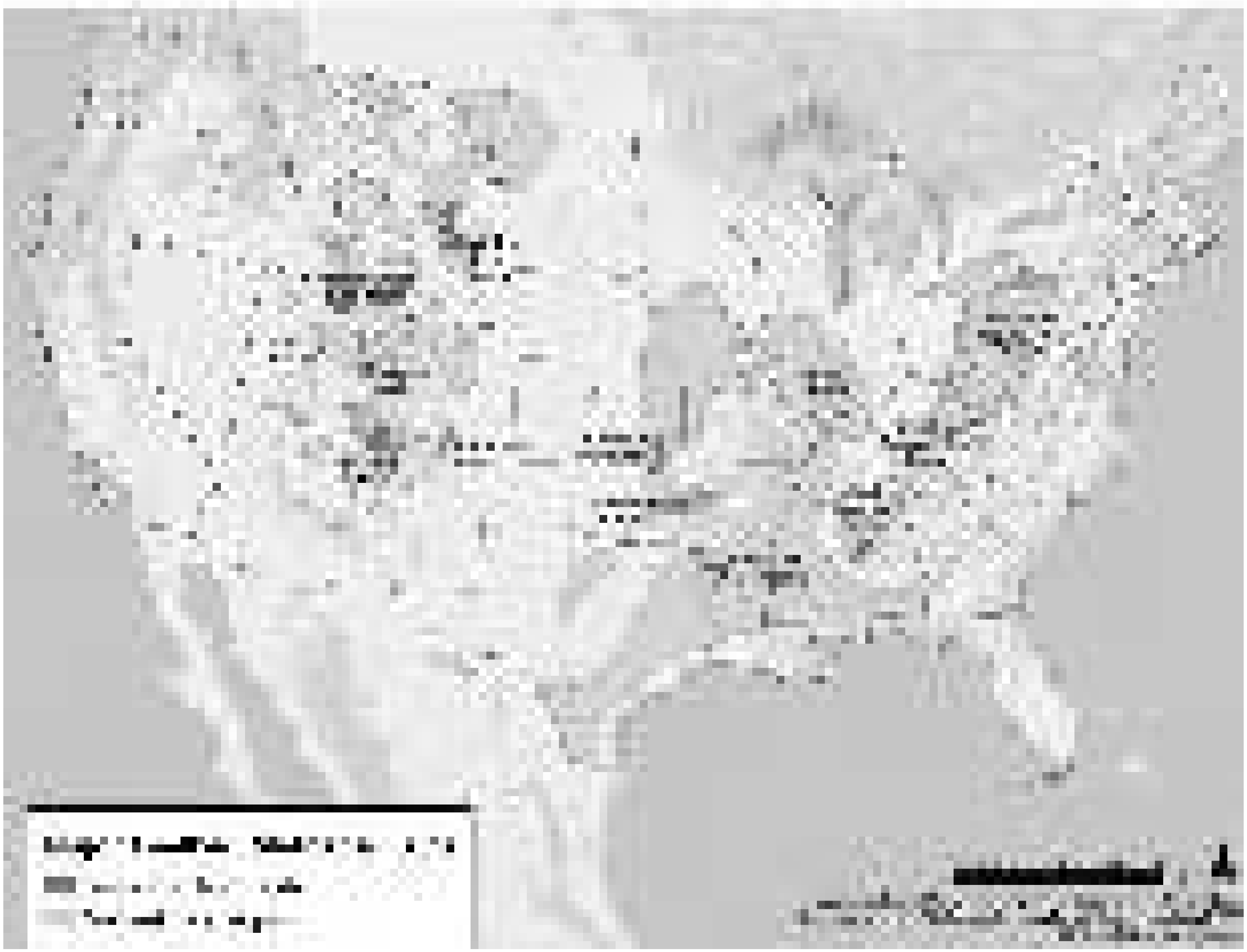


Figure 3-4. Coalbed methane fields and coal basins in the contiguous United States.
The fields represent gas-bearing coal deposits that are or could be developed. Adapted from EIA (2011b).

How a hydraulic fracturing operation is conducted depends on the characteristics of the oil- or gas-bearing formation (such as the geology, depth, and other factors). Hydraulic fracturing operations in shales, such as the Marcellus and Haynesville, require that relatively large volumes of water and proppant to be pumped at high pressures through deep wells with long horizontal sections in the production zone. In some tight formations, such as in the Permian Basin, hydraulic fracturing can be conducted with smaller water volumes and using less pressure in shorter vertical or deviated wells (Gallegos and Varela, 2015). Hydraulic fracturing technologies can be applied to coalbed methane production in various ways, for example, with much smaller water volumes and no proppant, or with water-based gels or foams and proppant. Coalbed methane production sometimes involves no hydraulic fracturing, with only pumping of the naturally occurring formation water out of the coalbeds to enable the release and production of the trapped methane.[1]

3.3 Hydraulic Fracturing and the Life of a Well

A variety of activities take place at a well site over the course of the operational life of a hydraulically fractured oil and gas production well. Not all of these activities are within the scope of this assessment (that includes water acquisition, chemical mixing, well injection, produced water handling, and wastewater disposal and reuse). However, in this chapter we include some information on a wider range of activities related to the well site to provide context for the reader.

The overview of well operations presented in this section is broad, illustrates common activities, and describes some specific operational details. The details of well preparation, hydraulic fracturing and production operations, and closure can vary between companies, reservoirs, and states, and even from well to well. The activities involved in well development and operations may be conducted by the well owner and/or operator, their representatives, and/or service companies working for the well owner.

Figure 3-5 shows the general sequence and duration of activities at a hydraulic fracturing well site, including the activities that comprise the five stages of the hydraulic fracturing water cycle (noted above and defined in Chapter 1). The hydraulic fracturing event itself is the period of the most operational activity during the life of a well and is short in duration compared to the other well site activities. The hydraulic fracturing activity typically lasts from about a day to several weeks (U.S. EPA, 2016c; Halliburton, 2013; NYSDEC, 2011). The subsequent phase of oil and gas production, during which produced water also flows from the well, is the longest phase during the life of the well and can last decades (King and Durham, 2015).[2]

[1] Some subsurface geologic formations, including coalbeds and oil and gas reservoirs, can contain naturally occurring water that is commonly referred to as "formation water," "native water," or (if salty) "native brines."

[2] In general, produced water is water that flows from the subsurface through oil and gas wells to the surface as a by-product of oil and gas production. See Section 3.3.3 and Chapter 7 for more details.

Figure 3-5. General timeline and summary of activities that take place during the preparation and through the operations of an oil or gas well site at which hydraulic fracturing is used.

3.3.1 Site Preparation and Well Construction

Before hydraulic fracturing and production can occur, preliminary steps include assessing and preparing the site, and drilling and constructing the production well.

3.3.1.1 Site Assessment and Preparation

Selecting a suitable well site requires an assessment of geologic (subsurface) and geographic (surface) factors. Geophysical surveys of the subsurface can be conducted using data gathering techniques from the land surface or subsurface, and rock samples may be gathered from outcrops or from exploratory or test wells. Other information is obtained by well logging in which geophysical instruments that collect data on subsurface conditions are lowered into or installed in a well (Kundert and Mullen, 2009).[1] Analyzing all of this information together enables operators to develop an understanding of the potential reservoir characteristics (such as permeability and the presence of natural fractures and water), the position of such formations in relation to other

[1] Well logging is used to obtain information on mechanical integrity, well performance, and reservoir properties that can affect oil and gas production. Well logging data from other wells in the nearby area also provides information on the reservoir. More information on well logging is found in Chapter 6 and Appendix D.

formations, including water-bearing zones, and details about the quantity and quality of the oil and gas resource.

Geographic factors involved in well site assessment include topography and land cover; proximity to roads, pipelines, water sources, other oil and gas wells, and abandoned oil or gas wells; possible well setback requirements; potential for site erosion; location relative to environmentally sensitive areas; and location relative to populated areas (Drohan and Brittingham, 2012; Arthur et al., 2009a).[1] Land ownership also plays an important role in well site selection. During site assessment and before site development and well drilling, the well owner/operator obtains a mineral rights lease, negotiates with landowners, and applies for necessary permits from the appropriate federal, state, and local authorities (Hyne, 2012). This initial site assessment phase of the process may take several months (King and Durham, 2015; King, 2012).

The site is typically surveyed to plan and finalize well site location and access. Sometimes an access road may need to be built to accommodate trucks delivering equipment and supplies to be used at the site (Hyne, 2012). The operator levels and grades the well site to manage drainage, complete access routes, and prepare the well pad. The well pad is a smaller area within the broader well site where the production well will be drilled and the hydraulic fracturing activities will be concentrated. Well pads can range in size from less than an acre to several acres depending on the scope of the operations (King, 2012; NYSDEC, 2011). Multiple wells can be located on a single well pad at a well site (King, 2012; NYSDEC, 2011)

To manage the various fluids that are used for or generated during operations, storage pits (sometimes referred to as impoundments) are excavated, graded and constructed on the well site, and/or steel tanks are installed. These are used to hold water and materials (such as drilling mud) related to the well-drilling activities, water used in the hydraulic fracturing process, or the produced water that is generated post-fracturing (Hyne, 2012). Pit construction is generally governed by local regulations. In some areas, regulations may prohibit the use of pits or require pits to be lined to prevent fluid seepage into the shallow subsurface. One alternative to constructing a pit for drilling fluids is the use of a closed loop drilling system that stores, partly treats, and recycles the drilling fluid (Astrella and Wiemers, 1996). Often piping is installed along the surface or in the shallow subsurface of the well site to deliver water for hydraulic fracturing, remove produced water, or transport the oil and gas once production begins (Arthur et al., 2009a).

Water may be acquired from local surface water or groundwater resources, or reused from other well sites. Water is required for the drilling phase as well as for hydraulic fracturing (Chapter 4). Figure 3-6 depicts the pumping of water for well site operation from a local surface water source.

After site and well pad preparation, drill rigs and associated equipment (including the drill rig platform, generators, well blowout preventer, fuel storage tanks, cement pumps, drill pipe, and casing) are brought onto the site.

[1] Regarding well setbacks, some states and sometimes local city or county governments can have requirements that define how close an oil and gas well can be located to drinking water supplies or other water bodies.

Figure 3-6. Surface water being pumped for oil and gas development.
Photo credit: Arkansas Water Science Center (USGS).

3.3.1.2 Well Drilling and Construction

Wells are generally drilled and constructed by repeating several basic steps. The operator begins by using the drill rig (temporarily located on the well pad) to hoist a section of long drill pipe up and attaching a drill bit to the bottom of the drill pipe. The drill rig is then used to rotate and advance the drill pipe/drill bit combination (also known as the drill string) downward through the soil and rock. As the drill string continues moves downward, new sections of pipe are added at the surface, enabling the drilling to proceed deeper (Hyne, 2012). During drilling, a drilling fluid is pumped down through the center of the drill string to the drill bit to lubricate and cool it, and to help remove the drill cuttings from the well (King, 2012).[1]

Drilling is temporarily halted at certain pre-determined intervals, the drill string is removed from the wellbore (also called the borehole), and long sections of another type of steel pipe called casing are lowered into the wellbore and set in place.[2] Cement is then pumped into the space between the outside of the casing and the wellbore. This process is repeated, with the next interval of drilling

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[1] Drilling fluids, sometimes called drilling mud, consist primarily of water, foam, oil or air, with the most common drilling mud consisting mainly of water and clay (Williamson, 2013). Drill cuttings are the small pieces of broken and ground-up rock generated during the drilling process.
[2] The wellbore is the drilled hole and can refer to both the open hole or an uncased portion of the well.

using a smaller diameter drill bit that fits inside the existing casing. The result can be multiple layers of casing and cement with surface casing and cement typically set below the groundwater resource to be protected. Figure 3-7 illustrates different types of casing as defined by their locations within the well, shows multiple casing and cement layers, and shows examples of two wells with differences in the extent of cement.[1]

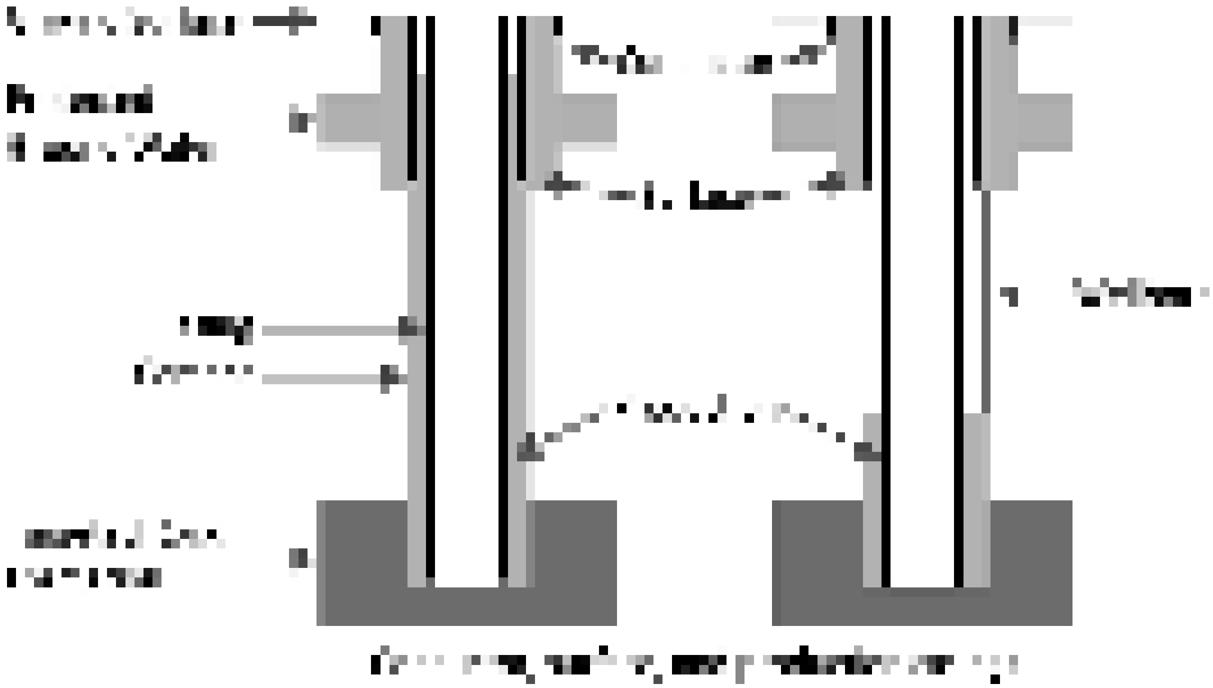

Figure 3-7. Illustration of well construction showing different types of casing and cement.

The well on the left is cemented continuously from the surface to the production zone and the well on the right has cement in sections, including sections cemented across protected groundwater.

The cement protects the casing from corrosion by formation water, helps physically support the casing in the borehole, and stabilizes the borehole against collapse or deformation (Renpu, 2011).[2] The casing and cement help to isolate geologic zones of high pressure, isolate water-bearing zones, and maintain the integrity of the production well for transporting oil and gas to the surface. Casing and cement provide important barriers that keep fluids within the well (oil, gas, hydraulic fracturing fluids) isolated and separated from fluids outside the well (formation water) (Hyne, 2012). Figure 3-8 shows sections of casing ready for installation.

[1] In different portions of the well, multiple concentric sections of casing of different diameters can be used as shown by the surface and production casings in Figure 3-7. The largest casing diameter can range between 30 in. (76 cm) to 42 in. (107 cm) with casing diameters typically larger in the shallower portions of a well and smaller in the deeper portions (Hyne, 2012). See Appendix D for details on well construction and casing diameters.

[2] Some naturally occurring formation water can be very saline (salty or briny), which can be corrosive to metal.

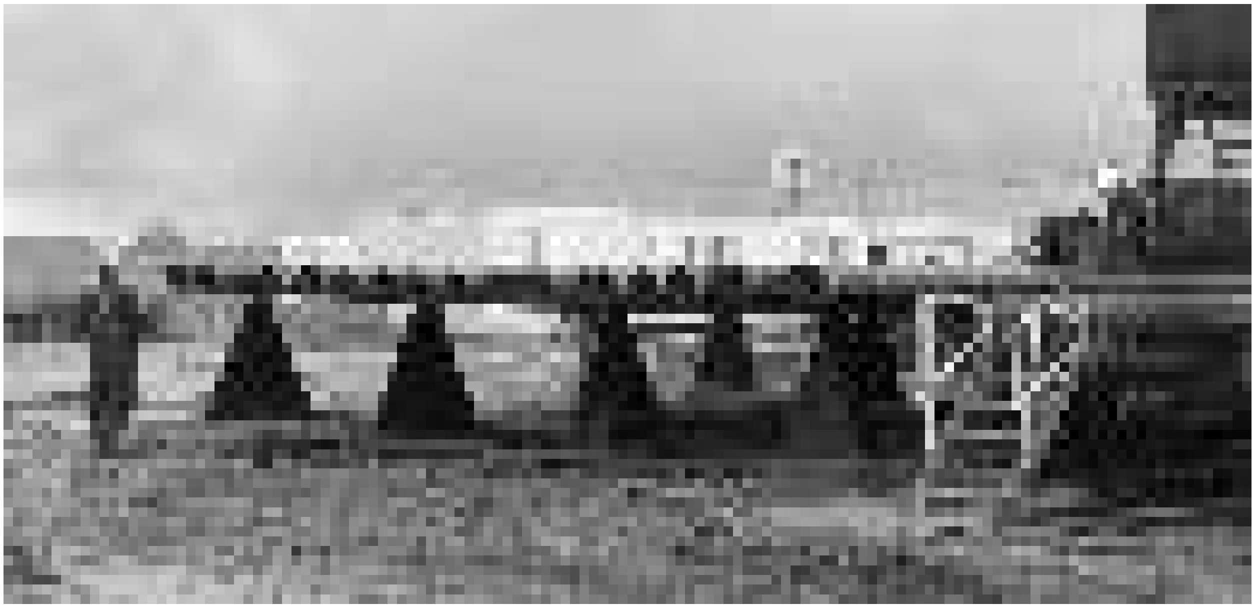

Figure 3-8. Sections of well casing ready for installation at a well site in Colorado.
Photo credit: Gregory Oberley (U.S. EPA).

Some wells are cemented continuously from the surface down to the production zone. Other wells are partially cemented with, for example, cement from the surface to some distance below the deepest protected groundwater zone and perhaps cement across high pressure or water- or oil-bearing zones. Sometimes there can be multiple casing and cement layers (Figure 3-7). There are advantages, in some situations, to not fully cementing the casing as long as high pressure or water- and oil-bearing zones are cemented. For example, some sections may not be cemented to allow monitoring of the pressure in the space between the casing and the borehole or to prevent damage to weak rock formations due to the weight of the cement[1] (King and Durham, 2015; API, 2009).

Although wells are initially drilled vertically (more or less straight down), the sections of the wells that are hydraulically fractured in the production zone of the reservoir can be vertical, deviated, or horizontal (Figure 3-1). The operator determines the well orientation that will provide the best access to the targeted zone(s) within a reservoir and that will align the production section of the well with natural fractures and other geologic structures in a way that helps improve production. Deviated wells may be "S" shaped or continuously slanted. So-called "horizontal wells" have one or more extensions or branches oriented approximately 90 degrees from the vertical portion of the well; these horizontal sections are often referred to as "laterals." The lengths of laterals can range from 2,000 to 10,000 ft (600 to 3,000 m) or more (Hyne, 2012; Miskimins, 2008; Bosworth et al., 1998). Multiple laterals can extend in different directions from a single well (and multiple wells can be located on a single well site). This allows access to more of the production zone with a higher well density in the subsurface, which can be required for unconventional reservoirs, while having fewer well sites on the land surface.

[1] The use of lighter cement or special cementing techniques can also prevent damage of weaker rock formations. See Chapter 6 and Appendix D for more details on well construction and cementing.

Once well construction is completed, the operator can move the drilling rig and related drilling equipment, install the wellhead (the top portion of the well), and prepare the well for hydraulic fracturing and subsequent production of oil and gas. Chapter 6 and Appendix D contain more details on well construction, casing, and cement.

Figure 3-9 (from northeastern Pennsylvania) and Figure 3-10 (from northwestern North Dakota) show, in the context of the local landscape, well sites during well drilling and construction prior to hydraulic fracturing activities.

Figure 3-9. Aerial photograph of two hydraulic fracturing well sites and a service road in Springville Township, Pennsylvania.
Photo credit: Image@J Henry Fair / Flights provided by LightHawk.

Figure 3-10. Aerial photograph of hydraulic fracturing well sites near Williston, North Dakota.
Photo credit: Image@J Henry Fair / Flights provided by LightHawk.

3.3.2 Hydraulic Fracturing

The hydraulic fracturing phase is an intense phase of work in the life of the well that involves complex operational activities at the well site. This phase of work is short in duration, compared to other work phases in the life of a well, and typically lasts less than two weeks per well. It consists of multiple activities, is typically a process done in repetitive stages, and requires a variety of equipment and materials. During this phase of work, the well is prepared for hydraulic fracturing, specialized equipment is hauled to the well site, the hydraulic fracturing fluid components –the water, proppant, and additives– are moved to the well site, and the hydraulic fracturing fluid is mixed and injected under pressure through the well and into the targeted production zone in the subsurface (Figure 3-11).

Figure 3-11. Well site with equipment (and pits in the background) in preparation for hydraulic fracturing in Troy, Pennsylvania.
Image from NYSDEC (2015). Reprinted with permission.

3.3.2.1 Injection Process

The section of well located in the production zone can be prepared for the injection and fracturing process in several different ways. One approach is used when the production casing and cement extend all the way into the production zone; this requires the use of focused explosive charges to perforate (blast holes in) the casing and cement in a segment of the well within the production zone. In another approach, known as a formation packer completion, only the casing, equipped with holes that can be opened and closed, is extended into the production zone. The resulting perforations or holes allow the injected hydraulic fracturing fluids to flow out of the well to fracture the reservoir rock and allow the oil and gas to flow into the well. Another technique is an open hole completion in which the casing is set and cemented just to the edge of the production zone, so the borehole extends open (with no casing or cement) into the production zone. In open hole completions, oil and gas flow directly into the borehole and eventually into the cased section of the well leading to the surface (Hyne, 2012; Cramer, 2008; Economides and Martin, 2007).

After the subsurface portion of the well is prepared for injection, a wellhead assembly is temporarily installed on the wellhead to which high pressure fluid lines are connected for injection of the fluids into the well. Figure 3-12 shows three wellheads with injection piping attached in preparation for hydraulic fracturing injection. Pressures required for fracturing can vary widely depending on depth, formation pressure, and rock type and can range from 2,000 psi to 12,000 psi (U.S. EPA, 2016c; Salehi and Ciezobka, 2013; Abou-Sayed et al., 2011; Thompson, 2010).

Figure 3-12. Three wellheads on a multi-well pad connected to the piping used for hydraulic fracturing injection.
Photo credit: DOE/NETL

The portion of the well to be fractured can sometimes be done all at once or done in multiple interval (U.S. EPA, 2016c; GWPC and ALL Consulting, 2009). When done in multiple intervals, shorter lengths or segments of the well are closed-off (using equipment inserted down into the well) and fractured independently in “stages” (Lee et al., 2011). Fluids are first injected to clean the well (removing any cement or debris). Then, for each stage fractured, a series of hydraulic fracturing fluid mixtures is injected to initiate fractures and carry the proppant into the fractures (Hyne, 2012; GWPC and ALL Consulting, 2009). The fracturing process can require moving millions of gallons of fluids around the well site through various hoses and lines, blending and mixing the fluids with proppant, and injecting the mixture at high pressures down the well. For more details on hydraulic fracturing chemical mixtures and stages, see Chapter 5.

The hydraulic fracturing produces propped-open fractures that extend into the production zone and create more flow paths that contact a greater volume of the oil- and gas-bearing rock within the production zone of the reservoir. This increase in flow paths and in the volume of the production zone accessed by the production well is how hydraulic fracturing increases production. In this regarding, hydraulic fracturing can be considered a production or well "stimulation" technique.

The process and the fracturing pressures during injection are closely monitored throughout the fracturing event. Microseismic monitoring (a geophysical survey technique) can be used to estimate the horizontal and vertical extent of the fractures created and, used with other monitoring and operational data, provides important information for designing subsequent fracture jobs (Cipolla et al., 2011). Engineers can design fracture systems using modeling software to help optimize the process. More details of injection, fracturing, and related monitoring are provided in Chapter 6 and Appendix D.

3.3.2.2 Fracturing Fluids

To conduct the chemical mixing and preparation of the hydraulic fracturing fluids, water- and chemical-filled tanks and other storage containers are transported and installed on site. The components that make up the hydraulic fracturing fluid for injection are commonly mixed on a truck-mounted blender on the well pad. Hoses and pipes are used to transfer the water, proppant, and chemicals from storage units to the mixing equipment and to the well into which the mixed hydraulic fracturing fluid will be injected. The injection process happens in stages with specific chemicals added at different times during each stage. The composition of the hydraulic fracturing fluid, therefore, can change over time during the process (Knappe and Fireline, 2012; Fink, 2003). See Chapter 5 for more details on mixing and staged injection.

Hydraulic fracturing fluids (sometimes referred to as "fluid systems") are generally either water-based or gel-based. Other fluid systems include foams or emulsions made with nitrogen, carbon dioxide, or hydrocarbons; acid-based fluids; and others (Montgomery, 2013; Saba et al., 2012; Gupta and Hlidek, 2009; Gupta and Valkó, 2007; Halliburton, 1988). Water-based systems are used more often with the most common type being "slickwater" formulations, which include polymers as friction reducers and are typically used in very low permeability reservoirs such as shales (Barati and Liang, 2014). Because slickwater fluids are thinner (have lower viscosity) they do not as easily carry sand proppant into fractures, so larger volumes of water and greater pumping pressures are required to effectively transport proppants into fractures. In contrast, gelled fluids (used in "gel fracs") are more viscous, and more proppant can be transported with less water as compared to slickwater fractures (Brannon et al., 2009). Gel fracs are generally used in reservoirs with higher permeability (Barati and Liang, 2014).

The composition of a typical water-based hydraulic fracturing fluid by volume is 90% to 97% water, 2% to 10% proppant, and 2% or less additives (U.S. EPA, 2015a; OSHA, 2014a, b; Carter et al., 2013; Knappe and Fireline, 2012; Spellman, 2012; Sjolander et al., 2011; SWN, 2011). In a detailed study, the EPA analysis of FracFocus 1.0 data for nearly 39,000 wells nationally in 2011 and 2012 indicates that the fracturing fluid injected into a well consists of nearly 90% water, 10% proppant, and less than 1% additives (on a mass basis) (U.S. EPA, 2015a). The proportions of water, proppant, and additives in the fracturing fluid, and the specific additives used, can vary depending on a number of factors, including the rock type and the chemistry of the reservoir, whether oil or gas is being produced, operator preference, and to some degree on local or regional availability of chemicals (Arthur et al., 2014; Spellman, 2012; GWPC and ALL Consulting, 2009; Gupta and Valkó, 2007). Hydraulic fracturing fluid composition and chemical use changes as processes are tested and refined by companies and operators. These changes are driven by economics, scientific and technological developments, and concerns about environmental and health impacts. Further detail on hydraulic fracturing fluid systems is presented in Chapter 5.

Sources of water for hydraulic fracturing fluid include groundwater, surface water, and reused wastewater (URS Corporation, 2011; Blauch, 2010; Kargbo et al., 2010). The water may be brought to the production well from an offsite regional source via trucks or piping, or it may be more locally sourced (for example, pumped from a nearby river or a groundwater well). Selection of water source depends upon availability, cost, water quality needs, and the logistics of delivering it to the site. Figure 3-13 shows a row of water tankers storing water on a well site. Chapter 4 provides additional details on water acquisition and the amounts of water used for hydraulic fracturing.

Figure 3-13. Water tanks (blue, foreground) lined up for hydraulic fracturing at a well site in central Arkansas.
Photo credit: Martha Roberts (U.S. EPA).

Proppants are most commonly silicate minerals, primarily quartz sand (GWPC and ALL Consulting, 2009). Sand proppants can be coated with resins that make them more durable. Ceramic materials are also sometimes used as proppants due to their high strength and resistance to crushing and deformation (Beckwith, 2011).

Additives generally constitute less than 2.0% of hydraulic fracturing fluids (Carter et al., 2013; Knappe and Fireline, 2012; GWPC and ALL Consulting, 2009). The EPA analyzed additive data in the EPA FracFocus 1.0 project database and estimated that chemicals used as additives were about 0.43% (the median value by mass) of the total amount of fluid injected for hydraulic fracturing (U.S. EPA, 2015a). Given the total volume of hydraulic fracturing fluid, these small percentages of chemicals in the fluid mean that a typical hydraulic fracturing job can handle, mix, and inject tens of thousands of gallons of chemicals. Chapter 5 includes details on the number, types, and estimated quantities of chemicals typically used in hydraulic fracturing.

3.3.3 Fluid Recovery, Handling, and Disposal or Reuse

At the end of the hydraulic fracturing process, the pressurized injection is stopped and the direction of fluid flow reverses. Initially, the fluid flowing back into the well and to the surface is mostly the injected fracturing fluid (sometimes referred to as flowback). The composition of the fluid changes over time, though, and after the first few weeks or months the proportion of hydraulic fracturing fluid flowing back into the well decreases and the proportion of formation water flowing into the well and to the surface increases (NYSDEC, 2011). In this assessment, the water that flows from the subsurface through oil and gas wells to the surface as a by-product of oil and gas production is referred to as produced water. The amount of produced oil or gas flowing into the well gradually increases until it is the primary constituent of the fluid emerging from the well at the surface. Produced water continues to flow from the production well along with the oil or gas throughout the operating life of the production well (Barbot et al., 2013). See Chapter 7 for details, descriptions, and discussions of the chemical composition and quantities of produced water recovered.

Produced water is sometimes referred to as hydraulic fracturing wastewater. Along with other liquid waste collected from the well pad (such as rainwater runoff), it is typically stored temporarily on-site in pits (Figure 3-14) or tanks. This wastewater can be moved offsite via truck or pipelines for treatment and reuse or for disposal. Most hydraulic fracturing wastewater in the United States is disposed of by injection into deep, porous geologic rock formations, often located away from the production well site. This disposal-by-injection occurs not through oil and gas production wells, but through wastewater injection wells regulated by EPA Underground Injection Control (UIC) programs under the Safe Drinking Water Act.[1] See Chapter 8 for a brief discussion of wastewater injection.

Figure 3-14. A pit on the site of a hydraulic fracturing operation in central Arkansas.
Photo credit: Caroline E. Ridley (U.S. EPA).

[1] States may be given federal EPA approval to run a UIC program under the Safe Drinking Water Act. Most oil- and gas-related UIC programs are implemented by the states although some are implemented by the EPA.

Other wastewater handling options include discharge to surface water bodies either with or without treatment, evaporation or percolation pits, or reuse for subsequent fracturing operations either with or without treatment (U.S. EPA, 2012h; U.S. GAO, 2012). Decisions regarding wastewater handling are driven by factors such as cost (including costs of temporary storage and transportation), availability of facilities for treatment, reuse, or disposal, and regulations (Rassenfoss, 2011). Chapter 8 contains details of the treatment, reuse, and disposal of wastewater.

3.3.4 Oil and Gas Production

After the hydraulic fracturing activity is completed, the fracturing-related equipment is removed and operators drain, fill in with soil, and regrade pits that are no longer needed unless multiple wells are drilled and fractured on the same pad. The well pad size is reduced as the operation moves toward the production phase (NYSDEC, 2011). Prior to and during production, the operator runs production tests to determine the maximum flow rate that the well can sustain and to determine optimum equipment settings (Hyne, 2012; Schlumberger, 2006). During production, monitoring of mechanical integrity and performance (with pressure tests, corrosion monitoring, etc.) can be conducted to ensure that the well is performing as intended. Such well tests and monitoring may be required by state regulations.

Produced gas typically flows from the well through a pipe to a “separator” that separates the gas from water and any liquid oil and gas (NYSDEC, 2011). The finished gas is typically piped to a compressor station where it is pressurized and then piped to a main pipeline for sale (Hyne, 2012). Production at oil wells proceeds similarly, although oil/water or oil/water/gas separation typically occurs on the well pad, no compressor is needed, and the oil can be hauled by truck or train, or piped from the well pad to offsite storage and sale facilities.[1]

During the life of the well it may be necessary to repair components of the well and replace old equipment. Sometimes the well is re-fractured to boost production.[2] Routine maintenance activities, often referred to as “workovers,” may be done with the well still in production (Vesterkjaer, 2002) or sometimes require stopping production and removing the wellhead to clean out debris or repair components of the well (Hyne, 2012). More extensive re-workings of a well, sometimes referred to as “re-completions,” can include making additional perforations in the well in new sections to produce oil and/or gas from another production zone, lengthening the borehole, or drilling new horizontal extensions (laterals) from an existing borehole.

3.3.4.1 Production Rates and Duration

The production life of a well depends on a number of factors, such as the amount of oil or gas in the reservoir, the reservoir pressure, the rate of production, and the economics of well operations, including the price of oil and gas. In hydraulically fractured wells in unconventional reservoirs,

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[1] In some oil production operations, the oil reservoir being tapped may include some natural gas that is extracted along with oil through the production wells. In cases where no facilities or pipelines are in place to handle the natural gas or move it to a market, the gas can be “flared” (ignited and burned at the well site) or vented into the atmosphere.

[2] Sometimes boosting or reinvigorating production in a well is referred to as “well stimulation.” In some cases, well stimulation can refer to either the initial well hydraulic fracturing event or the re-fracturing of a well.

initial high production is typically followed by a rapid drop and then a slower decline in production (Patzek et al., 2013). The production phase may be 40 to 60 years in tight gas reservoirs (Ross and King, 2007) or range from 5 to 70 years in a gas- or liquids-rich shale (King and Durham, 2015). However, because the current hydraulic fracturing-led production surge is less than a decade old with limited well production history, there is an incomplete picture of production declines and it is not known how much and for how long these wells will ultimately produce (Patzek et al., 2013).

3.3.5 Site and Well Closure

Once a well reaches the end of its useful life, it is removed from production and disconnected from any pipelines that transferred produced oil or gas offsite. The well is then sealed to prevent any movement of fluids inside or along the borehole. This is done by removing the wellhead, cutting the casing off below ground surface, and then sealing portions of the well with one or more cement or mechanical plugs placed permanently in sections of the well. Spaces between plugs may be filled with a thick clay (bentonite) or drilling mud (NPC, 2011b). State regulations identify plugging locations within the borehole and the materials for plugging (Calvert and Smith, 1994). After plugging and cementing, a steel plate is welded on top of the well casing to provide a complete seal (API, 2010). Permanently closing a well like this is called “plugging” a well. Some states require formal notification of the location of these plugged wells. Proper plugging prevents fluids at the surface from seeping down the borehole and migration of fluids through the borehole (NPC, 2011b). See Chapter 6 for more details regarding fluid movement in wells and through the borehole.

To complete site closure, any remaining production-related equipment is removed and the site land cover and topography are restored to pre-well pad conditions to the extent possible. Some surface structures from the former operations may be left in place for subsequent reuse.

3.4 How Widespread is Hydraulic Fracturing?

There is no national database or complete national registry of wells that have been hydraulically fractured. However, hydraulic fracturing activity for oil and gas production in the United States is substantial based on various reports and data sources. According to the Interstate Oil and Gas Compact Commission (IOGCC), close to 1 million wells had been hydraulically fractured in the United States by the early 2000s (IOGCC, 2002). A recent U.S. Geological Survey report estimated approximately 1 million wells with 1.8 million hydraulic fracturing treatment records from 1947 to 2010 (more than one fracturing event, or treatment, can be conducted on a single well) (Gallegos and Varela, 2015). Roughly a third of these 1 million wells were drilled and hydraulically fractured between 2000 and 2013/2014 based on estimates from FracFocus (2016); Baker Hughes (2015); Gallegos et al. (2015); DrillingInfo (2014a); IHS Inc. (2014). This timeframe marks the beginning of modern hydraulic fracturing (refer to Text Box 3-1). Figure 3-15 shows the location of the approximately 275,000 oil and gas wells that were drilled and hydraulically fractured between 2000 and 2013 across the United States based on well and locational data from DrillingInfo (DrillingInfo, 2014a).

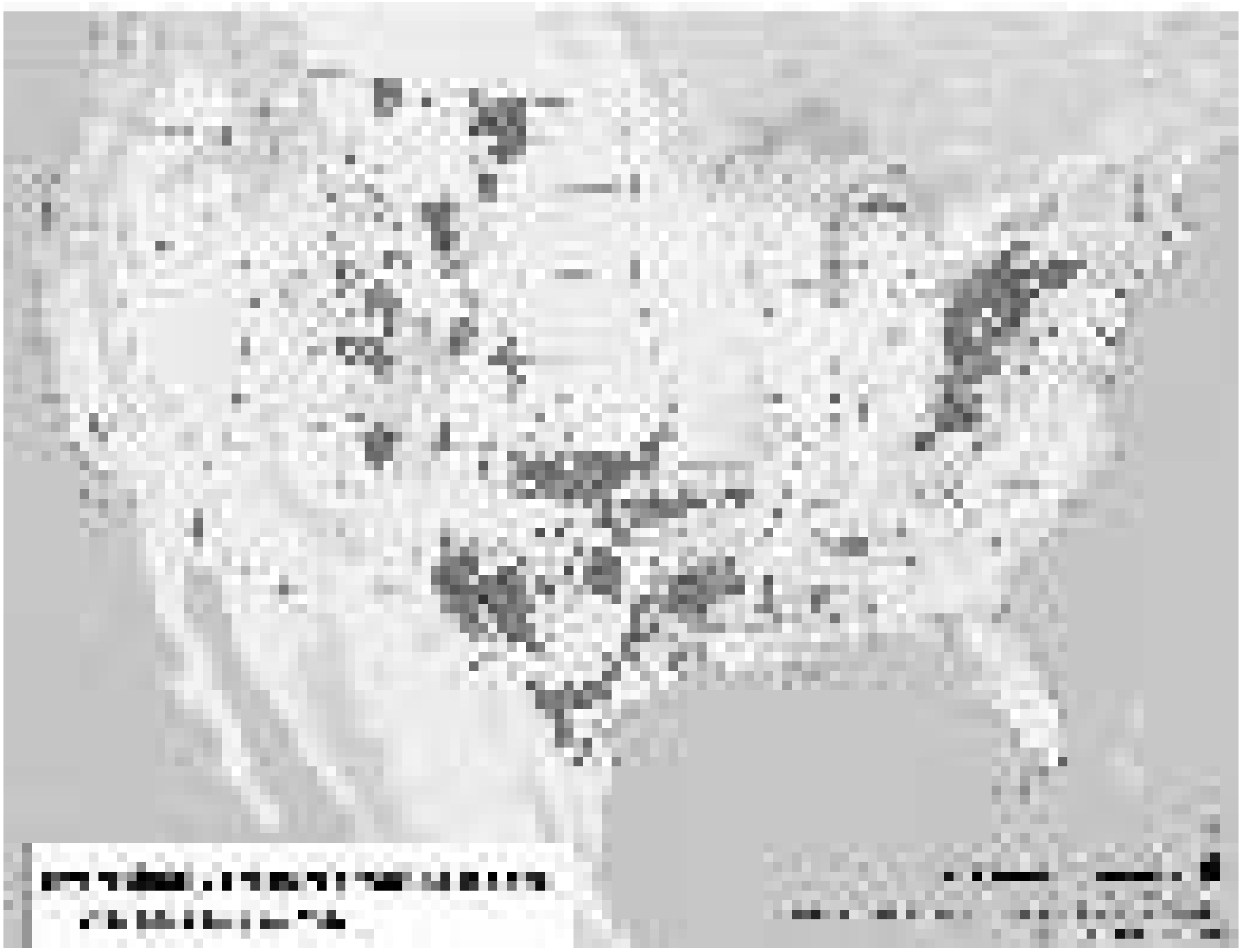

Figure 3-15. Locations of the approximately 275,000 wells drilled and hydraulically fractured between 2000 and 2013.
Based on data from the DrillingInfo Database.

The following two satellite photographs show hydraulic fracturing well sites in a regional context. These Landsat images show the locations, number, and density of hydraulic fracturing well sites across landscapes in northwest Louisiana (Figure 3-16) and western Wyoming (Figure 3-17). The orange circles around some of the well sites identify them as operations for which well information was reported to the FracFocus 1.0 registry and included in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c). Note that some of the well sites in the Landsat images, taken in 2014, are for wells that were constructed after the development of the EPA FracFocus 1.0 project database.

Figure 3-16. Landsat photo showing hydraulic fracturing well sites near Frierson, Louisiana. Imagery from USGS Earth Resources Observation and Science, Landsat 8 Operational Land Imager (scene LC80250382014232LGN00) captured 8/20/2014, accessed 5/1/2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/). Inset imagery from United States Department of Agriculture National Agriculture Imagery Program (entity M 3209351_NE 15_1_20130703_20131107) captured 7/3/2013, accessed 5/1/2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/). FracFocus well locations are from the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

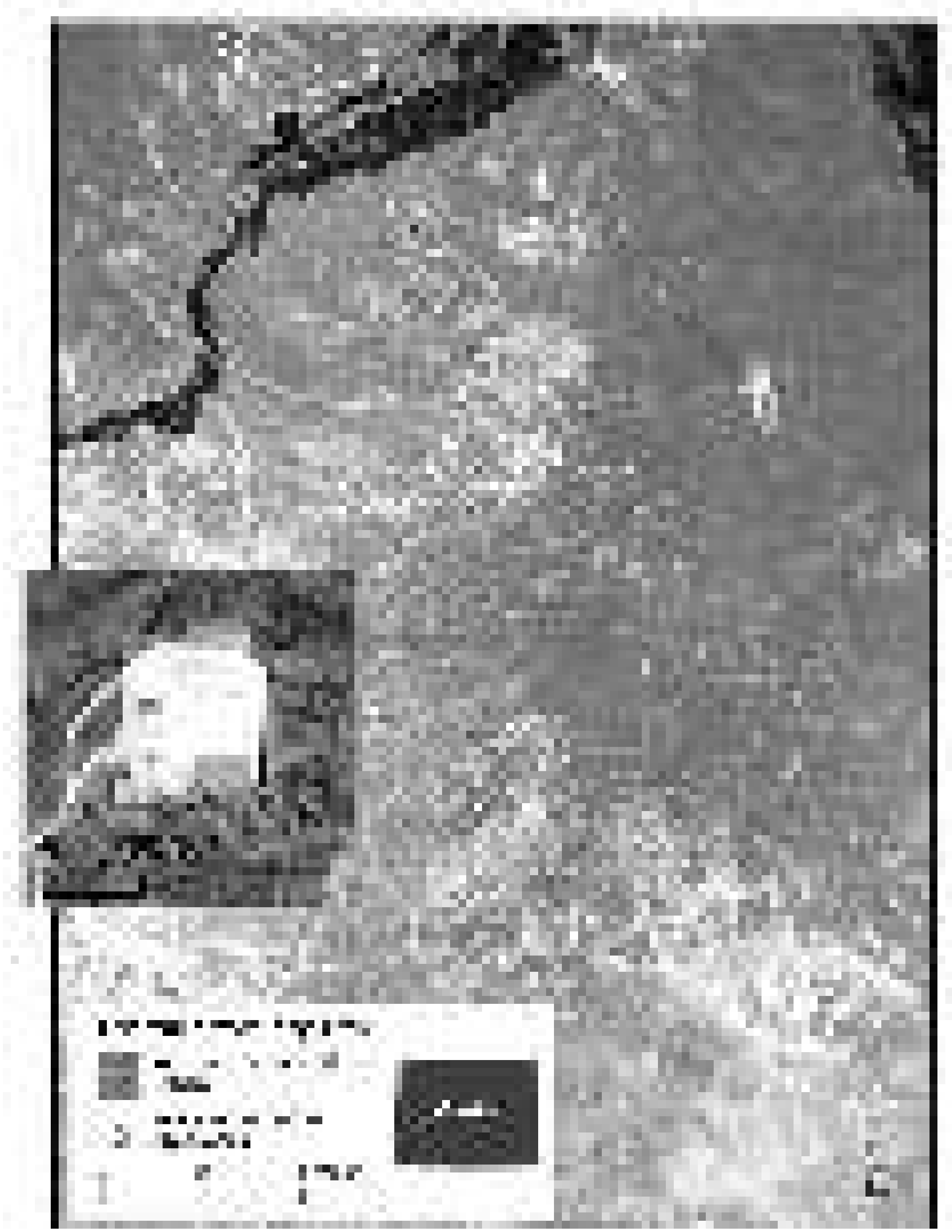

Figure 3-17. Landsat photo showing hydraulic fracturing well sites near Pinedale, Wyoming.
Imagery from USGS Earth Resources Observation and Science, Landsat 8 Operational Land Imager (scene LC80370302014188LGN00) captured 7/7/2014, accessed 5/1/2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/). Inset imagery from United States Department of Agriculture National Agriculture Imagery Program (entity M 4210927_NW 12_1_20120623_20121004) captured 6/23/2012, accessed 5/1/2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/). FracFocus well locations are from the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

3.4.1 Number of Wells Fractured per Year

Approximately 25,000 to 30,000 new oil and gas wells were hydraulically fractured each year in the United States between 2011 and 2014 based on data from several commercial data sets and publicly available data from organizations that track drilling and hydraulic fracturing activities (Table 3-1). These estimates do not include fracturing activity in older, existing wells (wells more than one-year old that may or may not have been hydraulically fractured in the past). Likely following the decline in oil prices (starting in about 2014) and gas prices (in about 2008), the estimated number of new hydraulically fractured wells declined to about 20,000 in 2015 according to well information submitted to FracFocus (FracFocus, 2016). Future drilling activity and the annual number of new wells will be influenced by future oil and gas prices.

Table 3-1. Estimated number of new wells hydraulically fractured nationally by year from various sources.

Data from FracFocus (2016); Baker Hughes (2015); DrillingInfo (2014a); IHS Inc. (2014) as provided in Gallegos et al. (2015).

Data Source	2011	2012	2013	2014
IHS	29,650	31,073	29,114	11,980 [a]
DrillingInfo	23,144	22,865	15,903 [b]	NA
Baker Hughes	NA	24,948	25,368	26,548
FracFocus [c]	14,025	22,471	26,400	28,285

[a] The IHS well count for 2014 is incomplete as it represents data only for 8 months (January through August).

[b] The DrillingInfo well count for 2013 is incomplete because some months are missing from some state data sets.

[c] The FracFocus 2011 and 2012 counts are underestimates because reporting well information to FracFocus was voluntary when it began in 2011. The number of states requiring reporting to FracFocus has increased over time. See FracFocus discussion below. The FracFocus well counts for 2011 and 2012 are from the EPA FracFocus 1.0 project database (U.S. EPA, 2015c) developed from the FracFocus national registry, and the FracFocus counts for 2013 and 2014 are from (FracFocus, 2016).

The Information Handling Services (IHS) annual well count estimates presented in Table 3-1 are from IHS data made available in a U.S. Geological Survey publication that evaluated well data from 2000 to 2014 (Gallegos et al., 2015). The IHS data are compiled from a variety of public and private sources and are commercially available from IHS Energy. A well is identified as a hydraulic fracturing well apparently based on well operational information. Gallegos et al. (2015) estimated, based on the IHS data, that approximately 371,000 wells were hydraulically fractured between January 2000 and August 2014.

DrillingInfo, another commercial database, is developed using data obtained from individual state oil and gas agencies (DrillingInfo, 2014a). Because DrillingInfo data does not identify whether a well has been hydraulically fractured, EPA relied on information about well orientation and the oil- or gas-producing rock formation type to infer which wells were likely hydraulically fractured. This is a similar approach to that used by the EPA for estimating oil and gas well counts for its greenhouse gas inventory work (U.S. EPA, 2013c).[1] Using this approach, we estimate from the DrillingInfo data the annual numbers presented in Table 3-1 above and also estimate that a total of approximately 275,000 oil and gas wells were drilled and hydraulically fractured between 2000 and 2013.[2]

Well counts tracked by Baker Hughes provide another estimate of new wells fractured annually. This field service company compiles new-well information based on its extensive field work in oil and gas producing areas and through state agencies. Baker Hughes started compiling this publicly available well count data in 2012, but stopped in 2014. The well count data are categorized into 14 basins containing reservoirs that are mostly unconventional (and, therefore, likely hydraulically fractured wells) and one "other" category (Baker Hughes, 2015). The well count estimates in the table above are for the 14 basins and, therefore, are considered estimates of new wells hydraulically fractured in each year.

FracFocus is a national registry for operators of hydraulically fractured oil and gas wells to report information about well location and depth, date of operations, and water and chemical use. The registry, publicly accessible online (www.fracfocus.org), was developed by the Groundwater Protection Council and the Interstate Oil and Gas Compact Commission. Submission of information to FracFocus was initially voluntary (starting in January 2011), but many states now require reporting of hydraulic fracturing well activities to FracFocus. As of May 2015, 23 states required reporting to FracFocus (Konschnik and Dayalu, 2016). The annual well counts in the table above are from the EPA FracFocus 1.0 project database for 2011 and 2012 (U.S. EPA, 2015c) and from the FracFocus 2016 Quarterly Report for 2013 and 2014 (FracFocus, 2016). The well counts in the earliest years are underestimates because not all states required oil and gas well operators to submit hydraulic fracturing data to FracFocus.[3] The FracFocus registry has undergone several updates since its launch in 2011. For more details on FracFocus, see FracFocus (2016), Konschnik and Dayalu (2016), U.S. EPA (2015a), U.S. EPA (2015c), and DOE (2014a).[4]

In addition to these new well counts, some portion of existing wells are also re-fractured. Several studies indicate that re-fracturing occurs in less than 2% of wells. Shires and Lev-On (2012)

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[1] Using the DrillingInfo data, EPA assumed that all horizontal wells were hydraulically fractured in the year they started producing and assumed that all wells within a shale, coalbed, or low-permeability formation, regardless of well orientation, were hydraulically fractured in the year they started producing. More details are provided in (U.S. EPA, 2013c). Not all coalbed methane wells are hydraulically fractured, but coalbed methane wells represent gas production that sometimes uses hydraulic fracturing. Given the small percent of coalbed methane wells relative to all hydraulically fractured wells and the lack of data that distinguishes whether or not coalbed wells are hydraulically fractured, EPA included coalbed production wells into all counts of wells that are hydraulically fractured.

[2] The different well count totals from IHS and DrillingInfo are likely due to different sources of data, different approaches for defining hydraulically fractured wells in those sources, and somewhat different timeframes. The higher IHS count likely includes hydraulically fractured vertical and deviated wells in conventional reservoirs (the DrillingInfo estimate does not) and covers a time period that is a year or more longer.

[3] We compared state records of hydraulic fracturing wells in North Dakota, Pennsylvania, and West Virginia in 2011 and 2012 to those reported to FracFocus during those same years and found the FracFocus wells counts underestimated the number of fracturing jobs in those states by approximately 30% on average. See Chapter 4, Text Box 4-1.

[4] Analyses of the FracFocus data based on the EPA FracFocus 1.0 project database (U.S. EPA, 2015c) are presented in Chapter 4 regarding water volumes and in Chapter 5 regarding chemical use.

suggested that the rate of re-fracturing in natural gas wells is about 1.6% whereas analysis for the EPA's 2012 Oil and Gas Sector New Source Performance Standards indicated a re-fracture rate of 1% for gas wells (U.S. EPA, 2012f). The percentage of hydraulically fractured producing gas wells that were re-fractured in a given year ranged from 0.3% to 1% across the 1990-2013 period according to the EPA's Inventory of U.S. Greenhouse Gas Emissions and Sinks (U.S. EPA, 2015h).

The above rates are calculated by comparing the number of re-fractured wells in a single year to all hydraulically fractured wells cumulatively over a multi-year time period. However, when calculating the rates of wells that conduct re-fracturing in a given year compared to the total number of wells in that same year, the re-fracturing rate is higher. Data provided to the EPA's Greenhouse Gas Reporting Program (GHGRP) for 2011 to 2013 suggest that 9-14% of the gas wells hydraulically fractured *in each year* were pre-existing wells undergoing re-fracturing (U.S. EPA, 2014b).[1] Another rate presenting a somewhat different measure (estimated by an EPA review of well records from 2009 to 2010) found that 16% of the surveyed wells had been re-fractured at least once (U.S. EPA, 2016c).[2]

In summary, a complete count of the number of hydraulically fractured wells in the United States is hampered by a lack of a definitive and readily accessible source of information, and the fact that existing well and drilling databases and registries track information differently and therefore are not entirely comparable. There is also uncertainty about whether existing information sources are representative of the nation (or parts of the nation), whether they include data for all production well types, and to what degree they include activities in both conventional and unconventional reservoirs. Taking these limitations into account, however, it is reasonable to conclude that between approximately 25,000 and 30,000 new wells (and, likely, additional pre-existing wells) were hydraulically fractured each year in the United States from about 2011 to 2014, and approximately 20,000 wells were hydraulically fractured in 2015.

3.4.2 Hydraulic Fracturing Rates

Estimates of hydraulic fracturing rates, or the proportion of all oil and gas production wells that are hydraulically fractured, also indicate widespread use of the practice. Data from IHS Inc. (2014) indicate that approximately 62% of all new oil and gas wells in 2013 were hydraulically fractured. Data from DrillingInfo (2014a), indicate a similar rate of 64% of all new production wells in 2012. Estimates of hydraulic fracturing rates reported by states in response to an IOGCC survey tended to be considerably higher. Of eleven oil and gas producing states that responded to the survey, ten (Arkansas, Colorado, New Mexico, North Dakota, Ohio, Oklahoma, Pennsylvania, Texas, Utah, and West Virginia) estimated that 78% to 99% of new wells in their states were hydraulically fractured in 2012. Louisiana was the one exception, reporting a fracturing rate of 3.9% (IOGCC, 2015).

Hydraulic fracturing may be more prevalent in gas wells than in oil wells. A 2010 to 2011 survey of 20 natural gas production companies reported that 94% of the gas wells that they operated were

[1] The GHGRP reporting category that covers re-fracturing is "workovers with hydraulic fracturing." This re-fracturing data is for gas wells only (and does not include oil wells).

[2] This EPA report is based on a statistical survey so there is some uncertainty and a margin of error regarding the 16% re-fracturing rate. This rate includes both oil and gas wells. For more details, see Chapter 6 and U.S. EPA (2016).

fractured (Shires and Lev-On, 2012), a rate that is higher than many of the reported statistics for oil and gas together (presented in the previous paragraph). Recent EIA data on the portion of oil and gas production attributable to hydraulically fractured wells also suggest possibly higher rates of hydraulic fracturing for gas. In 2015, production from hydraulically fractured wells accounted for an estimated 67% of natural gas production (EIA, 2016d) and 51% of oil production (EIA, 2016c).

3.5 Trends and Outlook for the Future

Future oil and gas drilling and production activities, including hydraulic fracturing, will be primarily affected by the cost of well operation (partly driven by technology) and the price of oil and gas. Scenarios of increasing, stable, and decreasing hydraulic fracturing activity all appear to be possible (Weijermars, 2014). The section below provides some discussion on trends and future prospects for production quantities and locations.

Fossil fuels–oil, gas, and coal–have been dominant energy sources in the United States over the last half century (Figure 3-18). The relative importance of oil, gas, and coal has changed several times, with a significant recent shift starting in the mid-2000s as hydraulic fracturing transformed oil and gas production. Coal, the leading fossil fuel from the mid-1980s to the mid-2000s, has experienced a large decrease in production, dropping from approximately 33% of U.S. energy production in 2007 to approximately 20% (about 18 quadrillion Btus) by the end of 2015 (EIA, 2016a).[1] In contrast, natural gas production has risen to unprecedented levels, and oil production has resurged to levels not seen since the 1980s. Oil accounted for 15% of U.S. energy production in 2007 and increased to approximately 23% (about 20 quadrillion Btus) by the end of 2015, and natural gas as a portion of domestic energy production went from 31% to 37% (about 33 quadrillion Btus) (EIA, 2016a).

Figure 3-18. Primary U.S. energy production by source, 1950 to 2015.
Source: EIA (2016a).

[1] A Btu, or British thermal unit, is a measure of the heat (or energy) content of fuels. At the scale of national U.S. production, the graph in Figure 3-18 presents Btus in quadrillions, or a thousand million million (which is 10^{15}, or a 1 with 15 zeros).

The surge in both oil and gas production started in the mid- to late-2000s and was driven by market forces (supply and demand) and the related developments in and expanded use of hydraulic fracturing. Figure 3-19 focuses on the years 2000 to 2015 and presents data showing the steady increase in the portion of oil and gas production coming from hydraulically fractured wells. Oil and gas production associated with hydraulic fracturing was insignificant in 2000, but by 2015 it accounted for an estimated 51% of US oil production and 67% of US gas production (Figure 3-19).

Figure 3-19. U.S. production of oil (left) and gas (right) from hydraulically fractured wells from 2000 to 2015.
Source: EIA (2016c) (oil) and EIA (2016d) (gas), based on IHS Global Insight and DrillingInfo, Inc.

Hydraulic fracturing activities are concentrated geographically in the United States. In 2011 and 2012 about half of hydraulic fracturing wells were located in Texas with another quarter located in the four states of Colorado, Pennsylvania, North Dakota, and Oklahoma (U.S. EPA, 2015c). The maps in Figure 3-20 show changes starting in 2000 in the national geography of oil and gas production through the increased use of horizontal drilling, which frequently is associated with hydraulic fracturing. Some traditional oil- and gas-producing parts of the country, such as Texas, have seen an expansion of historical production activity as a result of modern hydraulic fracturing. Pennsylvania, a leading oil- and gas-producing state a century ago, has seen a resurgence in oil and gas activity. Other states that experienced a steep increase in production, such as North Dakota, Arkansas, and Montana, have historically produced less oil and gas.

3.5.1 Natural Gas

Drilling of new natural gas wells declined markedly as natural gas prices fell in 2008 (Figure 3-21). Nevertheless, over the coming decades natural gas production is expected to increase and that increase will be associated significantly with wells that are hydraulically fractured. Projections by EIA indicate that gas production from shale (and tight oil reservoirs) will almost double from 2015 to 2040 when it will constitute nearly 70% of total natural gas production (EIA, 2016d). Slight increases are projected for production from tight gas reservoirs and coalbed methane production is expected to continue fairly steady at relatively low levels (EIA, 2016a) (Figure 3-22). These projections are dependent on estimated future prices of natural gas and other assumptions, and

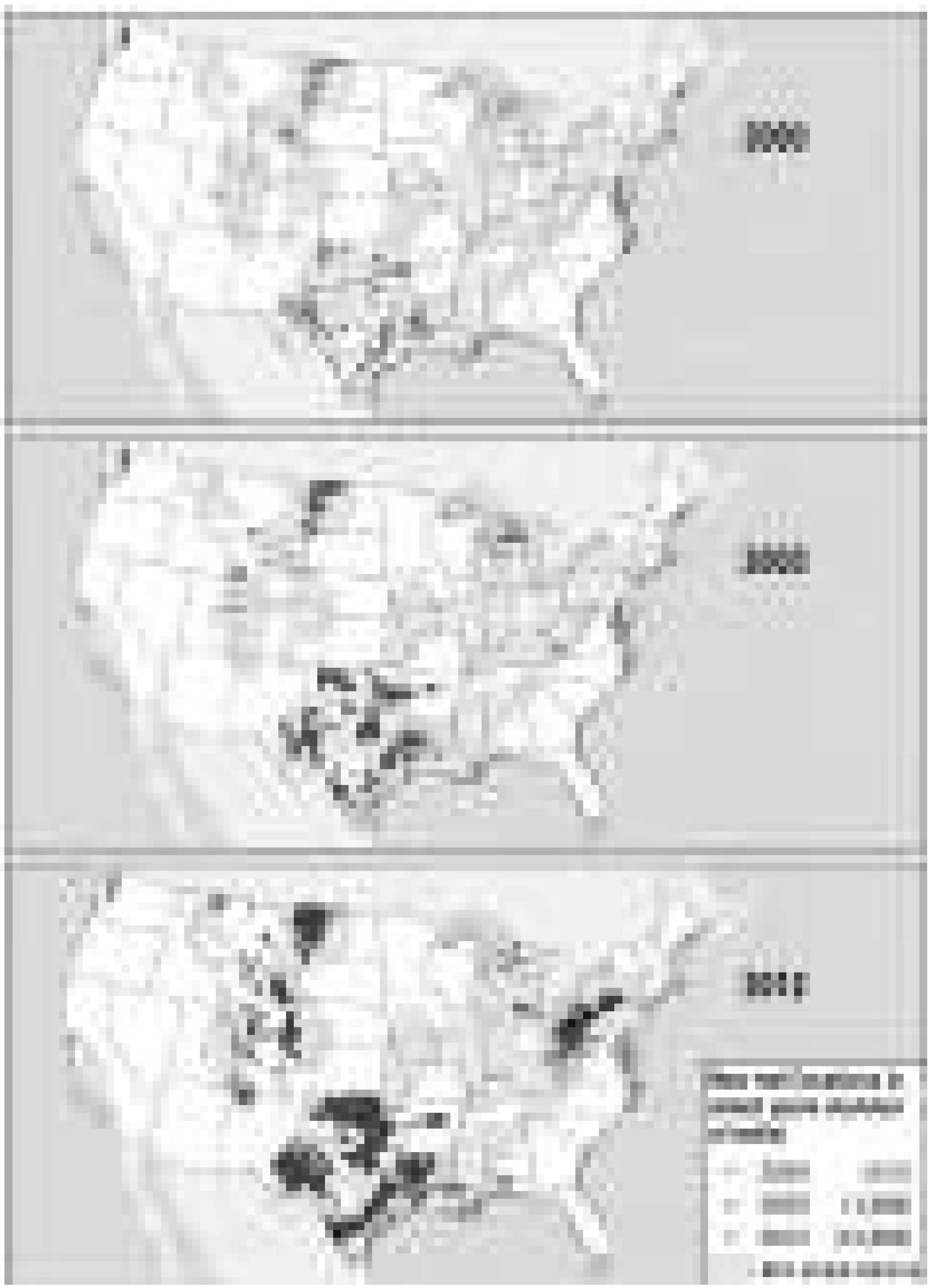

Figure 3-20. Location of horizontal wells that began producing oil or natural gas in 2000, 2005, and 2012.
Based on data from DrillingInfo (2014a).

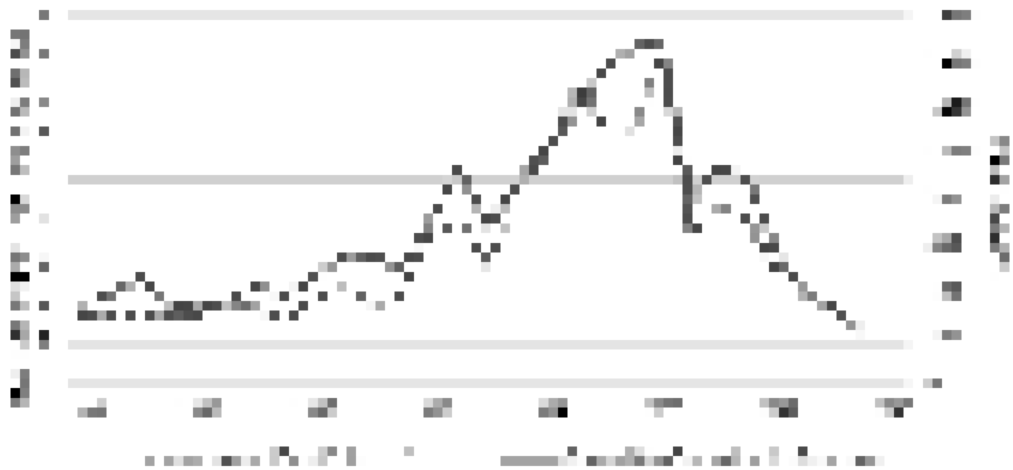

Figure 3-21. Natural gas prices and drilling activity, United States, 1988 to 2015.
Sources: EIA (2016b) and EIA (2016f).

Figure 3-22. Historic and projected natural gas production by source (trillion cubic feet).
Source: EIA (2016a).

the details are subject to change. Nonetheless, a continuing increase in production is generally suggested and the locations of historical production identified in Figure 3-23 indicate areas of continued and future hydraulic fracturing activities for natural gas production.

The geographic concentration and trends in shale gas production by play (and identified by state) are shown in Figure 3-23. The Barnett Shale, where the modern hydraulic fracturing boom started, was the largest producer of shale gas until about 2010, producing 1.5 trillion cubic feet (tcf) (42.5 billion cubic meters [bcm]) that year and remains a significant producer. In 2009, the Marcellus and Haynesville plays produced 0.12 and 0.43 tcf (3.4 and 12.2 bcm), respectively, but by 2011, production from the Haynesville play surpassed that in the Barnett play, and by 2013 the Marcellus Shale surpassed both the Barnett and the Haynesville to become the play with the most production.

By 2014, the Marcellus play was producing 4.8 tcf (135.9 bcm) of gas annually, with the Eagle Ford, Haynesville, and Barnett each producing roughly 1.5 tcf (42.5 bcm). Estimates of technically recoverable resources, a general indicator of potential future production, are noted for the Marcellus (about 150 tcf [4.25 trillion cubic meters]), Haynesville (73 tcf [2.07 trillion cubic meters]), Eagle Ford in Texas (55 tcf [1.56 trillion cubic meters]), and Utica in Ohio, Pennsylvania, and West Virginia (55 tcf [1.56 trillion cubic meters]). This suggests that these four plays will be active contributors of shale gas production for the foreseeable future (EIA, 2013).[1] Other gas plays with significant resources include the Fayetteville in Arkansas, the Woodford in Oklahoma, and the Mancos in Colorado.

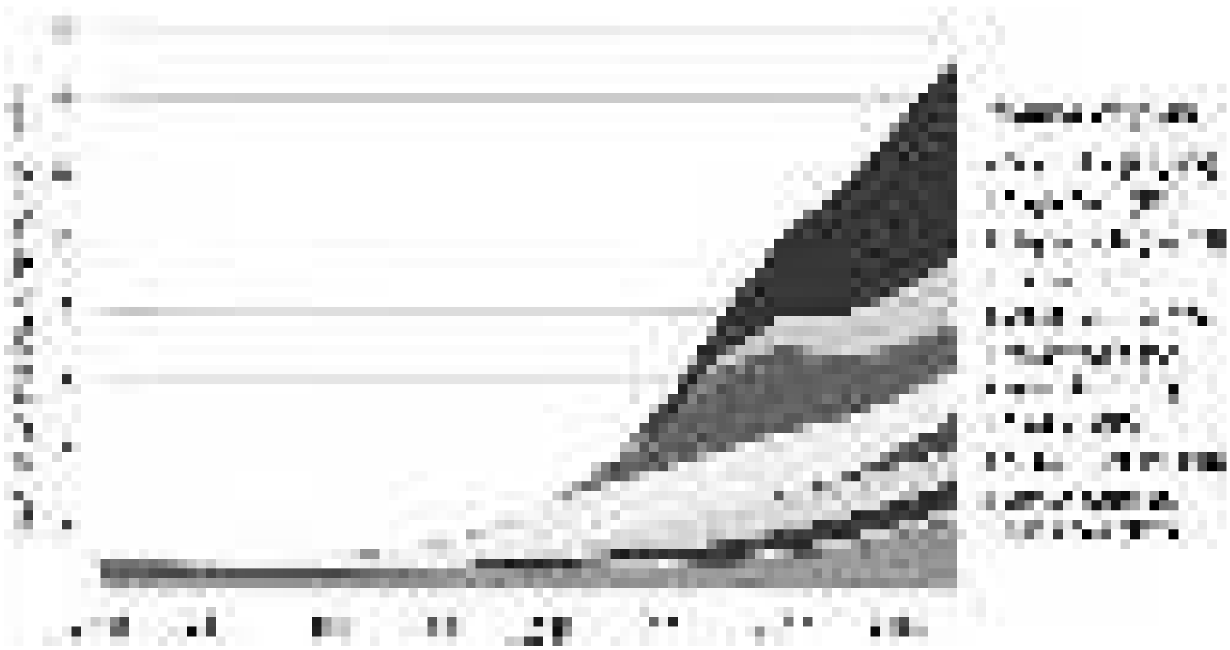

Figure 3-23. Production from U.S. shale gas plays, 2000-2014.
Source: EIA (2016g). The graph shows shale plays in the same vertical order as the legend.

3.5.2 Oil

While prices and drilling activity for natural gas were peaking between 2005 and 2008 and then falling (Figure 3-21), prices and drilling for oil were rising. These peaked between 2011 and 2014, and then rapidly declined as well (Figure 3-24). EIA projections to 2040 indicate a continued growth in total U.S. oil production, although the projected growth is not as fast or as large as that projected for natural gas. Tight oil production, presumably from hydraulically fractured wells, is expected to account for much of the projected growth (Figure 3-25); by 2040, tight oil is expected to account for nearly 65% of all U.S. crude oil production (EIA, 2016d). These production projections are dependent on estimated future prices of oil and other assumptions and, therefore, will likely be revised over time as energy markets and prices change. Currently, these projections

[1] Technically recoverable resources are the volumes of oil or natural gas that could be produced with current technology, regardless of oil and natural gas prices and production costs (EIA, 2013).

indicate a continuing, but lower rate of growth (as compared to the period from about 2005 to 2015). The locations of historical production identified in Figure 3-26 indicate areas of continued and future hydraulic fracturing activities for oil.

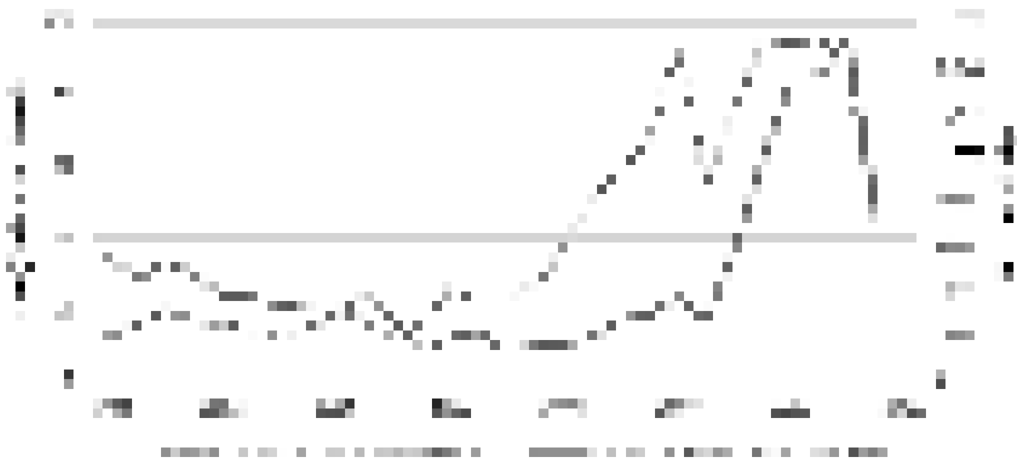

Figure 3-24. Crude oil prices and drilling activity, United States, 1988 to 2015.
Sources: EIA (2016b) and EIA (2016e).

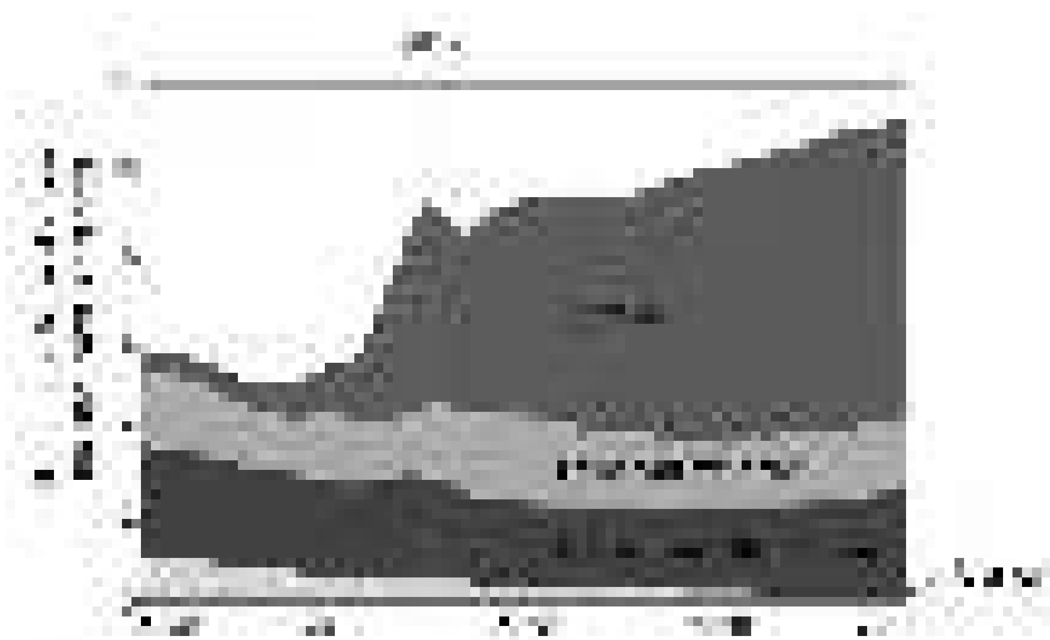

Figure 3-25. Historic and projected oil production by source (million barrels per day).
Source: EIA (2016a).

The geographic concentration and trends in tight oil production by play (and identified by state) are shown in Figure 3-26. Early tight oil production in the United States was centered in the Permian Basin in west Texas and New Mexico, at plays that included the Spraberry and the Bonespring. After 2009, the Bakken play (centered in western North Dakota) and the Eagle Ford play (in Texas) emerged as the largest-producing tight oil plays. Oil production in the Bakken increased from 99 million bbls (16,000 million L) in 2009 to 394 million bbls (63,600 million L) in 2014 (EIA, 2016g). Production from Eagle Ford increased from 12 million bbls (2,000 million L) in 2009 to 498 million bbls (79,100 million L) in 2014 (EIA, 2016g).

General estimates of potential resources suggest that future tight oil production in the United States will continue to be led by Texas and North Dakota. Technical recoverable resources are estimated at about 23 billion bbls (3,600 billion L) for the Bakken, about 21 billion bbls (3,300 billion L) for the Permian Basin, and about 10 billion bbls (1,600 billion L) for Eagle Ford (EIA, 2015). Other plays with significant estimated resources include the Niobrara-Codell in Colorado and Wyoming and the Granite Wash in Oklahoma and Texas (EIA, 2012).

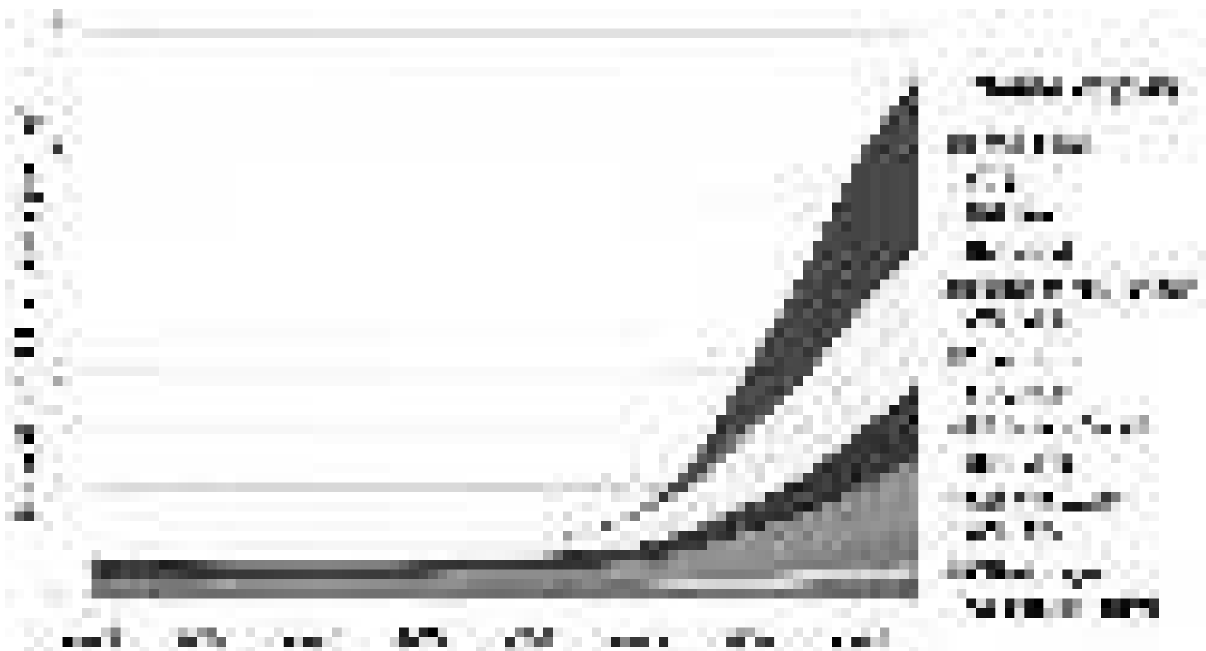

Figure 3-26. Production from U.S. tight oil plays, 2000-2014.
Source: EIA (2016g). The graph shows tight oil plays in the same vertical order as the legend.

3.6 Conclusions

Hydraulic fracturing is the injection of hydraulic fracturing fluids through the production well and into the subsurface oil or gas reservoir under pressures great enough to fracture the reservoir rock. The fractures allow for increased flow of oil and/or gas from the reservoir into the well. Water used in the hydraulic fracturing fluid is typically obtained from sources in the vicinity of the well. Water that naturally occurs in the oil and gas reservoir rocks often flows into the production well and through the well to the surface as a byproduct of the oil and gas production process. This byproduct water, commonly referred to as produced water, requires handling and management.

Many well site and operational activities are conducted to prepare a site and well for hydraulic fracturing and oil and/or gas production. The actual hydraulic fracturing event is of relatively short duration, usually several weeks or less, but it is also a phase of work with numerous complex operational activities to handle, mix, and inject the hydraulic fracturing fluid under pressure through the production well. The injected hydraulic fracturing fluid typically contains mostly water, includes a proppant (commonly sand) which ensures that the fractures remain propped open after injection, and contains less than two percent additives (chemicals) that improve the fluid properties for fracturing. These small percentages of additives, given the total volume of hydraulic fracturing fluids, mean that a typical hydraulic fracturing job can use tens of thousands of gallons of chemicals.

Since about 2005, the combination of hydraulic fracturing and directional drilling pioneered in the Barnett Shale in Texas has become widespread in the oil and gas industry. Hydraulic fracturing combined with directional drilling is now a standard industry practice. It has significantly contributed to the surge in United States oil and gas production, and accounted for slightly more than 50% of oil production and nearly 70% of gas production in 2015. Hydraulic fracturing has resulted in expanded production from unconventional shale and so-called tight oil or gas reservoirs that had previously been largely unused. This hydraulic fracturing-based production activity is geographically concentrated. About three-quarters of new hydraulic fracturing wells in 2011 and 2012 were located in five states (Texas, Colorado, Pennsylvania, North Dakota, and Oklahoma) with about half of all wells located in Texas.

There is no national database or complete national registry of wells that have been hydraulically fractured in the United States. Based on the data available from various commercial and public sources, we estimate that 25,000 to 30,000 new wells were drilled and hydraulically fractured in the United States annually between 2011 and 2014. In addition to these new wells, some existing wells not previously fractured were fractured, and some that had been fractured in the past were re-fractured. New drilling of hydraulic fracturing wells, influenced by oil and gas prices, peaked in the United States between 2005 and 2008 for gas and between 2011 and 2014 for oil. Following price declines, the number of new hydraulically fractured wells in 2015 was about 20,000. Future drilling and production will be influenced by future gas and oil prices. Despite recent declines in prices and new drilling, oil and gas production in the United States continues at historically high levels with projections of continued growth in the medium and long term led by hydraulic fracturing-based production from unconventional reservoirs.

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Chapter 4. Water Acquisition

Abstract

In this chapter, the EPA examined the potential impacts of water withdrawals for hydraulic fracturing on drinking water resource quantity and quality, and identified common factors affecting the frequency and severity of impacts. Groundwater and surface water resources used for hydraulic fracturing also currently serve or in the future may serve as drinking water sources, and water withdrawals for hydraulic fracturing can affect the quantity or quality of the remaining drinking water resource.

Hydraulic fracturing used a median of 1.5 million gallons (5.7 million liters) of water per well from 2011 through early 2013. Surface water supplies almost all water used for hydraulic fracturing in the eastern United States, whereas surface water or groundwater is used in the West. Reuse of hydraulic fracturing wastewater as a percentage of injected volume is generally low, with a median of 5% according to an EPA literature survey. Greater reuse occurs where disposal options are limited (e.g., the Marcellus Shale in Pennsylvania) and not necessarily where water availability is lowest.

Hydraulic fracturing generally uses and consumes a relatively small percentage of water when compared to total water use, water consumption, and water availability at the national, state, and county scale. There are exceptions, however. For example, EPA's analysis shows that counties in southern and western Texas have relatively high hydraulic fracturing water withdrawals and low water availability. These findings indicate where impacts are more likely to occur or be severe, but local information (i.e., at the scale of the drinking water resource) is needed to determine whether potential impacts have been realized. In some example cases (e.g., the Eagle Ford Shale in Texas, the Haynesville Shale in Louisiana), local impacts to drinking water resource quantity have occurred in areas with increased hydraulic fracturing activity. In these instances, hydraulic fracturing water withdrawals contributed to local impacts along with other water users and the lack of precipitation.

Drought or seasonal times of low water availability can increase the frequency and severity of impacts, while water management practices such as the establishment of low-flow criteria (termed passby flows), shifting from fresh to brackish water sources, or increasing the reuse of wastewater for hydraulic fracturing can help protect drinking water resources.

Uncertainty about the extent of impacts on drinking water resources stems from the lack of available data at the local scale. The EPA could assess the potential for impacts at the county scale, but often could not determine whether impacts occurred at drinking water withdrawal locations.

Overall, hydraulic fracturing uses and consumes a relatively small percentage of water at the county scale, but not always, and impacts can still occur depending on the local balance between withdrawals and availability. Regional or local-scale factors, such as drought or water management, can modify this balance to increase or decrease the frequency or severity of impacts.

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4. Water Acquisition

4.1 Introduction

Water is a crucial component of nearly all hydraulic fracturing operations, generally making up 90 – 97% of the total fluid volume injected into a well (Chapter 5).[1] Ground- and surface water resources that serve as sources of water for hydraulic fracturing also provide water for public water supplies or private drinking water wells. For this reason, water withdrawals for hydraulic fracturing can impact drinking water resources by changing the quantity or quality of the remaining resource.[2] In this chapter, we consider potential impacts of water acquisition for hydraulic fracturing on both drinking water resource quantity and quality, and, where possible, identify factors that affect the frequency or severity of impacts.[3]

We define impacts broadly in this assessment to include any change in the quantity or quality of drinking water resources; see Chapter 1 for more information. This definition applies reasonably well to the subsequent chapters (Chapters 5-8); however, by this definition, even the smallest water withdrawals would be considered impacts. Recognizing this, we focus on a smaller subset of potential impacts, specifically water withdrawals that have the potential to limit the availability of drinking water or alter its quality. Whether water withdrawals have this potential depends primarily on the balance between water use and availability at the local scale.[4,5] By "local" in this chapter, we refer to the scale at which impacts to drinking water resources are expected to occur. This usually means a given surface water (e.g., river or stream) or groundwater resource (i.e., aquifer), or a given watershed where we have detailed information about local water dynamics (e.g., case studies). We note the scale at which data are available and findings are reported.

[1] This range is based on multiple sources that either present hydraulic fracturing fluid composition as a function of volume (e.g., 95% of the total volume injected) or as a function of mass (e.g., 90% of the total mass injected). See Chapter 5 for additional information.

[2] Surface water withdrawals can affect water quality by altering in-stream flow and decreasing the dilution of pollutants or changing water chemistry (Section 4.5.3). Groundwater withdrawals may alter water quality by inducing vertical mixing of fresh groundwater with contaminated water from the land surface or underlying formations, or by promoting changes in reduction-oxidation conditions and mobilizing chemicals from geologic sources (Section 4.5.1).

[3] Water acquired for use in other oil and gas development steps besides hydraulic fracturing is beyond the scope of this chapter, including the water used in well drilling and well pad preparation and water removal for the production of coalbed methane. Furthermore, water released to the atmosphere via gas combustion is also outside the scope of this chapter.

[4] Throughout this chapter we use the terms "water use" and "water withdrawals" interchangeably to refer to the water that is acquired for hydraulic fracturing operations.

[5] There is no standard definition for water availability, and it has not been assessed recently at the national scale (U.S. GAO, 2014). Instead, a number of water availability indicators have been suggested (e.g., Roy et al., 2005). Here, availability is most often used to qualitatively refer to the amount of a location's water that could, currently or in the future, serve as a source of drinking water (U.S. GAO, 2014), which is a function of water inputs to a hydrologic system (e.g., rain, snowmelt, groundwater recharge) and water outputs from that system occurring either naturally or through competing demands of users. Where specific numbers are presented, we note the specific water availability indicator used.

A variety of factors can modify the balance between water use and availability. For example, multiple hydraulically fractured wells require more water than a single well, making it critical to assess the cumulative effects of multiple wells over a given area or time period. Furthermore, the combined effects of multiple water users pumping from the same aquifer can compound stress on already declining groundwater supplies. Alternatively, locally high rates of hydraulic fracturing wastewater reuse may help offset the need for fresh water withdrawals. These and other factors are discussed throughout the chapter.

This chapter proceeds roughly in two halves. In the first half, we address water use and consumption by hydraulic fracturing.[1] We provide an overview of the types of water used for hydraulic fracturing (Section 4.2); the amount of water used per well (Section 4.3); and then estimates of hydraulic fracturing water use and consumption at the national, state, and county scale, both in absolute terms and relative to total water use and consumption (Section 4.4). Although most available data and literature pertain to water use, we discuss water consumption because hydraulic fracturing consumes a substantial proportion of the water it uses, so that a proportion of the water is lost from the local hydrologic cycle. See Section 4.4 and Chapter 2 for more information.

In the second half of the chapter, we assess the potential for impacts by location in certain states (and major oil and gas regions within select states) where hydraulic fracturing currently occurs (Section 4.5; Appendix B.2). For each state and region, we discuss the water used and consumed by hydraulic fracturing, and then compare it to water availability. We do this using several lines of evidence: (1) literature information (both quantitative and qualitative) on state and regional hydraulic fracturing water use and availability; (2) comparisons between our county level estimates of hydraulic fracturing water use and an index of water availability; and (3) local case studies from the Eagle Ford play in Texas, the Upper Colorado River Basin in Colorado, and the Susquehanna River Basin in Pennsylvania.[2] The use of case studies provides insight into the local, sub-county scale, where impacts are most likely to be observed in both space and time.

Overall, this chapter provides a national assessment of where potential impacts to drinking water quantity and quality are most likely due to water acquisition for hydraulic fracturing. We utilize case studies where data are available to understand local dynamics and whether impacts are indeed realized. In the absence of case studies, we use county level data to assess where potential impacts are most likely. Finally, we identify the common factors affecting the frequency and severity of impacts. We provide a synthesis of our findings in Section 4.6.

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[1] We refer specifically to "water consumption" when data are available or it is explicitly noted in the scientific literature. However, when specific information is not available, we use "water use" or "water withdrawals" as general terms to refer to both water use and consumption by hydraulic fracturing.

[2] The EPA's *Review of Well Operator Files for Hydraulically Fractured Oil and Gas Production Wells* (i.e., the "Well File Review;" see Text Box 6-1) was originally planned to inform the water acquisition stage of the hydraulic fracturing water cycle, but did not yield any useable information on this topic, and is therefore not cited as a source of information in this chapter. Although information in some well files was of good quality, the well files generally contained insufficient or inconsistent information on nearby surface water and groundwater quality, injected water volumes, and wastewater volumes and disposition; therefore, these data were not summarized (U.S. EPA, 2015n).

4.2 Types of Water Used

The three major sources of water for hydraulic fracturing are surface water (i.e., rivers, streams, lakes, and reservoirs), groundwater, and reused hydraulic fracturing wastewater.[1,2,3] These sources often vary in their initial water quality and in how they are provisioned to hydraulic fracturing operations. In this section, we provide an overview of the sources (Section 4.2.1), water quality (Section 4.2.2), and provisioning of water (Section 4.2.3) required for hydraulic fracturing. Detailed information on the types of water used by state and locality is presented in Section 4.5.

4.2.1 Source

Whether water used in hydraulic fracturing originates from surface water or groundwater resources is largely determined by the type of locally available water sources. Water transportation costs can be high, so the industry tends to acquire water from nearby sources if available (Nicot et al., 2014; Mitchell et al., 2013a; Kargbo et al., 2010). Surface water supplies most of the water for hydraulic fracturing in the eastern United States, whereas surface water or groundwater is used in the more semi-arid to arid western states. In western states that lack available surface water resources, groundwater generally supplies the majority of water needed for fracturing (Table 4-1). Brackish sources of groundwater can be important for reducing demand on fresh groundwater resources in certain regions (e.g., the Permian Basin and Eagle Ford Shale in Texas; see Section 4.5.1).[4] Local policies also may direct the industry to seek withdrawals from designated sources (U.S. EPA, 2013a); for instance, some states have encouraged the industry to seek water withdrawals from surface water rather than groundwater due to concerns over aquifer depletion. See Section 4.5.4 and Section 4.5.5 for more information.

[1] We use the term "hydraulic fracturing wastewater" to refer to produced water that is managed using practices that include, but are not limited to, reuse in subsequent hydraulic fracturing operations, treatment and discharge, and injection into disposal wells. The term is being used in this study as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes (see Chapter 8 and Appendix J, the Glossary, for more detail).

[2] Throughout this chapter we sometimes refer to "reused hydraulic fracturing wastewater" as simply "reused wastewater," because this is the dominant type of wastewater reused by the industry. When referring to other types of reused wastewater not associated with hydraulic fracturing (e.g., acid mine drainage, wastewater treatment plant effluent), we specify the source of the wastewater.

[3] We use the term "reuse" regardless of the extent to which the wastewater is treated (Nicot et al., 2014); we do not distinguish between reuse and recycling except when specifically reported in the literature.

[4] We use the term "fresh water" to qualitatively refer to water with relatively low TDS that is most readily and currently available for drinking water. We do not use the term to imply an exact TDS limit.

Table 4-1. Estimated proportions of hydraulic fracturing source water from surface water and groundwater.

Location	Surface water	Groundwater	Year or time period of estimate
Louisiana—Haynesville Shale	87%[a]	13%[a]	2009 - 2012
Oklahoma—Statewide	63%[b]	37%[b]	2011
Pennsylvania—Marcellus Shale, Susquehanna River Basin	92%[c]	8%[c]	2008 - 2013
Texas—Barnett Shale	50%[d]	50%[d]	2011 - 2013
Texas—Eagle Ford Shale	10%[e]	90%[e]	2011
Texas—TX-LA-MS Salt Basin[f]	30%[e]	70%[e]	2011
Texas—Permian Basin	0%[e]	100%[e]	2011
Texas—Anadarko Basin	20%[e]	80%[e]	2011
West Virginia—Statewide, Marcellus Shale	91%[g]	9%[g]	2012

[a] Percentages calculated from fracturing supply water usage data only. Rig supply water and other sources were excluded as they fall outside the scope of this assessment. Data from October 1, 2009, to February 23, 2012, for 1,959 Haynesville Shale natural gas wells (LA Ground Water Resources Commission, 2012).

[b] Proportion of surface water and groundwater permitted in 2011 by Oklahoma's 90-day provisional temporary permits for oil and gas mining. Temporary permits make up the majority of water use permits for Oklahoma oil and gas mining (Taylor, 2012).

[c] Calculated from SRBC (2016) data from July 2008 to December 2013.

[d] Nicot et al. (2014).

[e] Nicot et al. (2012).

[f] Nicot et al. (2012) refer to this region of Texas as the East Texas Basin.

[g] Estimated proportions are for 2012, the most recent estimate for a full calendar year available from West Virginia DEP (2014). Data from the West Virginia DEP show the proportion of water purchased from commercial brokers as a separate category and do not specify whether purchased water originated from surface water or groundwater. Therefore, we excluded purchased water in calculating the relative proportions of surface water and groundwater shown in Table 4-1 (West Virginia DEP, 2014).

The reuse of wastewater from past hydraulic fracturing operations reduces the need for withdrawals of fresh surface water or groundwater.[1] In a survey of literature values from 10 states, basins, or plays, we found a median of 5% of the water used in hydraulic fracturing comes from reused hydraulic fracturing wastewater, with this percentage varying by location (Table 4-2).[2,3]

[1] Hydraulic fracturing wastewater may be stored on-site in open pits, which may also collect rainwater and runoff water. We do not distinguish between the different types of water that are collected on-site during oil and gas operations, and assume that most of the water collected on-site at well pads is hydraulic fracturing wastewater.

[2] Throughout this chapter, we preferentially report medians where possible because medians are less sensitive to outlier values than averages. Where medians are not available, averages are reported.

[3] This chapter examines reused wastewater as a percentage of injected volume because reused wastewater may offset total fresh water acquired for hydraulic fracturing. In contrast, Chapter 8 of this assessment discusses the total percentage of the generated wastewater that is reused rather than managed by different means (e.g., disposal in Class II wells). This distinction is sometimes overlooked, which can lead to a misrepresentation of the extent to which wastewater is reused to offset total fresh water used for hydraulic fracturing.

Table 4-2. Percentage of injected water volume that comes from reused hydraulic fracturing wastewater in various states, basins, and plays.

See Section 4.5 and Appendix B.2 for additional discussion of reuse practices by state and locality and variation over time where data are available.

State, basin, or play	Estimate of the percentage of injected water volume that comes from reused hydraulic fracturing wastewater[a]	Year or time period of estimate (NA = not available)
California—Monterey Shale	13%[b]	2014
Colorado—Wattenberg Field, Denver-Julesburg Basin	0%[c]	NA
Pennsylvania—Statewide	19%[d]	2014
Pennsylvania–Marcellus Shale, Susquehanna River Basin	16%[e]	2008 – 2013
Texas—Barnett Shale	5%[f]	2011
Texas—Eagle Ford Shale	0%[f]	2011
Texas—TX-LA-MS Salt Basin[g]	5%[f]	2011
Texas—Permian Basin (far west portion)	0%[f]	2011
Texas—Permian Basin (Midland portion)	2%[f]	2011
Texas—Anadarko Basin	20%[f]	2011
West Virginia—Statewide	15%[h]	2012
Overall Mean[i]	8%	
Overall Median[j]	5%	

[a] All estimates in this table refer to the percentage of injected water volume that comes from reused hydraulic fracturing wastewater. However, different literature sources used slightly different terminology when referring to this percentage. In the table footnotes below, we reference the terminology reported in the literature source cited.

[b] Produced water as a percentage of total water volume for 480 well stimulations according to completion reports between January 1, 2014, and December 10, 2014 (CCST, 2015a). All but two of these stimulations were conducted in Kern County, California (the remaining two were completed in Ventura County, California). Well stimulations mostly consisted of hydraulic fracturing operations, but also included smaller numbers of matrix acidizing and acid fracturing operations (CCST, 2015a).

[c] Reflects an assumption of reuse practices by Noble Energy in the Wattenberg Field of Colorado's Denver-Julesburg Basin, as reported by Goodwin et al. (2014).

[d] Percentage of recycled water used in hydraulic fracturing in 2014 based on data from the Pennsylvania Bureau of Topographic and Geologic Survey (Schmid and Yoxtheimer, 2015). This percentage was higher at 23% in 2013, but we present the most recent estimate available in the above table. The slight decline to 19% in 2014 may be explained by the fact that some completion reports had not yet been processed when these data were published, yet the data generally show an upward trend over time in reuse as a percentage of injected volume (Schmid and Yoxtheimer, 2015).

[e] Flowback water as a percentage of total water injected from July 2008 to December 2013 (SRBC, 2016). This percentage was 22% in 2013 alone (SRBC, 2016).

[f] Estimated percentage of recycling/reused water in 2011 (Nicot et al., 2012).

[g] Nicot et al. (2012) refer to this region of Texas as the East Texas Basin.

[h] Reused fracturing water as a percentage of total water used for hydraulic fracturing in 2012, calculated from data provided by the West Virginia DEP (2014).

[i] Calculated based on the values presented in Table 4-2, excluding the value for Pennsylvania's Susquehanna River Basin to avoid double counting with the statewide value. The overall mean is not weighted by the number of wells in a given state, basin, or play.

[j] Calculated based on the values presented in Table 4-2, excluding the value for Pennsylvania's Susquehanna River Basin to avoid double counting with the statewide value. The overall median is not weighted by the number of wells in a given state, basin, or play.

Available data on reuse trends indicate increased reuse as a percentage of injected volume over time in both Pennsylvania and West Virginia, likely due to the lack of nearby disposal options in Class II injection wells regulated by the Underground Injection Control (UIC) Program (Section 4.5.3).

The reuse of wastewater for hydraulic fracturing is limited by the amount of water that returns to the surface during production (Nicot et al., 2012). In the first 10 days of well production, 5% to almost 50% of the hydraulic fracturing fluid volume can be collected, with values varying across geologic formations (Chapter 7, Table 7-1). Longer duration measurements are rare, but between 10% and 30% of the hydraulic fracturing fluid volume has been collected in the Marcellus Shale in Pennsylvania over nine years of production, while over 100% has been collected in the Barnett Shale in Texas over six years of production (Chapter 7, Table 7-2).[1] Assuming that 10% of hydraulic fracturing fluid volume is collected in the first 30 days and 100% of the wastewater is reused, it would take 10 wells to produce enough water to hydraulically fracture a new well. As more wells are hydraulically fractured in a given area, the potential for wastewater reuse increases.

The decision to reuse hydraulic fracturing wastewater appears to be driven by economics and the quality of the wastewater, and not concerns over local water availability (Section 4.2.2). Water transportation costs (i.e., trucking, piping), the availability of Class II wells, and local regulations can play a role in determining whether hydraulic fracturing wastewater is reused to offset the need for fresh water withdrawals (Schmid and Yoxtheimer, 2015). Besides hydraulic fracturing wastewater, other wastewaters may be reclaimed for use in hydraulic fracturing. These include acid mine drainage, wastewater treatment plant effluent, and other sources of industrial and municipal wastewater (Nicot et al., 2014; Ziemkiewicz et al., 2013). Limited information is available on the extent to which these other wastewaters are used.

4.2.2 Quality

Water quality is an important consideration when sourcing water for hydraulic fracturing. Fresh water is most often used to maximize hydraulic fracturing fluid performance and to ensure compatibility with the geologic formation being fractured. This finding is supported by the EPA's analysis of disclosures to the FracFocus Chemical Disclosure Registry (version 1.0; hereafter, the EPA FracFocus report) (U.S. EPA, 2015b), as well as by regional analyses from Texas (Nicot et al.,

[1] It is possible to collect over 100% of the hydraulic fracturing fluid volume because water from the formation returns to the surface along with the injected water.

2012) and the Marcellus Shale (Mitchell et al., 2013a).[1,2] Fresh water was the most commonly cited water source by companies included in an analysis of nine hydraulic fracturing service companies on their operations from 2005 to 2010 (U.S. EPA, 2013a). Three service companies noted that the majority of their water was fresh, because it required minimal testing and treatment (U.S. EPA, 2013a).[3] The majority of the nine service companies recommended testing for certain water quality parameters (pH and maximum concentrations of specific cations and anions) in order to ensure compatibility among the water, other fracturing fluid constituents, and the geologic formation (U.S. EPA, 2013a).

The reuse of hydraulic fracturing wastewater may be limited to an extent by water quality. Over the production life of a well, the quality of the wastewater produced begins to resemble the quality of the water naturally found in the geologic formation and may be characterized by high concentrations of total dissolved solids (TDS) (Goodwin et al., 2014). High concentrations of TDS and other individual dissolved constituents in wastewater, including specific cations (calcium, magnesium, iron, barium, strontium), anions (chloride, bicarbonate, phosphate, and sulfate), and microbial agents, can interfere with hydraulic fracturing fluid performance by producing scale in the borehole or by interfering with certain additives in the hydraulic fracturing fluid (e.g., high TDS may inhibit the effectiveness of friction reducers) (Gregory et al., 2011; North Dakota State Water Commission, 2010). Due to these limitations, wastewater can require treatment or blending with fresh water to meet the level of water quality desired in the hydraulic fracturing fluid formulation.[4]

Options for treating hydraulic fracturing wastewater to facilitate reuse are available and being used by the industry in some cases. For example, filter socks, centrifuge, dissolved air flotation, or settling technologies can remove suspended solids, and physical/chemical precipitation or electrocoagulation can remove dissolved metals (Schmid and Yoxtheimer, 2015). For more information on treatment of hydraulic fracturing wastewater, see Chapter 8.

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[1] The FracFocus Chemical Disclosure Registry (often referred to as FracFocus; www.fracfocus.org) is a national hydraulic fracturing chemical disclosure registry managed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission. FracFocus was created to provide the public access to reported chemicals used for hydraulic fracturing within their area. It was originally established in 2011 (version 1.0) for voluntary reporting by participating oil and gas well operators. Six of the 20 states discussed in this assessment required disclosure to FracFocus at various points between January 1, 2011, and February 28, 2013, the time period analyzed by the EPA; another three of the 20 states offered the choice of reporting to FracFocus or the state during this same time period (see Appendix Table B-5 for states and disclosure start dates) (U.S. EPA, 2015b).

[2] Of all disclosures reviewed that indicated a source of water for the hydraulic fracturing base fluid, 68% listed “fresh” as the only source of water used. Note, 29% of all disclosures considered in the EPA’s FracFocus report included information on the source of water used for the base fluid (U.S. EPA, 2015b).

[3] Service companies did not provide data on the percentage of fresh water versus non-fresh water used for hydraulic fracturing (U.S. EPA, 2013a).

[4] The EPA FracFocus report suggests that fresh water makes up the largest proportion of the base fluid when blended with water sources of lesser quality (U.S. EPA, 2015b).

4.2.3 Provisioning

Water for hydraulic fracturing is typically either self-supplied by the industry or purchased from public water systems.[1] Self-supplied water for fracturing generally refers to permitted direct withdrawals from surface water or groundwater or the reuse of wastewater. Nationally, self-supplied water is more common, although there is much regional variation (U.S. EPA, 2015b; CCST, 2014; Mitchell et al., 2013a; Nicot et al., 2012). Water purchased from municipal public water systems can be provided either before or after treatment (Nicot et al., 2014). Water for hydraulic fracturing is also sometimes purchased from smaller private entities, such as local land owners (Nicot et al., 2014).

4.3 Water Use Per Well

In this section, we provide an overview of the amount of water used per well during hydraulic fracturing. We discuss water use in the life cycle of oil and gas operations (Section 4.3.1) and national per well estimates and associated variability (Section 4.3.2). More detailed locality-specific information on water use per well is provided in Section 4.5.

4.3.1 Hydraulic Fracturing Water Use in the Life Cycle of Oil and Gas

Water is needed throughout the life cycle of oil and gas production and use, including both at the well for processes such as well pad preparation, drilling, and fracturing (i.e., the upstream portion), and later for end uses such as electricity generation, home heating, or transportation (i.e., the downstream portion) (Jiang et al., 2014; Laurenzi and Jersey, 2013). Most of the upstream water usage and consumption occurs during hydraulic fracturing (Jiang et al., 2014; Clark et al., 2013; Laurenzi and Jersey, 2013).[2] Water use per well estimates in this chapter focus on hydraulic fracturing in the upstream portion of the oil and gas life cycle, as the downstream portion of the lifecycle is outside the scope of this assessment.[3]

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[1] According to Section 1401(4) of the Safe Drinking Water Act, a public water system is defined as system that provides water for human consumption from surface water or groundwater through pipes or other infrastructure to at least 15 service connections, or an average of at least 25 people, for at least 60 days per year. Public water systems may either be publicly or privately owned.

[2] Laurenzi and Jersey (2013) reported that hydraulic fracturing accounted for 91% of upstream water consumption, based on industry data for 29 wells in the Marcellus Shale. (91% was calculated from their paper by dividing hydraulic fracturing fresh water consumption (13.7 gal (51.9 L)/Megawatt-hour (MWh)) by total upstream fresh water consumption (15.0 gal (56.8 L)/MWh) and multiplying by 100). Similarly, Jiang et al. (2014) reported that 86% of water consumption occurred at the fracturing stage for the Marcellus Shale, based on Pennsylvania Department of Environmental Protection (PA DEP) data on 500 wells. The remaining water was used in several upstream processes (e.g., well pad preparation, well drilling, road transportation to and from the wellhead, and well closure once production ended). Clark et al. (2013) estimated lower percentages (30%−80%) of water use at the fracturing stage for multiple formations. Although their estimates for the fraction of water used at the fracturing stage may be low due to their higher estimates for transportation and processing, the estimates by Clark et al. (2013) similarly illustrate the importance of the hydraulic fracturing stage in water use, particularly in terms of the upstream portion of the life cycle.

[3] When the full life cycle of oil and gas production and use is considered (i.e., both upstream and downstream water use), most water is used and consumed downstream. For example, in a life cycle analysis of hydraulically fractured gas used for electricity generation, Laurenzi and Jersey (2013) reported that only 6.7% of water consumption occurred upstream (15.0 gal (56.8 L)/MWh), while 93.3% of fresh water consumption occurred downstream for power plant cooling via

4.3.2 National Estimates and Variability in Water Use Per Well for Hydraulic Fracturing

At its most basic level, the volume of water used per well for hydraulic fracturing equals the concentration of water in the hydraulic fracturing fluid multiplied by the total volume of the fluid injected. In turn, the total volume of fluid injected generally equals the volume of fluid in the fractures, plus the volume of the well itself, plus any fluid lost due to "leakoff" or other unintended losses.[1]

Nationally, most operators employ fracturing fluids with water as a base fluid, meaning the concentration of water in the fluid is high (U.S. EPA, 2015b; Yang et al., 2013; GWPC and ALL Consulting, 2009). The EPA inferred that more than 93% of reported disclosures to FracFocus used water as a base fluid (U.S. EPA, 2015b). The median reported concentration of water in the hydraulic fracturing fluid was 88% by mass, with 10th and 90th percentiles of 77% and 95%, respectively. Only roughly 2% of disclosures (761 wells) reported the use of non-aqueous substances as base fluids, typically either liquid-gas mixtures of nitrogen or carbon dioxide. Both of these formulations still contained substantial amounts of water, as water made up roughly 60% (median value) of the fluid in them (U.S. EPA, 2015b). Other formulations were rarely reported. Fluid formulations are discussed further in Chapter 5.

On average, hydraulic fracturing requires more than a million gallons (3.8 million liters) of water per well. Jackson et al. (2015) reported a national average of 2.4 million gal (9.1 million L) of water per well, calculated from FracFocus disclosures between 2010 and 2013. According to the EPA's project database of disclosures to FracFocus 1.0 (hereafter the EPA FracFocus 1.0 project database), the median volume of water used per well was 1.5 million gal (5.7 million L) between 2011 and early 2013, based on 37,796 disclosures nationally (U.S. EPA, 2015b, c).[2] Data on reported Information Handling Services well numbers and median volumes in Gallegos et al. (2015) show that overall per well volumes have increased in recent years from approximately 1.5 million gal (5.7 million L) in 2011 to 2.7 million gal (10.2 million L) in 2014.[3]

The recent increase in water use per well has been driven primarily by the proportional increase in horizontal wells (Gallegos et al., 2015) (Figure 4-1). Increases in horizontal well length affect total volumes injected primarily by allowing a larger fracture volume to be stimulated (Economides et al., 2013). As horizontal wells get longer, fracture, well, and total volumes all increase. Importantly, increases in the well length and water use per well do not necessarily mean an increase in water intensity (the amount of water used per unit energy extracted). Goodwin et al. (2014) found water

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evaporation (209.0 gal (791.2 L)/MWh). Similar results were found for gas extraction in the Eagle Ford Shale (Scanlon et al., 2014b).

[1] Leakoff is the fraction of the hydraulic fracturing fluid that infiltrates into the formation (e.g., through an existing natural fissure) and is not recovered during production. This water lost to the formation can be a substantial fraction of the water injected (O'Malley et al., 2015). See Chapter 6 for more information about leakoff and some recent findings related to the relationship between hydraulic fracturing fluid volume and fracture volume.

[2] All water use data included in the EPA's FracFocus 1.0 project database were obtained from disclosures made to FracFocus. Although disclosures were made on a per well basis, a small proportion of the wells were associated with more than one disclosure (i.e., 876 out of 37,114, based on unique API numbers) (U.S. EPA, 2015c). For the purposes of this chapter, we discuss water use per disclosure in terms of water use per well.

[3] Derived from supporting information in Gallegos et al. (2015). Calculated by multiplying the median volume by the number of wells for each well type, then summing volumes across well types, and dividing by the total number of wells.

intensity did not increase in the Denver Basin despite increases in well length and water use per well.

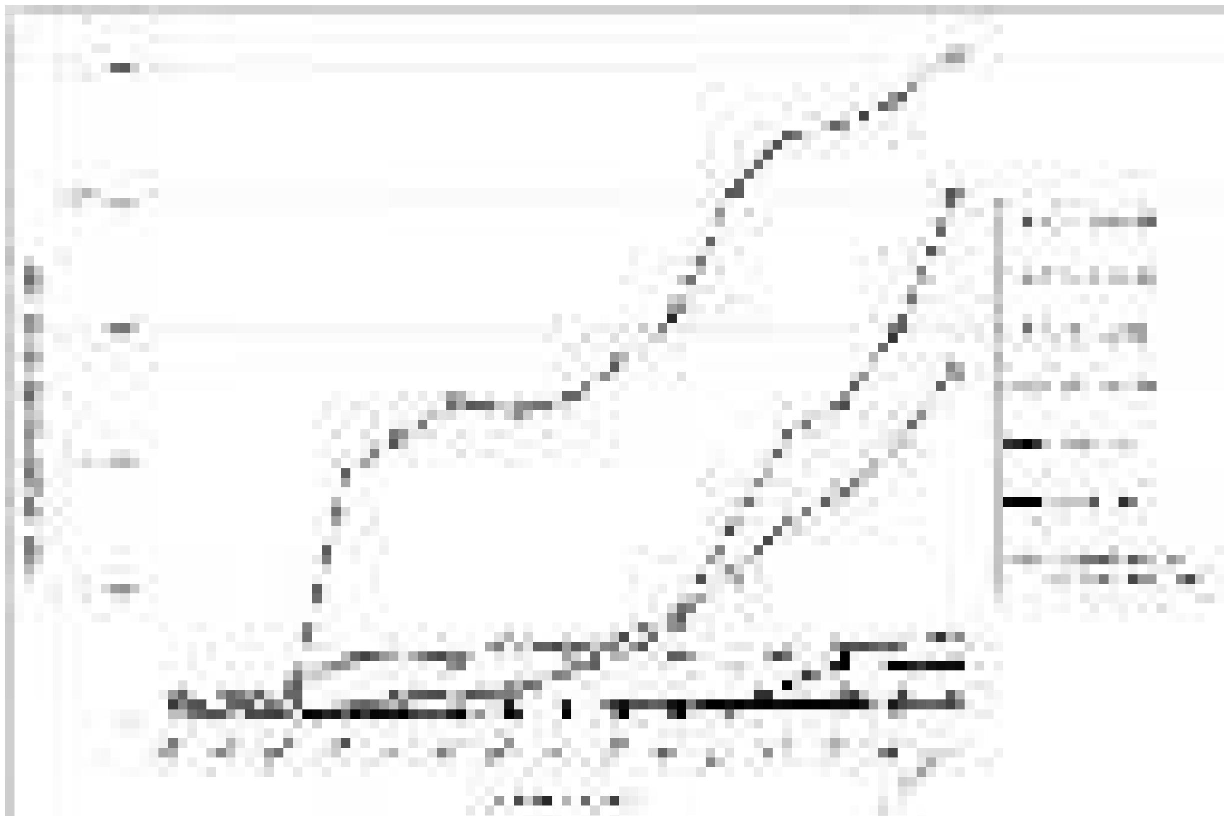

Figure 4-1. Median water volume per hydraulically fractured well nationally, expressed by well type and completion year.
Adapted using data from Gallegos et al. (2015). Note: shown in orange is the estimated median across all well types, derived from Gallegos et al. (2015) supporting information Tables S2 and S3. Calculated by multiplying the median volume by the number of wells for each well type, then summing volumes across well types, and dividing by the total number of wells for each year. This estimated median across all well types reflects the central tendency of the data, and was calculated because the individual data are proprietary and not published, preventing the calculation of an overall median.

There is substantial variation around these per well estimates. For instance, the 10th and 90th percentiles from the EPA FracFocus 1.0 project database are 74,000 gal and 6 million gal (280,000 L and 23 million L) per well, respectively.[1] Even in specific basins, plays, and within a single oil and gas field, water use per well varies widely. For example, Laurenzi and Jersey (2013) reported volumes ranging from 1 to 6 million gal (3.8 to 23 million L) per well (10th to 90th percentile) in the Wattenberg Field in Colorado.

Of the major unconventional formation types discussed in Chapter 2 (shales, tight formations-including tight sands or sandstones, and coalbeds), coalbeds generally require less water per well.

[1] Although the EPA FracFocus report shows 5th and 95th percentiles, we report 10th and 90th percentiles throughout this chapter to further reduce the influence of outliers.

Coalbed methane (CBM) comes from coal seams that often have a high initial water content and tend to occur at much shallower depths (U.S. EPA, 2015k). In part because of the shallower depths, shorter well lengths result in lower water use per well, often by an order of magnitude or more compared to operations in shales or tight formations (e.g., Murray, 2013).

4.4 Hydraulic Fracturing Water Use and Consumption at the National, State, and County Scale

In this section, we provide an overview of water use and consumption for hydraulic fracturing at the national, state, and county scale. We then compare these values to total water use and consumption at these scales. We do this to contextualize hydraulic fracturing water use and consumption with total water use and consumption, and to illustrate whether hydraulic fracturing is a relatively large or small user and consumer of water at these scales. Later, we compare hydraulic fracturing water use to water availability estimates at the county scale (Text Box 4-2).

Water use is water withdrawn for a specific purpose, part or all of which may be returned to the local hydrologic cycle. Water consumption is water that is removed from the local hydrologic cycle following its use (e.g., via evaporation, transpiration, incorporation into products or crops, consumption by humans or livestock), and is therefore unavailable to other water users (Maupin et al., 2014). Hydraulic fracturing water consumption can occur through evaporation from storage ponds, the retention of water in the subsurface through imbibition, or disposal in Class II wells, among other means.

Hydraulic fracturing water use is a function of the water use per well and the total number of wells fractured at a given spatial scale during the time period analyzed, calculated from the EPA FracFocus 1.0 project database (U.S. EPA, 2015c). Water consumption estimates are derived from United States Geological Survey (USGS) water use data, and therefore both use and consumption are presented with the published water use numbers being first.

4.4.1 National and State Scale

Hydraulic fracturing uses and consumes billions of gallons of water each year in the United States, but at the national and state scales, it is a relatively small user and consumer of water compared to total water use and consumption. According to the EPA's FracFocus 1.0 project database, hydraulic fracturing used 36 billion gal (136 billion L) of water in 2011 and 52 billion gal (197 billion L) in 2012, yielding an average annually of 44 billion gal (167 billion L) of water in 2011 and 2012 across all 20 states in the project database (U.S. EPA, 2015b, c). National water use for hydraulic fracturing can also be estimated by multiplying the water use per well by the number of wells hydraulically fractured. If the median water use per well (1.5 million gal) (5.7 million L) from the EPA's FracFocus 1.0 project database is multiplied by 25,000 to 30,000 wells fractured annually (Chapter 3), national water use for hydraulic fracturing is estimated to range from 38 to 45 billion gal (142 to 170 billion L) annually. Other calculated estimates have ranged higher than this, including estimates of approximately 80 billion gal (300 billion L) (Vengosh et al., 2014) and 50 to 72 billion gal (190-273 billion L) (U.S. EPA, 2015e). These estimates are higher due to differences in the estimated water use per well and the number of wells used as multipliers. For example, Vengosh et al. (2014) derived the estimate of approximately 80 billion gal (300 billion L) by multiplying an average of 4.0 million gal (15 million L) per well (estimated for shale gas wells) by 20,000 wells (the approximate total number of fractured wells in 2012).[1]

All of these estimates of water use for hydraulic fracturing are small relative to total water use and consumption at the national scale. The USGS compiles national water use estimates every five years in the National Water Census, with the most recent census conducted in 2010 (Maupin et al., 2014).[2] The USGS publishes water use, not consumption estimates, yet by applying consumption factors for each use category in the 2010 National Water Census, we derived estimates of total water consumption. We also used a consumption factor to estimate hydraulic fracturing water consumption from values in the EPA FracFocus 1.0 project database.[3] Comparing these estimates, average annual hydraulic fracturing water use in 2011 and 2012 was less than 1% of total 2010 annual water use for all of the 20 states combined where operators reported water use to FracFocus in 2011 and 2012. Hydraulic fracturing water consumption followed the same pattern when compared to total water consumption (Appendix Table B-1).[4]

At the state scale, hydraulic fracturing also generally uses billions of gallons of water, but accounts for a low percentage of total water use or consumption. Of all states in the EPA FracFocus 1.0 project database, operators in Texas used the most water (47% of water use reported in the EPA FracFocus 1.0 project database) (U.S. EPA, 2015c) (Appendix Table B-1). This was due to the large number of wells in that state, since hydraulic fracturing water use is proportional to the number of wells. Over 94% of reported water use occurred in just seven of the 20 states in the EPA FracFocus 1.0 project database (listed in order of highest statewide hydraulic fracturing water use): Texas, Pennsylvania, Arkansas, Colorado, Oklahoma, Louisiana, and North Dakota (U.S. EPA, 2015c) (Appendix Table B-1). Hydraulic fracturing is a small percentage when compared to total water use (<1%) and consumption (<3%) in each individual state (Appendix Table B-1). Other studies have shown similar results, with hydraulic fracturing water use and consumption ranging from less than

[1] This could result in an overestimation because the estimate of 20,000 wells was derived in part from FracFocus, and these wells are not necessarily specific to shale gas; they may include other types of wells that use less water (e.g., CBM). The estimate of 1.5 million gal (5.7 million L) per well based on the U.S. EPA (2015c) FracFocus 1.0 project database likely leads to a more robust estimate when used to calculate national water use for hydraulic fracturing because it includes wells from multiple formation types (i.e., shale, tight sand, and CBM), some of which use less water than shale gas wells on average.

[2] The National Water Census includes uses such as public supply, irrigation, livestock, aquaculture, thermoelectric power, industrial, and mining at the national, state, and county scale. The 2010 National Water Census included hydraulic fracturing water use in the mining category; there was no designated category for hydraulic fracturing alone.

[3] See footnotes for Appendix Table B-1 or for Table 4-3 for a description of the consumption estimate calculations.

[4] Water use percentages were calculated by averaging annual water use for hydraulic fracturing in 2011 and 2012 for a given state or county (U.S. EPA, 2015c), and then dividing by 2010 USGS total water use (Maupin et al., 2014) and multiplying by 100. Note, the annual hydraulic fracturing water use reported in FracFocus was not added to the 2010 total USGS water use value in the denominator, and is simply expressed as a percentage compared to 2010 total water use or consumption. This was done because of the difference in years between the two datasets, and because the USGS 2010 Water Census (Maupin et al., 2014) included hydraulic fracturing water use estimates in their mining category. This approach is consistent with that of other literature on this topic; see Nicot and Scanlon (2012). Consumption estimates were calculated in the same manner, except consumption, not use, values were employed. County level data from the USGS 2010 Water Census are available online at http://water.usgs.gov/watuse/data/2010/ (accessed November 11, 2014).

1% of total use in West Virginia (West Virginia DEP, 2013), Colorado (Colorado Division of Water Resources et al., 2014), and Texas (Nicot et al., 2014; Nicot and Scanlon, 2012), to approximately 4% in North Dakota (North Dakota State Water Commission, 2014).

4.4.2 County Scale

Water use and consumption for hydraulic fracturing is also relatively small in most, but not all, counties in the United States (Table 4-3; Figure 4-2; Figure 4-3a,b; and Appendix Table B-2). Based on the EPA FracFocus 1.0 project database, reported fracturing water use in 2011 and 2012 was less than 1% compared to 2010 USGS total water use in 299 of the 401 reporting counties (Figure 4-3a; Appendix Table B-2). However, hydraulic fracturing water use was 10% or more compared to total water use in 26 counties, 30% or more in nine counties, and 50% or more in four counties (Table 4-3; Figure 4-3a). McMullen County in Texas had the highest percentage at over 100% compared to 2010 total water use.[1] Total consumption estimates followed the same pattern, but with more counties in the higher percentage categories (hydraulic fracturing water consumption was 10% or more compared to total water consumption in 53 counties; 30% or more in 25 counties; 50% or more in 16 counties; and over 100% in four counties) (Table 4-3; Figure 4-3b).

Estimates based on the EPA's FracFocus 1.0 project database may form an incomplete picture of hydraulic fracturing water use in a given state or county, because the majority of states with data in the project database did not require disclosure to FracFocus during the time period analyzed (U.S. EPA, 2015b). We conclude that this likely does not substantially alter the overall patterns observed in Figure 4-3a,b. See Text Box 4-1 for further details. These percentages also depend both upon the absolute water use and consumption for hydraulic fracturing and the relative magnitude of other water uses and consumption in that state or county. For instance, a rural county with a small population might have relatively low total water use prior to hydraulic fracturing.[2] Also, just because water is used in a certain county does not necessarily mean it originated in that county. The cost of trucking water can be substantial (Slutz et al., 2012), and the industry tends to acquire water from nearby sources when possible (Section 4.2.1); however, water can also be piped in from more distant, regional supplies. Despite these caveats, it is clear that hydraulic fracturing is generally a relatively small user (and consumer) of water at the county level, with the exception of a small number of counties where water use and consumption for fracturing can be high relative to other uses and consumption.

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[1] Estimates of use or consumption exceeded 100% when hydraulic fracturing water use averaged for 2011 and 2012 exceeded total water use or consumption in that county in 2010.

[2] For example, McMullen County, Texas, mentioned above contains a small number of residents (707 people in 2010, according to the U.S. Census Bureau (2014).

Table 4-3. Average annual hydraulic fracturing water use and consumption in 2011 and 2012 compared to total annual water use and consumption in 2010, by county.

Only counties where hydraulic fracturing water was 10% or greater compared to 2010 total water use are shown (for full table, see Appendix Table B-2). Average annual hydraulic fracturing water use data in 2011 and 2012 from the EPA's FracFocus 1.0 project database (U.S. EPA, 2015c). Total annual water use data in 2010 from the USGS (Maupin et al., 2014). States listed by order of appearance in the chapter.

State	County	Total annual water use in 2010 (millions of gal)[a]	Average annual hydraulic fracturing water use in 2011 and 2012 (millions of gal)[b]	Hydraulic fracturing water use compared to total water use (%)[c]	Hydraulic fracturing water consumption compared to total water consumption (%)[c,d]
Texas	McMullen	657.0	745.9	113.5	350.4
	Karnes	1861.5	1055.2	56.7	120.1
	La Salle	2474.7	1288.7	52.1	93.7
	Dimmit	4073.4	1794.2	44.0	81.3
	Irion	1335.9	411.4	30.8	74.5
	Montague	3989.5	925.3	23.2	77.8
	De Witt	2394.4	546.6	22.8	48.6
	Loving	781.1	138.4	17.7	94.1
	San Augustine	1131.5	182.1	16.1	50.8
	Live Oak	1916.3	294.0	15.3	40.1
	Wheeler	6522.6	858.0	13.2	21.5
	Cooke	4533.3	454.3	10.0	29.9
Pennsylvania	Susquehanna	1617.0	751.3	46.5	123.4
	Sullivan	222.7	66.5	29.9	79.8
	Bradford	4354.5	1059.4	24.3	78.2
	Tioga	2909.1	566.3	19.5	47.3
	Lycoming	5854.6	704.6	12.0	33.8
West Virginia	Doddridge	405.2	78.5	19.4	69.4
Ohio	Carroll	1127.9	152.7	13.5	37.3
North Dakota	Mountrail	1248.3	449.4	36.0	98.3
	Dunn	1076.8	309.5	28.7	43.1
	Burke	394.2	63.6	16.1	40.8
	Divide	806.7	102.2	12.7	18.6

State	County	Total annual water use in 2010 (millions of gal)[a]	Average annual hydraulic fracturing water use in 2011 and 2012 (millions of gal)[b]	Hydraulic fracturing water use compared to total water use (%)[c]	Hydraulic fracturing water consumption compared to total water consumption (%)[c,d]
Arkansas	Van Buren	1587.8	899.6	56.7	168.8
Louisiana	Red River	1606.0	569.6	35.5	83.2
	Sabine	1522.1	395.2	26.0	76.6

[a] County level data accessed from the USGS website (http://water.usgs.gov/watuse/data/2010/) on November 11, 2014. Total water withdrawals per day were multiplied by 365 days to estimate total water use for the year (Maupin et al., 2014).

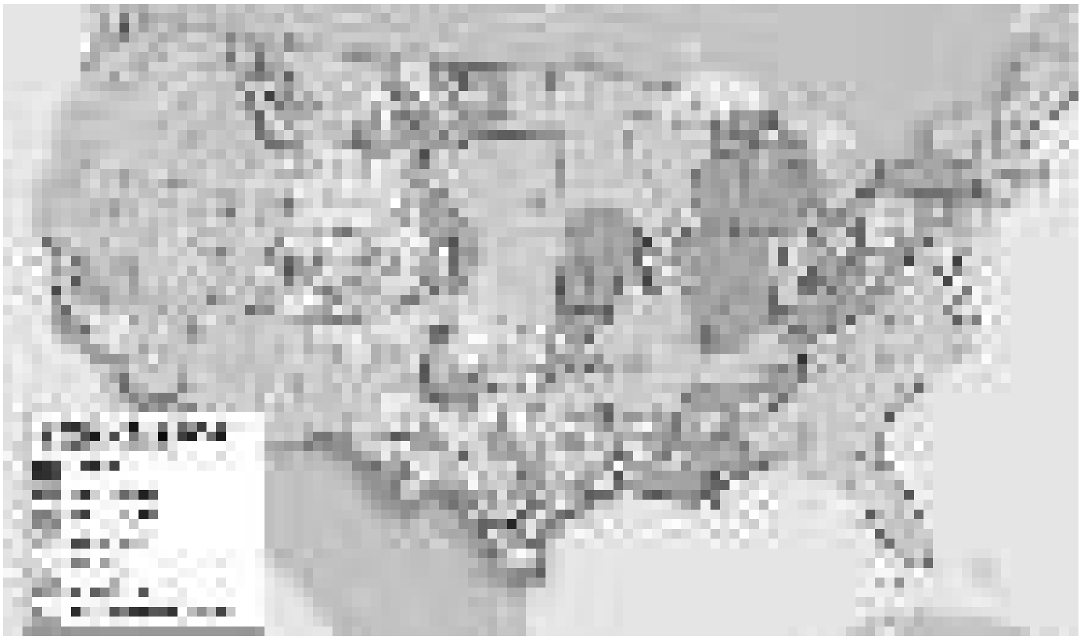
[b] Average of water used for hydraulic fracturing in 2011 and 2012 calculated from the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

[c] Percentages were calculated by averaging annual water use for hydraulic fracturing reported in FracFocus in 2011 and 2012 for a given state or county (U.S. EPA, 2015c), and then dividing by 2010 USGS total water use (Maupin et al., 2014) and multiplying by 100.

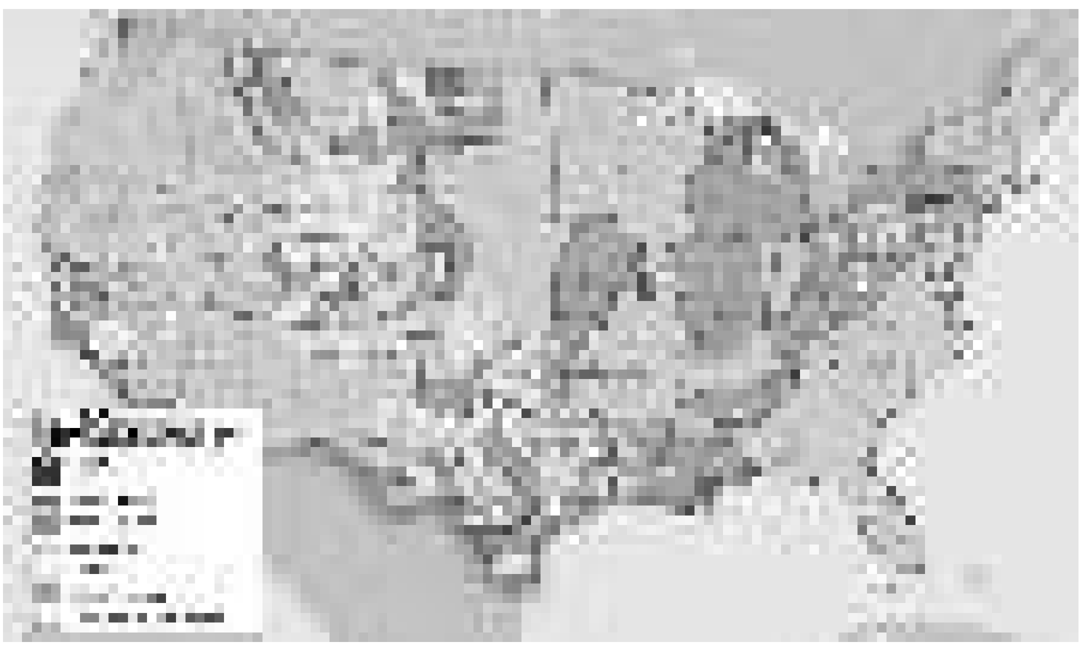
[d] Consumption values were calculated with use-specific consumption rates predominantly from the USGS, including 19.2% for public supply, 19.2% for domestic use, 60.7% for irrigation, 60.7% for livestock, 14.8% for industrial uses, 14.8% for mining (Solley et al., 1998), and 2.7% for thermoelectric power (Diehl and Harris, 2014). We used rates of 71.6% for aquaculture from Verdegem and Bosma (2009) ((evaporation per kg fish + infiltration per kg)/total water use per kg); and 82.5% for hydraulic fracturing (consumption value calculated by taking the median value for all reported produced water/injected water percentages in Tables 7-1 and 7-2 of this assessment and then subtracting from 100%). If a range of values was given, the midpoint was used. Note, this aspect of consumption is likely a low estimate since much of this produced water (injected water returning to the surface) is not subsequently treated and reused, but rather disposed of in Class II wells – see Chapter 8.

Figure 4-2. Average annual hydraulic fracturing water use in 2011 and 2012 by county.

Source: U.S. EPA (2015c). Water use in millions of gallons (Mgal). Counties shown with respect to major U.S. Energy Information Administration (EIA) shale basins (EIA, 2015). Orange borders identify states that required some degree of reporting to FracFocus in 2011 and 2012.



(a)



(b)

Figure 4-3. (a) Average annual hydraulic fracturing water use in 2011 and 2012 compared to total annual water use in 2010, by county, expressed as a percentage; (b) Average annual hydraulic fracturing water consumption in 2011 and 2012 compared to total annual water consumption in 2010, by county, expressed as a percentage.

Average annual hydraulic fracturing water use data in 2011 and 2012 from the EPA's FracFocus 1.0 project database (U.S. EPA, 2015c). Total annual water use data in 2010 from the USGS (Maupin et al., 2014). See Table 4-3 for descriptions of calculations for estimating consumption. Counties shown with respect to major U.S. EIA shale basins (EIA, 2015). Orange borders identify states that required some degree of reporting to FracFocus in 2011 and 2012. Note: Values over 100% denote counties where the average annual hydraulic fracturing water use or consumption in 2011 and 2012 exceeded the total annual water use or consumption in that county in 2010.

Text Box 4-1. Using the EPA's FracFocus 1.0 Project Database to Estimate Water Use for Hydraulic Fracturing.

The FracFocus Chemical Disclosure Registry (often referred to as FracFocus; www.fracfocus.org) is a national hydraulic fracturing chemical disclosure registry managed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission. FracFocus was created to provide the public access to reported chemicals used for hydraulic fracturing within their area. It was originally established in 2011 (version 1.0) for voluntary reporting by oil and gas well operators. The EPA used the data available from FracFocus between January 1, 2011 and February 28, 2013 to develop the EPA FracFocus 1.0 project database; the database and a related EPA report were both peer reviewed and published (U.S. EPA, 2015b, c). Six of the 20 states discussed in this assessment required disclosure to FracFocus at various points during this time; another three of the 20 states offered the choice of reporting to FracFocus or the state during this same time period (U.S. EPA, 2015b). Estimates based on the EPA's FracFocus 1.0 project database could form an incomplete picture of hydraulic fracturing water use, because most states with data in the project database (14 out of 20) did not require disclosure to FracFocus during the time period analyzed (U.S. EPA, 2015b).

Water use for fracturing is a function of the water use per well and the total number of wells fractured over a given spatial area or a given period of time. For water use per well, we found seven literature values for comparison with values from the EPA's FracFocus 1.0 project database. On average, water use estimates per well in the project database were 77% of literature values (the median was 86%); Colorado's Denver Basin was the only location where the project database estimate as a percentage of the literature estimate was low (14%) (Appendix Table B-3). In general, water use per well estimates from the EPA's FracFocus 1.0 project database appear to align closely with the literature estimates for most areas for which we have data, with the exception of the Denver Basin of Colorado.

For the number of wells, we compared data in the EPA's FracFocus 1.0 project database to numbers available in state databases from North Dakota, Pennsylvania, and West Virginia (Appendix Table B-4). These were the state databases from which we could distinguish hydraulically fractured wells from other oil and gas wells. On average, we found that the EPA FracFocus 1.0 project database included 67% of the wells listed in state databases for 2011 and 2012 (Appendix Table B-4). Unlike North Dakota and Pennsylvania, West Virginia did not require operators to report fractured wells to FracFocus during this time period, possibly explaining its lower reporting rate. Multiplying the average EPA FracFocus 1.0 project database values of 77% for water use per well and 67% for well counts yields 52%. Thus, the EPA FracFocus 1.0 project database estimates for water use could be slightly over half of the estimates from these three state databases during this time period. These values are based on small sample sizes (seven literature values and three state databases) and should be interpreted with caution. Nevertheless, these numbers suggest that estimates based on the EPA's FracFocus 1.0 project database likely form an incomplete picture of hydraulic fracturing water use during this time period.

To assess how this might affect hydraulic fracturing water use estimates in this chapter, we doubled the water use value in the EPA's FracFocus 1.0 project database for each county, an adjustment much higher than any likely underestimation. Even with this adjustment, fracturing water use was still less than 1% compared to 2010 total water use in the majority of the 401 U.S. counties represented in the EPA FracFocus 1.0 project database (299 counties without adjustment versus 280 counties with adjustment). The number of counties where hydraulic fracturing water use was 30% or more of 2010 total county water use increased from nine to 21 with the adjustment.

These results indicate that most counties have relatively low hydraulic fracturing water use relative to total water use, even when accounting for likely underestimates. Since consumption estimates are derived from use, these will also follow the same pattern. Thus, potential underestimates based on the EPA's FracFocus 1.0 project database likely do not substantially alter the overall pattern shown in Figure 4-3. Rather, underestimates of hydraulic fracturing water use would mostly affect the percentages in the small number of counties where fracturing already constitutes a higher percentage of total water use and consumption.

4.5 Potential for Impacts by Location

The potential for hydraulic fracturing water acquisition to impact drinking water availability or alter its quality depends on the balance between water withdrawals and water availability at a given location. Where water availability is high compared to the volume of water withdrawn for hydraulic fracturing, this water use can be accomodated. However, where water availability is low and hydraulic fracturing water use is high, these withdrawals are more likely to impact drinking water resources. The balance between withdrawals and availability can vary greatly by geographic location. Moreover, a combination of regional or site-specific factors can alter this balance, making impacts more or less likely, or more or less severe. For these reasons, we discuss the various factors and potential for impacts by geographic location in the following section.

We organize this discussion by state, addressing 15 states accounting for almost all disclosures reported in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c): Texas (Section 4.5.1); Colorado and Wyoming (Section 4.5.2); Pennsylvania, West Virginia, and Ohio (Section 4.5.3); North Dakota and Montana (Section 4.5.4); Arkansas and Louisiana (Section 4.5.5), Oklahoma and Kansas (Appendix B.2.1); and Utah, New Mexico, and California (Appendix B.2.2). We highlight the states that best illustrate concepts relating to the potential for impacts, or factors that affect the frequency or severity of these impacts in Section 4.5; the remaining states are discussed in Appendix B.2. Within Section 4.5 and Appendix B, we address each state in order of most hydraulically fractured wells to least, and combine states with similar geographies or activity. For certain states, we address major oil and gas regions separately (e.g., the Permian Basin in Texas). Each section describes the number of fractured wells in that state or region, the type of water used, water use per well, and water use estimates at the county scale. We then discuss the potential for impacts by comparing water use and water availability and addressing factors (e.g., drought or the amount of water reused to offset fresh water use) that might alter the frequency or severity of impacts. As noted in the chapter introduction, we use several lines of evidence to evaluate the potential for impacts and factors for each location. We use the scientific literature, county level assessments, and local case studies where available.

4.5.1 Texas

Hydraulic fracturing in Texas accounts for the bulk of the activity reported nationwide, comprising 48% of the disclosures in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c) (Figure 4-4; Appendix Table B-5). There are five major basins in Texas: the Permian, Western Gulf (includes the Eagle Ford play), Fort Worth (includes the Barnett play), TX-LA-MS Salt (includes the Haynesville play), and the Anadarko (Figure 4-5); together, these five basins contain 99% of Texas' reported wells (Appendix Table B-5).

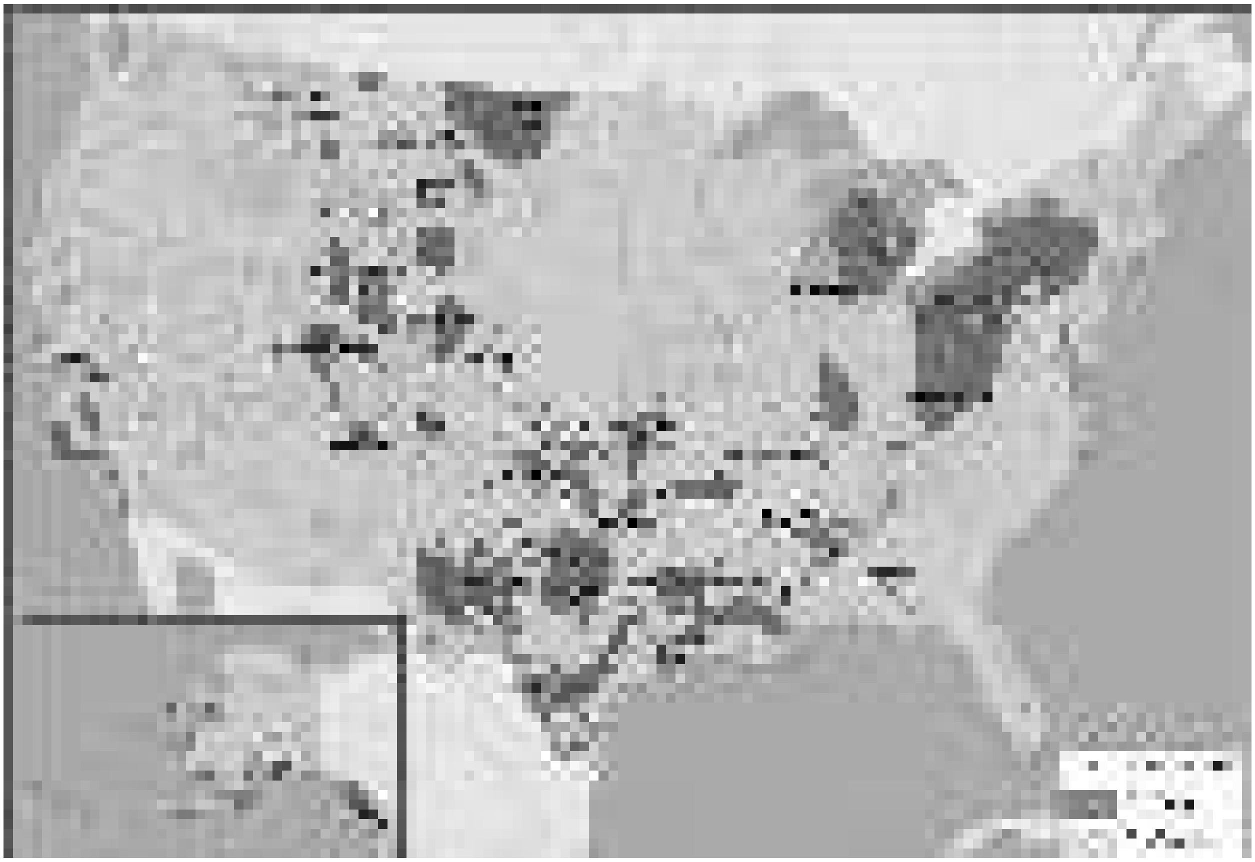


Figure 4-4. Locations of wells in the EPA FracFocus 1.0 project database, with respect to U.S. EIA shale plays and basins.
Note: Hydraulic fracturing can be conducted in geologic settings other than shale; therefore, some wells on this map are not associated with any EIA shale play or basin (EIA, 2015; U.S. EPA, 2015c).

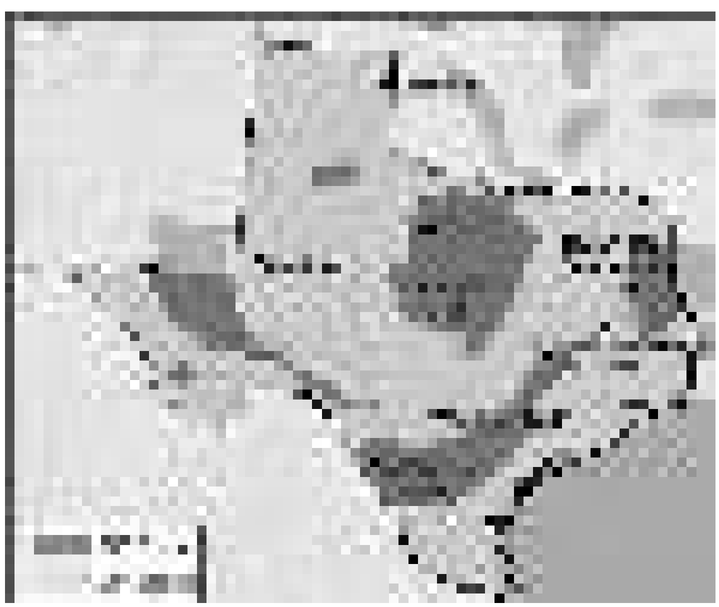


Figure 4-5. Major U.S. EIA shale plays and basins for Texas.
Source: EIA (2015).

Types of water used: What is known about water sources in Texas largely comes from direct surveys and interviews with industry operators and water suppliers (Nicot et al., 2014; Nicot et al., 2012). Overall, groundwater is the dominant source throughout most of the state (Nicot et al., 2014; Nicot et al., 2012) (Table 4-1). The exception is the Barnett Shale, where both surface water and groundwater are used in approximately equal proportions.

Hydraulic fracturing in Texas uses mostly fresh water (Nicot et al., 2012).[1] The exception is the far western portion of the Permian Basin, where brackish water makes up an estimated 80% of total hydraulic fracturing water use. Brackish water is used to a lesser extent in the Anadarko Basin, the Midland portion of the Permian Basin, and the Eagle Ford Shale (Table 4-4). Reuse of wastewater as a percentage of total water use is generally low (5% or less) in all major basins and plays in Texas, except for the Anadarko Basin in the Texas Panhandle, where it is 20% (Nicot et al., 2012) (Table 4-2).

Table 4-4. Estimated brackish water use as a percentage of total hydraulic fracturing water use in the main hydraulic fracturing areas of Texas, 2011.[a]
Adapted from Nicot et al. (2012).[b]

Play	Percentage
Barnett Shale	3%
Eagle Ford Shale	20%
Texas portion of the TX-LA-MS Salt Basin[c]	0%
Permian Basin—Far West	80%
Permian Basin—Midland	30%
Anadarko Basin	30%

[a] Nicot et al. (2012) define brackish water as any water with a total dissolved solids (TDS) content of >1,000 mg/L, but <35,000 mg/L, although they often limit that range to between 1,000 and 10,000 mg/L.

[b] Nicot et al. (2012) present the estimated percentages of brackish, recycled/reused, and fresh water relative to total hydraulic fracturing water use so that the percentages of the three categories sum to 100%.

[c] Nicot et al. (2012) refer to this region of Texas as the East Texas Basin.

The majority of water used in Texas for hydraulic fracturing is self-supplied via direct ground or surface water withdrawals (Nicot et al., 2014). Less often, water is purchased from local landowners, municipalities, larger water districts, or river authorities (Nicot et al., 2014).

Water use per well: Water use per well varies across Texas basins, with reported medians from 2011 to early 2013 of 3.9 million gal (14.8 million L) in the Fort Worth Basin, 3.8 million gal (14.4 million L) in the Western Gulf, 3.3 million gal (12.5 million L) in the Anadarko, 3.1 million gal (11.7 million L) in the TX-LA-MS Salt, and 840,000 gal (3.2 million L) in the Permian (Appendix

[1] The EPA FracFocus report shows that “fresh” was the only source of water listed in 91% of all disclosures reporting a source of water in Texas (U.S. EPA, 2015b). Nineteen percent of Texas disclosures included information related to water sources (U.S. EPA, 2015b).

Table B-5). Relatively low water use in the Permian Basin, which contains roughly half the reported wells in the state, is due to the abundance of vertical wells, mostly for oil extraction (Nicot et al., 2012).

Water use per well is increasing in most locations in Texas. In the Barnett Shale, water use per well increased from approximately 3 million gal (11 million L) in the mid-2000's to approximately 5 million gal (19 million L) in 2011 as the horizontal lengths of wells increased (Nicot et al., 2014). Similar increases in lateral length and water use per well were reported for the Texas-Haynesville, East Texas, and Anadarko basins, and most of the Permian Basin (Nicot et al., 2012; Nicot and Scanlon, 2012).[1]

Water use/consumption at the county scale: Water use and consumption for hydraulic fracturing can be significant in some Texas counties. Texas contains five of nine counties nationwide where operators used more than 1 billion gal (3.8 billion L) of water annually for hydraulic fracturing, and five of nine counties where fracturing water use in 2011 and 2012 was 30% or more compared to total water use in those counties in 2010 (Table 4-3, Figure 4-3a; Appendix Table B-2).[2]

According to detailed county level projections, water use for hydraulic fracturing is expected to increase with oil and gas production in the coming decades, peaking around the year 2030 (Nicot et al., 2012). These projections were made before the recent decline in oil and gas prices, and so are highly uncertain. If these projections hold, the majority of counties are expected to have relatively low water use for fracturing in the future, but hydraulic fracturing water use could equal or exceed 10%, 30%, and 50% compared to 2010 total county water use in 30, nine, and three counties, respectively, by 2030 (Appendix Table B-7).

Potential for impacts: Of all locations surveyed in this chapter, the potential for water quantity and quality impacts due to hydraulic fracturing water acquisition appears to be highest in southern and western Texas. This area includes the Anadarko, the Western Gulf (Eagle Ford play), and the Permian Basins. According to Ceres (2014), 28% and 87% of the wells fractured in the Eagle Ford play and Permian Basin, respectively, are in areas of high to extremely high water stress.[3] A comparison of hydraulic fracturing water use to water availability at the county scale also suggests the potential for impacts in this region (Text Box 4-2).

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[1] It should be noted that energy production also increases with lateral lengths, and therefore, water use per unit energy produced—typically referred to as water intensity—may remain the same or decline despite increases in per-well water use (Nicot et al., 2014; Laurenzi and Jersey, 2013).

[2] Texas also contains 10 of the 25 counties nationwide where hydraulic fracturing water consumption was greater than or equal to 30% of 2010 total water consumption (Table 4-3). Nicot and Scanlon (2012) found similar variation among counties when they compared hydraulic fracturing water consumption to total county water consumption for the Barnett play. Their consumption estimates ranged from 581 million gal (2.20 billion L) in Parker County to 2.7 billion gal (10.2 billion L) in Johnson County, representing 10.5% and 29.7% compared to total water consumption in those counties, respectively. Fracturing in Tarrant County, part of the Dallas Fort-Worth area, consumed 1.6 billion gal (6.1 billion L) of water, 1.4% compared to total county water consumption (Nicot and Scanlon, 2012).

[3] Ceres (2014) compared well locations to areas categorized by a water stress index, characterized as follows: extremely high (defined as annual withdrawals accounting for greater than 80% of surface flows); high (40–80% of surface flows); or medium-to-high (20–40% of surface flows).

Text Box 4-2. Hydraulic Fracturing Water Use as a Percentage of Water Availability Estimates.

Researchers at Sandia National Laboratories assessed county level water availability across the continental United States (Tidwell et al., 2013). Assessments of water availability in the United States are generally lacking at the county scale, and this analysis—although undertaken for siting new thermoelectric power plants—can be used to assess potential impacts of hydraulic fracturing water withdrawals.

The authors generated annual water availability estimates for five categories of water: unappropriated surface water, unappropriated groundwater, appropriated water potentially available for purchase, brackish groundwater, and wastewater from municipal treatment plants (Tidwell et al., 2013). In the western United States, water is generally allocated by the principle of prior appropriation—that is, first in time of use is first in right. New development must use unappropriated water or purchase appropriated water from vested users. In their analysis, the authors assumed 5% of appropriated irrigated water could be purchased; they also excluded wastewater required to be returned to streams and the wastewater fraction already reused.

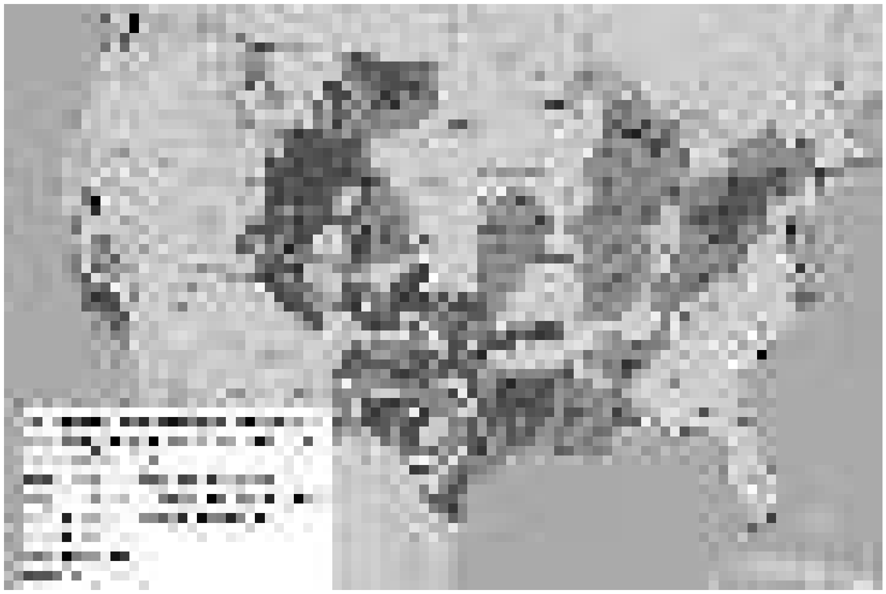
Given regulatory restrictions, they considered no fresh water to be available in California for new thermoelectric plants. Their definition of brackish water ranged from 3,000 to 10,000 ppm TDS, and from 50 to 2,500 ft (15-760 m) below the surface.

Combining their estimates of unappropriated surface water and groundwater and appropriated water potentially available for purchase, we derived a fresh water availability estimate for each county (except for those in California) and then compared this value to reported water use for hydraulic fracturing in 2011 and 2012 (U.S. EPA, 2015c). We also added the estimates of brackish groundwater and wastewater from municipal treatment plants to fresh water estimates to derive estimates of total water availability and did a similar comparison. Since the water availability estimates already take into account current water use for oil and gas operations, these results should be used only as indicator of areas where shortages might arise in the future. Here we focus on hydraulic fracturing water use compared to water availability. If we compared hydraulic fracturing water consumption to water availability, consumption would be lower relative to availability since by definition, water consumption is less than water use. Hence, water use versus availability acts as an upper-bound estimate, and includes consumption.

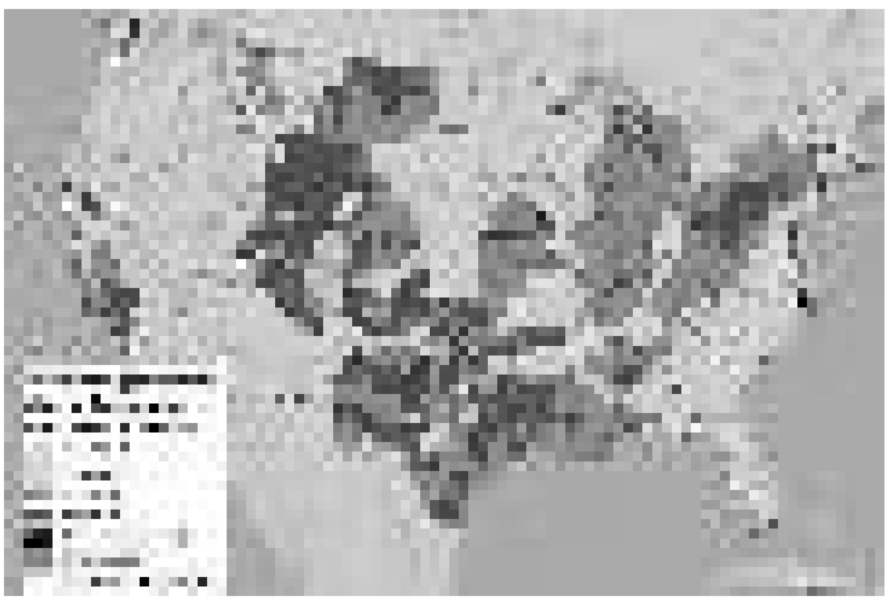
Overall, hydraulic fracturing water use represented less than 1% of fresh water availability in over 300 of the 395 counties analyzed (Figure 4-6a). This result suggests that there is ample water available at the county scale to accommodate hydraulic fracturing in most locations. However, there was a small number of counties where hydraulic fracturing water use was a relatively high percentage of fresh water availability. In 17 counties, fracturing water use actually exceeded the index of fresh water available; all of these counties were located in the state of Texas and were associated with the Anadarko, Barnett, Eagle Ford, and Permian basins/plays (Figure 4-5). In Texas counties with relatively high brackish water availability, hydraulic fracturing water use represented a much smaller percentage of total water availability (fresh + brackish + wastewater) (Figure 4-6b). This finding illustrates that potential impacts can be avoided or reduced in these counties through the use of brackish water or wastewater for hydraulic fracturing; a case study in the Eagle Ford play in southwestern Texas echoes this finding (Text Box 4-3).

(Text Box 4-2 is continued on the following page.)

Text Box 4-2 (continued). Hydraulic Fracturing Water Use as a Percentage of Water Availability Estimates.



(a)

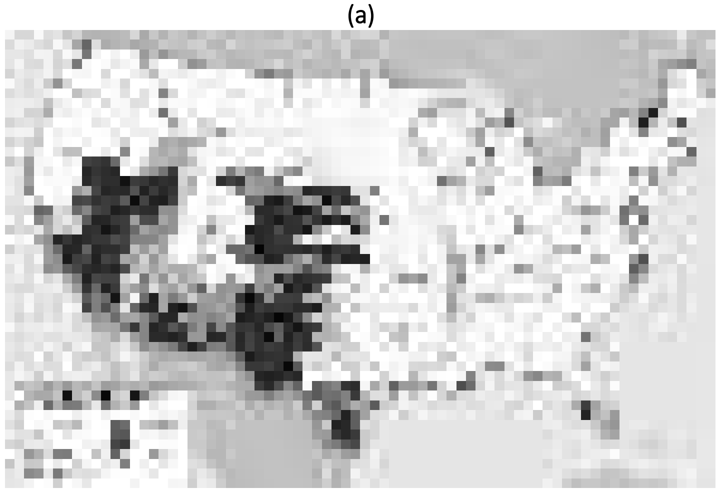


(b)

Figure 4-6. Average annual hydraulic fracturing water use in 2011 and 2012 compared to (a) fresh water available and (b) total water (fresh, brackish, and wastewater) available, by county, expressed as a percentage. Counties shown with respect to major U.S. EIA shale basins (EIA, 2015). Orange borders identify states that required some degree of reporting to FracFocus in 2011 and 2012. Data from U.S. EPA (2015c) and Tidwell et al. (2013); data from Tidwell et al. (2013) supplied from the U.S. Department of Energy (DOE) National Renewable Energy Laboratory on January 28, 2014 and available upon request from the U.S. DOE Sandia National Laboratories. The analysis by Tidwell et al. (2013) was done originally for thermoelectric power generation. As such, it was assumed that no fresh water could be used in California for this purpose due to regulatory restrictions, and therefore no fresh water availability data were given for California. The total water available for California is the sum of brackish water plus wastewater only.

Surface water availability is generally low in southern and western Texas (Figure 4-7a), and both fracturing operations and residents rely heavily on groundwater (Figure 4-7b). Similar to trends nationally, groundwater aquifers in Texas have experienced substantial declines caused by withdrawals (Konikow, 2013; TWDB, 2012; George et al., 2011). Groundwater in the Pecos Valley, Gulf Coast, and Ogallala aquifers in southern and western Texas is estimated to have declined by roughly 5, 11, and 44 mi^3 (21, 45.5, and 182 km^3), respectively, between 1900 and 2008 (Konikow, 2013).[1]

(a)



(b)

Figure 4-7. (a) Estimated annual surface water runoff from the USGS; (b) Reliance on groundwater as indicated by the ratio of groundwater pumping to stream flow and pumping.
Estimates for Figure 4-7a were calculated at the 8-digit hydrological unit code (HUC) scale by dividing annual average daily stream flow (from October 1, 2012, to September 30, 2013) by HUC area. Data accessed from the USGS (USGS, 2014c). Higher ratios (darker blues) in Figure 4-7b indicate greater reliance on groundwater. Figure adapted from Tidwell et al. (2012), using data provided by the U.S. Department of Energy's Sandia National Laboratories on December 12, 2014.

[1] The estimate of total net volumetric groundwater depletion for the Gulf Coast aquifer is the sum of the individual depletion estimates for the north (Houston area), central, and southern (Winter Garden area) parts of the Texas Gulf Coast aquifer. Groundwater depletion from the Carrizo-Wilcox aquifer is included in the estimate for the southern portion of the Gulf Coast aquifer (Konikow, 2013).

Groundwater quality degradation associated with aquifer pumping and the cumulative effects of all water users is well documented in the southern portion of the Ogallala aquifer. The quality of groundwater used by many private, public supply, and irrigation wells is poorest in the aquifer's southern portion, with elevated concentrations of TDS, chloride, nitrate, fluoride, manganese, arsenic, and uranium (Chaudhuri and Ale, 2014a; Gurdak et al., 2009; McMahon et al., 2007).[1] Extensive groundwater pumping can alter the quality of drinking water resources by inducing vertical mixing of high-quality groundwater with recharge water from the land surface that has been contaminated by nitrate or pesticides, or with lower-quality groundwater from underlying geologic formations (Gurdak et al., 2009; Konikow and Kendy, 2005). Pumping can also promote changes in reduction-oxidation (redox) conditions and thereby mobilize chemicals from geologic sources (e.g., uranium) (DeSimone et al., 2014). Similar patterns of groundwater quality degradation associated with prolonged aquifer depletion (i.e., salinization and contamination) have also been observed in other Texas aquifers, notably the northwest Edwards-Trinity (plateau), Pecos Valley, Carrizo-Wilcox, and southern Gulf Coast aquifers.[2]

The Texas Water Development Board (TWDB) estimates that overall demand for water (including water for hydraulic fracturing) out to the year 2060 will outstrip supply in southern and western Texas (TWDB, 2012). Furthermore, the TWDB expects groundwater supply in the major aquifers to decline by 30% between 2010 and 2060, mostly due to declines in the Ogallala aquifer (TWDB, 2012).[3,4] Irrigated agriculture is by far the dominant user of water from the Ogallala aquifer (Gurdak et al., 2009), but fracturing operations, along with other uses, now contribute to the aquifer's depletion.

The state has also experienced moderate to extreme drought conditions for much of the last decade, and the second-worst and longest drought in Texas history between March 2010 and November 2014 (TWDB, 2016; National Drought Mitigation Center, 2015) (Figure 4-8). Sustained drought conditions compound water availability concerns, and climate change is expected to place further stress on groundwater both now and in the future (Aghakouchak et al., 2014; Melillo et al., 2014) (Chapter 2). In their evaluation of the potential impact of climate change on groundwater recharge in the western United States, Meixner et al. (2016) show the largest declines in recharge are expected in specific aquifers in the southwestern United States, including the southern portion of the Ogallala aquifer, which is expected to receive 10% less recharge through the year 2050.

[1] Elevated levels of these constituents result from both natural processes and human activities, such as groundwater pumping (Chaudhuri and Ale, 2014a; Gurdak et al., 2009).

[2] Persistent salinity has been observed in west Texas, specifically in the southern Ogallala, northwest Edwards-Trinity (plateau), and Pecos Valley aquifers, largely due to prolonged irrigational groundwater pumping and ensuing alteration of hydraulic gradients leading to groundwater mixing (Chaudhuri and Ale, 2014b). High levels of groundwater salinization associated with prolonged aquifer depletion have also been documented in the Carrizo-Wilcox and southern Gulf Coast aquifers, underlying the Eagle Ford Shale in south Texas (Chaudhuri and Ale, 2014b; Konikow, 2013; Boghici, 2009). Further, elevated levels of constituents, including nitrate, lead, fluoride, chloride, sulfate, iron, manganese, and TDS, have been reported in the Carrizo-Wilcox aquifer (Boghici, 2009).

[3] TWDB (2012) defines groundwater supply as the amount of groundwater that can be produced given current permits and existing infrastructure. By contrast, TWDB (2012) defines groundwater availability as the amount of groundwater that is available regardless of legal or physical availability. Total groundwater availability in Texas is expected to decline by approximately 24% between 2010 and 2060 (TWDB, 2012).

[4] This message is echoed in the 2017 Texas State Water Plan (TWDB, 2016).

Groundwater moves slowly, and natural recharge rates are lower during times of drought (DeSimone et al., 2014). Consequently, as water withdrawals continue to outpace the rate of recharge, aquifer storage will decline further (USGS, 1999), potentially impacting both drinking water resource quantity and quality. For example, research from Steadman et al. (2015) in the Eagle Ford play shows that hydraulic fracturing groundwater consumption exceeds estimated recharge rates in the seven most active counties for drilling.

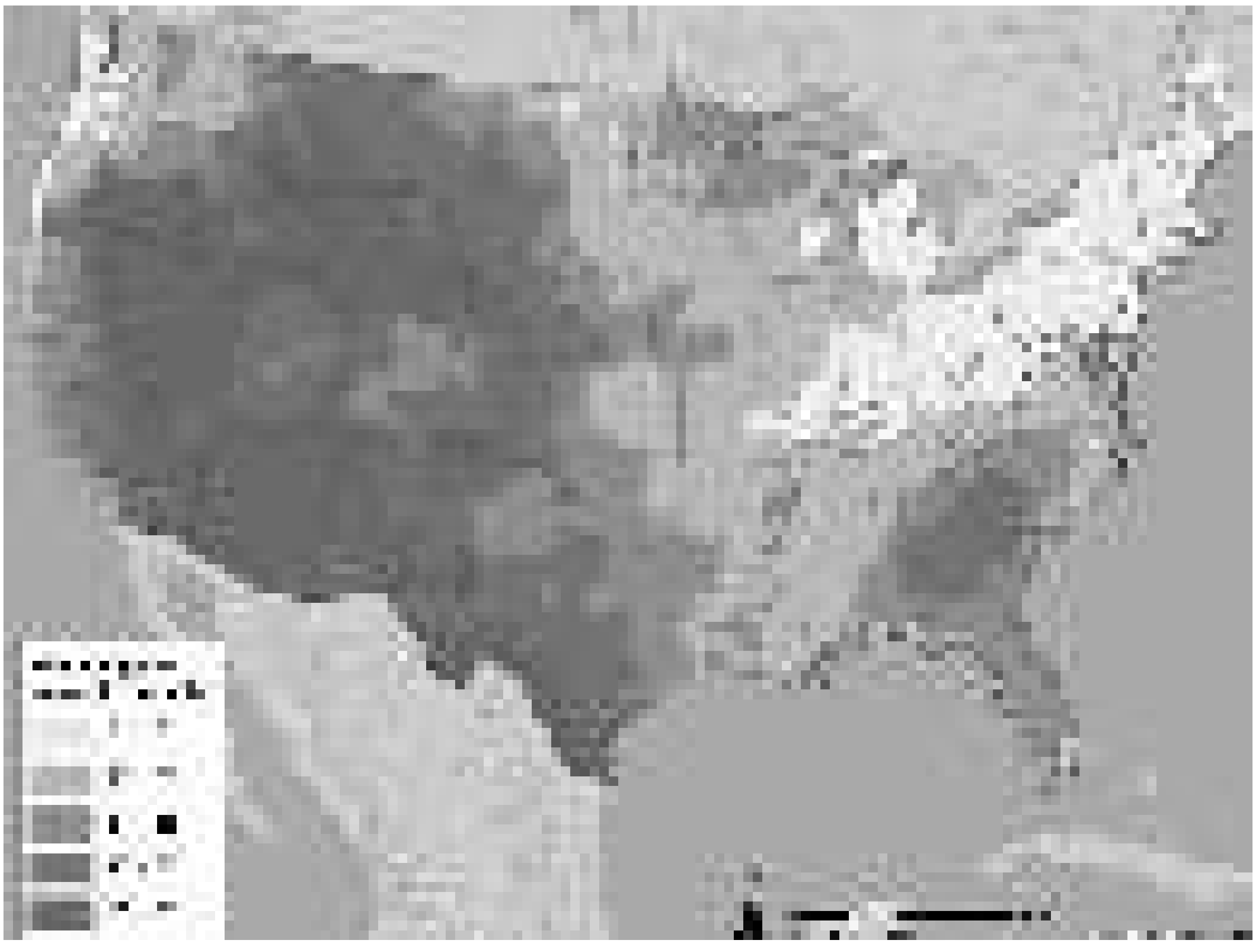


Figure 4-8. Percentage of weeks in drought between 2000 and 2013 by county.
Drought for a given week is defined as any portion of a given U.S. county having a weekly classification of moderate to exceptional drought (D1-D4 categorization) according to the National Drought Mitigation Center (http://droughtmonitor.unl.edu); number of weeks = 731.

A case study in the Eagle Ford play in southwestern Texas compared water demand for hydraulic fracturing with water supplies at the scale of the play, county, and 1 mi^2 (2.6 km^2) (Scanlon et al., 2014b). The authors observed generally adequate water supplies for hydraulic fracturing, except in specific locations, where they found excessive drawdown of groundwater locally in ~6% of the play area, with estimated declines of ~100-200 ft (31-61 m) after hydraulic fracturing activity increased in 2009 (Text Box 4-3).

Text Box 4-3. Case Study: Water Profile of the Eagle Ford Play, Texas.

Researchers from the University of Texas published a detailed case study of water supply and demand for hydraulic fracturing in the Eagle Ford play in southwestern Texas (Scanlon et al., 2014b). This effort assembled detailed information from state and local water authorities, and proprietary industry data on hydraulic fracturing, to develop a portrait of water resources in this 16-county area.

Scanlon et al. (2014b) compared water demand for hydraulic fracturing currently and over the projected play life (20 years) relative to water supply from groundwater recharge, groundwater storage (brackish and fresh), and stream flow. Using groundwater availability models developed by the Texas Water Development Board, they reported that water demand for hydraulic fracturing in 2013 was 30% of annual groundwater recharge in the play area, and over the 20-year play lifespan it was projected to be 26% of groundwater recharge, 5-8% of fresh groundwater storage, and 1% of brackish groundwater storage. The dominant water user in the play is irrigation (57 to 61% of water use, 62 to 65% of consumption), as compared with hydraulic fracturing (13% of water use and 16% of consumption). At the county level, projected water demand for hydraulic fracturing over the 20-year period was low relative to freshwater supply (ranging from 0.6-27% by county, with an average of 7.3%). Similarly, projected total water demand from all uses was low relative to supply, excluding two counties with high irrigation demands (Frio, Zavala), and one county with no known groundwater supplies (Maverick).

Although supply was found to be sufficient even in this semi-arid region, there were important exceptions, especially at sub-county scales. The researchers found no water level declines over much of the play area assessed (69% of the play area), yet in some areas they estimated groundwater drawdowns of 50 ft (15 m) or more (19% of the play area), 100 ft (31 m) or more (6% of the play area), and 200 ft (60 m) or more (approximately 2% of the play area). This was corroborated with well monitoring data that showed a sharp decline in water levels in several groundwater monitoring wells after hydraulic fracturing activity increased in 2009.

The researchers further concluded that shifting toward brackish groundwater is feasible, as evidenced by operators already doing so. This shift could further reduce impacts on fresh water resources and provide a large source of water for future hydraulic fracturing. In a 2011 estimate, approximately 20% of water used in the play came from brackish sources (Table 4-4), and anecdotal evidence suggests this practice has increased since then (Scanlon et al., 2014b). Projected hydraulic fracturing water use represents less than 1% of total brackish groundwater storage in the play area. By contrast, Scanlon et al. (2014b) concluded there is limited potential for reuse of wastewater in this play because of the small volumes that return to the surface during production (less than or equal to 5% of hydraulic fracturing water requirements).

In contrast to southern and western Texas, the potential for water quantity and quality effects appears to be lower in the north-central and eastern parts of the state, in areas including the Barnett and Haynesville plays. Residents obtain water for domestic use—which includes use of water for drinking—from a mixture of groundwater and surface water sources (Appendix Table B-6). Counties encompassing Dallas and Fort Worth rely mostly on publicly-supplied surface water (TWDB, 2012) (Appendix Table B-6). The Trinity aquifer in northeast Texas is projected to decline only slightly between 2010 and 2060 (TWDB, 2012). Nevertheless, Bene et al. (2007) estimate that hydraulic fracturing groundwater withdrawals will increase from 3% of total groundwater use in 2005 to 7%–13% in 2025, suggesting the potential for localized aquifer drawdown. Groundwater quality degradation associated with aquifer drawdown has been documented in the Trinity and Woodbine aquifers overlying much of the Barnett play, with both aquifers showing high levels of salinization (Chaudhuri and Ale, 2013).

Overall, the potential for impacts appears higher in western and southern Texas, compared to the northeast part of the state. Groundwater withdrawals for hydraulic fracturing, along with irrigation and other uses, may contribute to water quality degradation associated with intensive aquifer pumping in western and southern Texas. Areas with numerous high-capacity wells and large amounts of sustained groundwater pumping are most likely to experience groundwater quality degradation associated with withdrawals (Gurdak et al., 2009; McMahon et al., 2007). Further, given that Texas is prone to drought conditions and groundwater recharge is limited, the already declining aquifers in southern and western Texas are especially vulnerable to further groundwater depletion and resulting impacts to groundwater quantity and quality (Gurdak et al., 2009; Jackson et al., 2001). Impacts are likely to be localized drawdowns of groundwater, as shown by a detailed case study of the Eagle Ford play (Text Box 4-3). Scanlon et al. (2014b) suggested that a shift toward brackish water use could minimize potential future impacts to fresh water resources. This finding is consistent with our county level data (Text Box 4-2).

4.5.2 Colorado and Wyoming

Colorado had the second highest number of disclosures in the EPA FracFocus 1.0 project database, (13% of disclosures) (Figure 4-4 and Appendix Table B-5). We combine Colorado and Wyoming because of their shared geology of the Denver Basin (including the Niobrara play) and the Greater Green River Basin (Figure 4-9). There are three major basins reported for Colorado: the Denver Basin; the Uinta-Piceance Basin; and the Raton Basin. Together these basins contain 99% of reported wells in the state, although the bulk of the activity in Colorado is in the Denver Basin (Appendix Table B-5). Fewer wells (roughly 4% of disclosures in the EPA FracFocus 1.0 project database) are reported in Wyoming. There are two major basins reported for Wyoming (Greater Green River and Powder River) that together contain 86% of activity in the state (Appendix Table B-5).

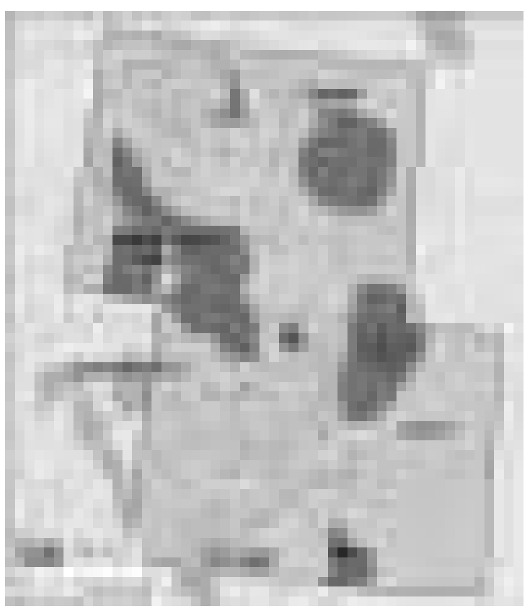


Figure 4-9. Major U.S. EIA shale plays and basins for Colorado and Wyoming.
Source: EIA (2015).

Types of water used: Water for hydraulic fracturing in Colorado and Wyoming comes from both groundwater and surface water, as well as reused wastewater (Colorado Division of Water Resources et al., 2014; BLM, 2013). Publicly available information on water sources for each state generally comes in the form of a list of potential sources, and detailed information on the types of water used for hydraulic fracturing is not readily accessible.[1] In northwestern Colorado's Garfield County (Uinta-Piceance Basin), the U.S. EPA (2015e) reports that any fresh water used for fracturing comes from surface water sources. In the Denver Basin (Niobrara play) of southeastern Wyoming, qualitative information suggests that groundwater supplies much of the water used for fracturing, although no data were available to characterize the ratio of groundwater to surface water withdrawals (AMEC Environment & Infrastructure, 2014; BLM, 2013; Tyrrell, 2012).

Non-fresh water sources, including industrial and municipal wastewater, brackish groundwater, and reused hydraulic fracturing wastewater, are sometimes listed as potential alternatives to fresh water for fracturing in both Colorado and Wyoming (Colorado Division of Water Resources et al., 2014; BLM, 2013); no data are available to show the extent to which these non-fresh water sources are used at the state or basin level. Based on discussions with industry, the U.S. EPA (2015e) reports that fresh water is used solely for drilling and reused wastewater supplies nearly all the water for hydraulic fracturing in Colorado's Garfield County. This estimate of reused wastewater as a percentage of injected volume is markedly higher than in other locations and likely results from the geologic characteristics of the Piceance tight sand formation, which has naturally high water content and produces large volumes of relatively high-quality wastewater (U.S. EPA, 2015e).

In contrast, a study by Goodwin et al. (2014) assumed no reuse of wastewater for hydraulic fracturing operations by Noble Energy in the Denver-Julesburg Basin of northeastern Colorado (Table 4-2). It is unclear whether this assumption is indicative of reuse practices of other companies in the Denver-Julesburg Basin. The difference in reused wastewater rates reported by the U.S. EPA (2015e) and Goodwin et al. (2014) may indicate an east-west divide in Colorado (i.e., low reuse in the east versus high reuse in the west), due at least in part to differences in wastewater volumes available for reuse. However, further information is needed to adequately characterize reuse patterns in Colorado.

Water use per well: Water use per well varies across Colorado, with median values of 1.8 million, 400,000, and 96,000 gal (6.8 million, 1.5 million, and 360,000 L) in the Uinta-Piceance, Denver, and Raton Basins, respectively, according to the EPA FracFocus 1.0 project database (Appendix Table B-5). Relatively low water volumes per well are reported in Wyoming (Appendix Table B-5). Low volumes reported for the Raton Basin of Colorado and the Powder River Basin of Wyoming are likely due to the prevalence of CBM extraction in these locations (U.S. EPA, 2015k; Sando et al., 2014).

More difficult to explain are the low volumes reported for the Denver Basin in the EPA FracFocus 1.0 project database. These values are lower than volumes reported in other non-CBM basins

[1] The Colorado Oil and Gas Conservation Commission collects information on the sources and quality of water used for hydraulic fracturing, including reused wastewater, with Form 5A, and has done so since June 2012; however, these data are in PDFs linked to individual wells and are not aggregated into a searchable database.

included in Appendix Table B-5. Goodwin et al. (2014) report much higher water use per well in the Denver Basin from 2010 to 2013, with a median of 2.8 million gal (10.6 million L) (although only usage for the Wattenberg Field was reported). Indeed, the 10th–90th percentiles (2.4-3.8 million gal) (9.1-14.4 million L) from Goodwin et al. (2014) are almost completely above those from the EPA FracFocus 1.0 project database for the Denver Basin (Appendix Table B-5).[1] However, it is difficult to draw clear conclusions because of differences in scale (i.e., field in Goodwin et al. (2014) versus basin in the EPA FracFocus 1.0 project database) and operators (i.e., Noble Energy in Goodwin et al. (2014) versus all in the EPA FracFocus 1.0 project database).

Trends in water use per well are generally lacking for Colorado, with the exception of those reported by Goodwin et al. (2014). They found that water use per well is increasing with well length in the Denver Basin; however, they also observed that water intensity (gallons of water per unit energy extracted) did not change, since energy recovery increased along with water use.

Water use/consumption at the county scale: Hydraulic fracturing operations in Colorado use billions of gallons of water, but this amount is a small percentage compared to total water used or consumed at the county scale. In both Garfield and Weld Counties, located in the Uinta-Piceance and Denver Basins, respectively, hydraulic fracturing used more than 1 billion gal (3.8 billion L) annually. Fracturing water use and consumption in these counties exceeded those in all other Colorado counties combined (Appendix Table B-2), but the water used for hydraulic fracturing in Garfield and Weld counties was less than 2% and 3% compared to 2010 total water use and consumption, respectively. In comparison, irrigated agriculture accounts for over 90% of the water used in both counties (Maupin et al., 2014). Overall, hydraulic fracturing accounts for less than 2% compared to 2010 total water use in all Colorado counties represented in the EPA FracFocus 1.0 project database (Appendix Table B-2). Water use estimates based on the EPA FracFocus 1.0 project database may be low relative to literature and state estimates (Text Box 4-1), but even if estimates from the project database were doubled, hydraulic fracturing water use and consumption would still be less than 4% and 6% compared to 2010 total water use and consumption, respectively, in each Colorado county.

In Wyoming, reported water use for hydraulic fracturing is small compared to Colorado (Appendix Table B-1). Fracturing water use and consumption did not exceed 1% of 2010 total water use and consumption, respectively, in any county (Appendix Table B-2). Unlike Colorado, Wyoming did not require disclosure to FracFocus during the time period analyzed by the EPA (U.S. EPA, 2015b) (Appendix Table B-5).

Colorado Division of Water Resources et al. (2014) projected that annual water use for hydraulic fracturing in the state would increase by approximately 16% between 2012 and 2015, but demand in later years is unclear. Even with an increase of 16% or more, hydraulic fracturing would still remain a relatively small user of water at the county scale in Colorado.

[1] Different spatial extents might explain these differences, since Goodwin et al. (2014) focus on 200 wells in the Wattenberg Field of the Denver Basin; however, Weld County is the center of activity in the Wattenberg Field, and the EPA FracFocus 1.0 project database contains 3,011 disclosures reported in Weld County, with a median water use per of 407,442 gal (1,542,340 L), similar to that for the basin as a whole.

Potential for impacts: The potential for water quantity and quality impacts due to hydraulic fracturing water withdrawals appears to be low at the county scale in Colorado and Wyoming because fracturing accounts for a low percentage of total water use and consumption (Figure 4-3a,b). This conclusion is also supported by the comparison of hydraulic fracturing water use to water availability at the county scale (Text Box 4-2; Figure 4-6a,b). However, counties in Colorado and Wyoming are large in their spatial extents, and any potential impacts will depend on site-specific factors affecting the balance between water use and availability at the local scale (i.e., at a given withdrawal point). In a multi-scale case study in the Upper Colorado River Basin, the U.S. EPA (2015e) did not identify any locations where fracturing currently contributed to locally high water use intensity due to the high rates of wastewater reuse reported. They did conclude, however, that future effects may be possible (Text Box 4-4).

Text Box 4-4. Case Study: Impact of Water Acquisition for Hydraulic Fracturing on Local Water Availability in the Upper Colorado River Basin.

The U.S. EPA (2015e) conducted a case study to explore the impact of hydraulic fracturing water demand on water availability at the river basin, county, and local scales in the semi-arid Upper Colorado River Basin (UCRB) of western Colorado. The study area overlies the Piceance geologic basin with natural gas in tight sands. Water withdrawal impacts were quantified using a water use intensity index (i.e., the ratio between the volume of water withdrawn at a site for hydraulic fracturing and the volume of available water). Researchers obtained detailed site-specific data on hydraulic fracturing water usage from state and regional authorities, and estimated available water supplies using observations at USGS gage stations and empirical and hydrologic modeling.

They found that water supplies accessed for oil and gas demand were concentrated in Garfield County, and most fresh water withdrawals were concentrated within the Parachute Creek watershed (198 mi^2). However, fresh water makes up a small proportion of the total water used for fracturing due to large quantities of high-quality wastewater produced from the Piceance tight sands. Based on discussions with industry, the U.S. EPA (2015e) reports that fresh water is used solely for drilling and reused wastewater supplies nearly all the water for hydraulic fracturing in Garfield County. Due to the high reuse rate, the U.S. EPA (2015e) did not identify any locations in the Piceance play where fracturing contributed to locally high water use intensity.

Scenario analyses demonstrated a pattern of increasing potential impact with decreasing watershed size in the UCRB. The U.S. EPA (2015e) examined hydraulic fracturing water use intensity under the current rates of both directional (S-shaped) and horizontal drilling. They showed that for the more water-intensive horizontal drilling, watersheds had to be larger to meet the same index of water use intensity (0.4) as that for directional drilling (100 mi^2 for horizontal drilling, as compared to 30 mi^2 for directional drilling). To date, most wells have been drilled directionally into the Piceance tight sands, although a trend toward horizontal drilling is expected to increase annual water use per well by about four times. Despite this increase, total hydraulic fracturing water use is expected to remain small relative to other users. Currently, irrigated agriculture is the largest water user in the UCRB.

Greater water demand could occur in the future if the water-intensive oil shale extraction industry becomes economically viable in the region. Projections for oil shale water demand indicate that the industry could increase water use for energy extraction in Garfield and Rio Blanco counties.

East of the Rocky Mountains in the Denver Basin, the potential for localized impacts exists given the combination of high hydraulic fracturing activity and low water availability (e.g., Weld County, Colorado), but lack of available data and literature at the local scale limits our ability to assess the potential for impacts in this location. Ceres (2014) concludes that all fractured wells in the Denver Basin are in high or extremely high water-stressed areas. Furthermore, the development of the Niobrara Shale in southeast Wyoming occurs in areas already impacted by high agricultural water use from the Ogallala aquifer, including the state's only three groundwater control areas, which were established as management districts in the southeast portion of the state in response to declining groundwater levels (AMEC Environment & Infrastructure, 2014; Wyoming State Engineer's Office, 2014; Tyrrell, 2012; Bartos and Hallberg, 2011). Groundwater withdrawals for hydraulic fracturing may have the potential to contribute to water quality degradation in these areas, depending on site-specific factors that may alter the balance between water use and availability.

Overall, the potential for impacts appears low at the county scale in Colorado and Wyoming, but local effects are certainly possible particularly east of the Rocky Mountains in the Denver Basin. Lack of available data and literature at the local scale limits our ability to assess the potential for impacts in this location.

4.5.3 Pennsylvania, West Virginia, and Ohio

Pennsylvania had the third most disclosures in the EPA FracFocus 1.0 project database (6.5% of disclosures) (Appendix Table B-5; Figure 4-4). We combine West Virginia and Ohio with Pennsylvania because they share similar geology overlying the Appalachian Basin (including the Marcellus, Devonian, and Utica stacked plays) (Figure 4-10); however, much less activity is reported in these two states (Appendix Table B-5).

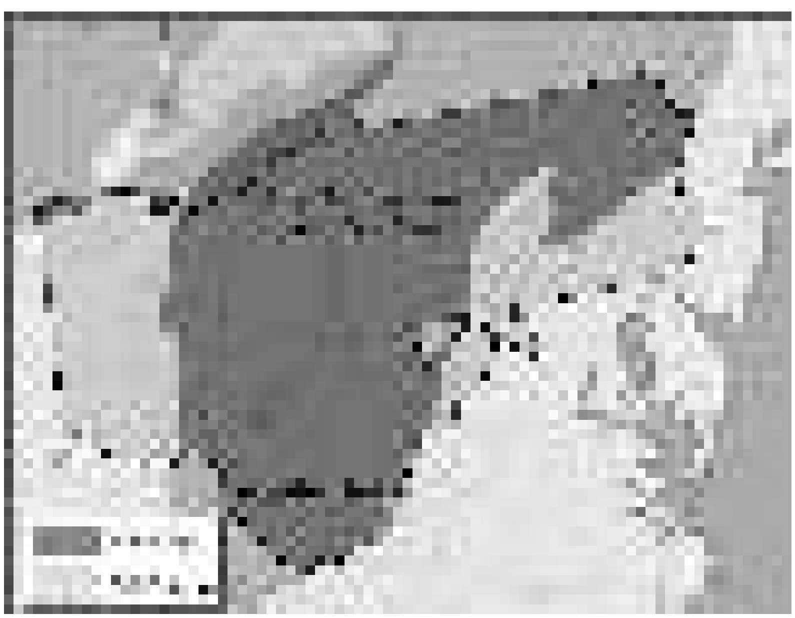


Figure 4-10. Major U.S. EIA shale plays and basins for Pennsylvania, West Virginia, and Ohio.
Source: EIA (2015).

Types of water used: Surface water is the primary water source for hydraulic fracturing in Pennsylvania, West Virginia, and Ohio (SRBC, 2016; Schmid and Yoxtheimer, 2015; West Virginia DEP, 2014; Mitchell et al., 2013a; West Virginia DEP, 2013; Ohio EPA, 2012b; STRONGER, 2011b) (Table 4-1). Further, the water used for hydraulic fracturing is most often fresh water in all three states. In both Pennsylvania's Susquehanna River Basin and throughout West Virginia, most water for hydraulic fracturing is self-supplied via direct withdrawals from surface water and groundwater (U.S. EPA, 2015e; West Virginia DEP, 2013). Operators also purchase water from public water systems, which may include a variety of commercial water brokers (West Virginia DEP, 2014; SRBC, 2013; West Virginia DEP, 2013). Municipal supplies are also used, particularly in urban areas of Ohio (STRONGER, 2011b).

Reused hydraulic fracturing wastewater as a percentage of total water used for fracturing was 19% in 2014 in Pennsylvania, and 15% in 2012 in West Virginia (Schmid and Yoxtheimer, 2015; West Virginia DEP, 2014) (Table 4-2). Available data indicate an increasing trend in reuse of wastewater over time in this region, likely due to the lack of nearby disposal options in Class II wells. Reused wastewater as a percentage of injected water volume ranged from approximately 2% to 19% in Pennsylvania (statewide) from 2009-2014 (Schmid and Yoxtheimer, 2015). This upward trend is also shown in Pennsylvania's SRB, where reuse as a percentage of total water injected reached 22% in 2013; the average reuse rate for 2008-2013 in the SRB was 16% (SRBC, 2016) (Table 4-2). In West Virginia, reuse as a percentage of injected volume ranged from 6% to 15% from 2010-2012 (West Virginia DEP, 2014). In Ohio's Marcellus and Utica Shales, reuse of wastewater is reportedly uncommon (STRONGER, 2011b), likely due to the prevalence of disposal wells in Ohio. See Chapter 8 for more information.

Aside from reused hydraulic fracturing wastewater, other types of wastewaters reused for hydraulic fracturing may include wastewater treatment plant effluent, treated acid mine drainage, and rainwater collected at various well pads (West Virginia DEP, 2014; SRBC, 2013; West Virginia DEP, 2013; Ziemkiewicz et al., 2013; Ohio EPA, 2012b). No data are available on the frequency of use of these other wastewaters.

Water use per well: Operators in these three states reported the third, fourth, and fifth highest median water use per well of the states we considered from the EPA FracFocus 1.0 project database, with 5.0, 4.2, and 3.9 million gal (18.9, 15.9, and 14.8 million L) in West Virginia, Pennsylvania, and Ohio, respectively (Appendix Table B-5). Hansen et al. (2013) report similar water use estimates for Pennsylvania and West Virginia for 2011 and 2012 (Appendix Table B-5). This correspondence is not surprising, as these estimates are also based on FracFocus data (via Skytruth). For 2011, the year overlapping with the time frame of the EPA FracFocus report (U.S. EPA, 2015b), Mitchell et al. (2013a) report an average of 2.3 million gal (8.7 million L) for vertical wells (54 wells) and 4.6 million gal (17.4 million L) for horizontal wells (612 wells) in the Pennsylvania portion of the Upper Ohio River Basin, based on records from PA DEP. The weighted average water use per well was 4.4 million gal (16.7 million L), similar to results based on the EPA FracFocus 1.0 project database listed above. In Pennsylvania's SRB, the long-term average water use per well from 2008-2013 was 4.3 million gal (16.3 million L). In 2013, the average water use per well increased to approximately 5.1 to 6.5 million gal (19.3 to 24.6 million L) due to increasing lengths of laterals in horizontal drilling (SRBC, 2016). Across the entire state of Pennsylvania, water use per well has increased over time, which may be explained by increasing horizontal well length, depth, and length of the completed interval (Schmid and Yoxtheimer, 2015).

Water use/consumption at the county scale: In this tri-state region, the highest water use for hydraulic fracturing is in northeastern Pennsylvania counties. On average, operators in Bradford County reported over 1 billion gal (3.8 billion L) used annually in 2011 and 2012 for fracturing; operators in three other counties (Susquehanna, Lycoming, and Tioga Counties) reported 500 million gal (1.9 billion L) or more used annually in each county (Table 4-3). On average, hydraulic fracturing water use is 3.2% compared to 2010 total water use for counties with disclosures in the EPA FracFocus 1.0 project database in these three states (Table 4-3; Appendix Table B-2). Susquehanna County in Pennsylvania has the highest percentages relative to 2010 total water use (47%) and consumption (123%).

Potential for impacts: Water availability is higher in Pennsylvania, West Virginia, and Ohio than in many western states, reducing the likelihood of impacts to drinking water resource quantity and quality. At the county scale, water supplies appear adequate to accommodate this use (Text Box 4-2; Figure 4-6a,b). However, impacts could still occur at the local scale (i.e., specific withdrawal points) as high water availability in a region does not preclude water stress, particularly if water withdrawals occur during seasonal low-flow periods (Entrekin et al., 2015). Without management of the rate and timing of withdrawals, surface water withdrawals for hydraulic fracturing have the potential to affect both drinking water quantity and quality (Mitchell et al., 2013a). For instance, withdrawals may alter natural stream flow regimes, potentially decreasing a stream's capacity to dilute contaminants (Gallegos et al., 2015; Mitchell et al., 2013a; Entrekin et al., 2011; NYSDEC, 2011; van Vliet and Zwolsman, 2008; IPCC, 2007; Environment Canada, 2004; Murdoch et al., 2000).

In a second, multi-scale case study, EPA showed that the potential for water acquisition impacts to drinking water resource quantity and quality increases at finer temporal and spatial resolutions (U.S. EPA, 2015e). They concluded that individual streams in Pennsylvania's SRB can be vulnerable to typical hydraulic fracturing water withdrawals depending on stream size, as defined by contributing basin area (U.S. EPA, 2015e) (Text Box 4-5). They observed infrequent (in less than 1% of withdrawals) high ratios of hydraulic fracturing water consumption to stream flow (high consumption-to-stream flow events). Further research from Barth-Naftilan et al. (2015) in Pennsylvania's Marcellus Shale (SRB and Ohio River Basin (ORB)) confirmed that stream flow alteration due to hydraulic fracturing surface water withdrawals increases at finer spatial scales (i.e., smaller watershed area). They showed that streams with drainage areas under 50 mi^2 (130 km^2) are the most vulnerable to stress induced by flow alteration (Barth-Naftilan et al., 2015).

Text Box 4-5. Case Study: Impact of Water Acquisition for Hydraulic Fracturing on Local Water Availability in the Susquehanna River Basin.

The U.S. EPA (2015e) conducted a second case study analogous to that in the UCRB (Text Box 4-4), to explore the impact of hydraulic fracturing water demand on water availability at the river basin, county, and local scales in the SRB in northeastern Pennsylvania. The study area overlies the Marcellus Shale gas reservoir. Water withdrawal impacts were quantified using a water use intensity index (Text Box 4-4). Researchers obtained detailed site-specific data on hydraulic fracturing water usage from state and regional authorities, and estimated available water supplies using observations at USGS gage stations and empirical and hydrologic modeling.

Most water for fracturing in the SRB is self-supplied by operators from rivers and streams with withdrawal points distributed throughout a wide geographic area. Public water systems provide a relatively small proportion of the water needed. Reuse of wastewater as a percentage of hydraulic fracturing fluid volume averaged 16% from 2008-2013, and has increased over time, reaching 22% in 2013 (SRBC, 2016) (Table 4-2). The Susquehanna River Basin Commission (SRBC) regulates water acquisition for hydraulic fracturing and issues permits that set limits on the volume, rate, and timing of withdrawals at individual withdrawal points; passby flow thresholds (hereafter, passby flows) halt water withdrawals during low flows.

The U.S. EPA (2015e) demonstrated that streams can be vulnerable from hydraulic fracturing water withdrawals depending on their size, as defined by contributing basin area. Small streams have the potential for impacts (i.e., high water use intensity) for all or most of the year. The U.S. EPA (2015e) showed an increased likelihood of impacts in small watersheds in the SRB (less than 10 mi^2 or 26 km^2). Furthermore, they showed that in the absence of passby flows, even larger watersheds (up to 600 mi^2 or 1,554 km^2) could be vulnerable during maximum withdrawal volumes and infrequent droughts. However, high water use intensity calculated from observed hydraulic fracturing withdrawals occurred at only a few withdrawal locations in small streams; local high water use intensity was not found at the majority of withdrawal points.

Detailed studies and state reports available throughout the Marcellus Shale region help provide an understanding of the potential impacts of hydraulic fracturing water withdrawals in both space and time at the local scale (SRBC, 2016; Barth-Naftilan et al., 2015; U.S. EPA, 2015e). In the SRB and ORB, water for hydraulic fracturing is taken from both large rivers and small headwater streams, with a considerable fraction of the water taken from small streams of small watersheds (Barth-Naftilan et al., 2015). The SRBC reports that most natural gas development in the SRB is focused in rural, headwater areas, where withdrawals have the potential to alter natural stream flow regimes (SRBC, 2016). In an analysis of the effects of water withdrawals on twelve streams in the SRB, Shank and Stauffer (2015) found that the largest withdrawals relative to stream size were from headwater streams, where daily withdrawals averaged 6.8% of average daily flows. However, they found water management in the form of low flow protections helped limit the potential for impacts.

Compared to conventional energy extraction, hydraulic fracturing consumes more water in a highly concentrated period of time (Patterson et al., 2016); thus, the cumulative impact of multiple wells withdrawing water from small streams, particularly during drought or seasonal low flows, has the potential to impact the quantity and quality of drinking water resources (Patterson et al., 2016). For instance, in modeling the potential future impact of hydraulic fracturing in the Delaware River Basin (DRB), Habicht et al. (2015) showed that under maximum well development, hydraulic fracturing water withdrawals from small streams could remove up to 70% of water during periods of low stream flow, and less than 3% during periods of normal stream flow.[1] Unlike groundwater withdrawals, any impacts to drinking water resource quantity and quality associated with surface water withdrawals are likely to persist for a shorter time period since the rate of replenishing water removed from the system is greater in surface water than groundwater (Alley et al., 1999) (Section 4.5.1).

The potential for water acquisition impacts to drinking water resource quality in this region is also greatest in small, unregulated streams, particularly under drought conditions or during seasonal low flows (U.S. EPA, 2015e; Vengosh et al., 2014; Mitchell et al., 2013a; Vidic et al., 2013; Rahm and Riha, 2012; Rolls et al., 2012; Kargbo et al., 2010; McKay and King, 2006). Surface water quality impacts may be of concern if a pollution discharge point (e.g., sewage treatment plant, agricultural runoff, or chemical spill) is immediately downstream of a hydraulic fracturing withdrawal point (U.S. EPA, 2015e; NYSDEC, 2011).[2] Potential water quality impacts associated with reduced water levels may also include possible interference with the efficiency of drinking water treatment plant operations, as increased contaminant concentrations in drinking water sources may necessitate additional treatment and ultimately impact drinking water quality (Water Research Foundation, 2014; Benotti et al., 2010).[3]

Water management policies in place in this region can help reduce the potential for impacts associated with hydraulic fracturing water withdrawals, including excessive lowering of water levels, unreliable water supplies, and degradation of water quality (SRBC, 2016; Barth-Naftilan et al., 2015; U.S. EPA, 2015e) (Text Box 4-5). For instance, the SRBC manages the quantity, location, and timing of withdrawals, using site-specific information to set instantaneous and daily withdrawal limits for all approved surface water and groundwater withdrawals. They also set low flow protections, known as passby flows, for most approved surface water withdrawals that require withdrawals to cease when stream flow drops below a prescribed threshold level (SRBC, 2016). Passby flows can reduce the frequency of high consumption-to-stream flow events, particularly in the smallest streams (Shank and Stauffer, 2015; U.S. EPA, 2015e).

Overall, there appears to be adequate surface water for hydraulic fracturing in Pennsylvania, West Virginia, and Ohio, but there is still the potential for impacts to both drinking water resource quantity and quality, particularly in small streams, if the rate and timing of withdrawals are not managed (U.S. EPA, 2015e). These potential impacts are expected to be localized in space (i.e.,

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[1] Presently there is a moratorium on hydraulic fracturing in the DRB, which spans Pennsylvania, Delaware, New Jersey, and New York. Habicht et al. (2015) modeled the potential future environmental impact of hydraulic fracturing in the DRB should the moratorium be lifted, allowing hydraulic fracturing to expand into this region in the future.

[2] Aside from direct surface water withdrawals, unmanaged withdrawals from public water systems can cause cross-contamination if there is a loss of pressure, allowing the backflow of pollutants from tank trucks into the distribution system. The state of Ohio has issued a fact sheet relevant to this potential concern, intended specifically for public water systems providing water to oil and gas companies (Ohio EPA, 2012a). To prevent potential cross-contamination, Ohio requires a backflow prevention device at cross-connections. For example, bulk loading stations that provide public supply water directly to tank trucks are required to have an air-gap device at the cross-connection to prevent the backflow of contaminants into the public water system (Ohio EPA, 2012a).

[3] For instance, an increased proportion of organic matter entering a treatment plant may increase the formation of trihalomethanes, byproducts of the disinfection process formed as chlorine reacts with organic matter in the water being treated (Water Research Foundation, 2014).

occurring at specific withdrawal points), and time (e.g., low flow periods). Passby flows appear to be an effective water management tool for reducing the potential for impacts from surface water withdrawals.

4.5.4 North Dakota and Montana

North Dakota was fourth in the number of disclosures in the EPA FracFocus 1.0 project database (5.9% of disclosures) (Appendix Table B-5; Figure 4-4). We combine Montana with North Dakota, because both overlie the Williston Basin (which contains the Bakken play, shown in Figure 4-11), although many fewer wells are reported for Montana (Appendix Table B-5). The Williston Basin is the only basin with significant activity reported for either state, though other basins are also present in Montana (e.g., the Powder River Basin).

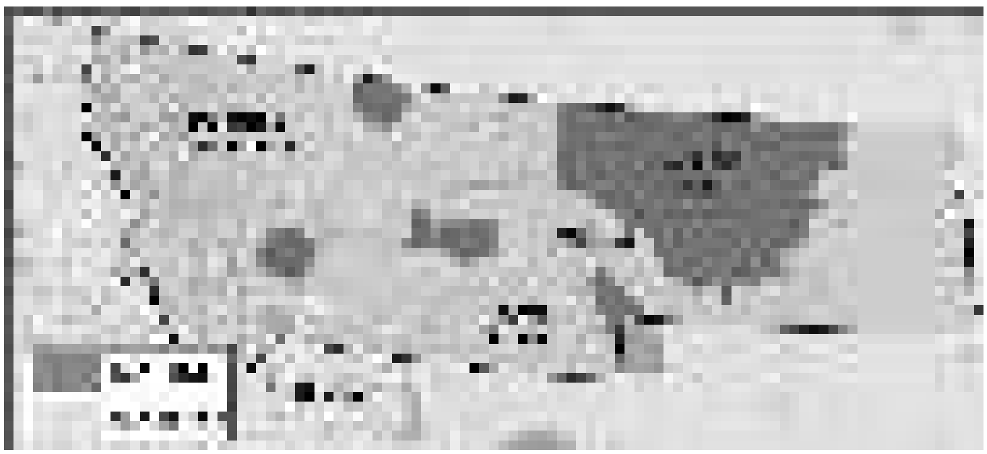


Figure 4-11. Major U.S. EIA shale plays and basins for North Dakota and Montana.
Source: EIA (2015).

Types of water used: Hydraulic fracturing in the Bakken play depends on both ground and surface water resources. Surface water from the Missouri River system provides the largest source of fresh water in the center of Bakken oil development (North Dakota State Water Commission, 2014; EERC, 2011, 2010; North Dakota State Water Commission, 2010). Apart from the Missouri River system, regional surface waters (e.g., smaller streams) do not provide a consistent supply of water for the oil industry due to seasonal stream flow variations. Sufficient stream flows generally occur only in the spring after snowmelt (EERC, 2011). Groundwater from glacial and bedrock aquifer systems has traditionally supplied much of the water needed for Bakken development, but concerns over limited groundwater supplies have led to limits on the number of new groundwater withdrawal permits issued (Ceres, 2014; Plummer et al., 2013; EERC, 2011, 2010; North Dakota State Water Commission, 2010).

The water used for Bakken development is mostly fresh. The EPA FracFocus report shows that "fresh" was the only source of water listed in almost all disclosures reporting a source of water in North Dakota (U.S. EPA, 2015b).[1] Reuse of Bakken wastewater is limited due to its high TDS, which

[1] Twenty-five percent of North Dakota disclosures included information related to water sources (U.S. EPA, 2015b).

presents challenges for treatment and reuse (Gadhamshetty et al., 2015). Industry is currently researching treatment technologies for reuse of this wastewater (Ceres, 2014; EERC, 2013, 2011).

Water for hydraulic fracturing is commonly purchased from municipalities or other public water systems in the region. The water is often delivered to trucks at water depots or transported directly to well pads via pipelines (EERC, 2011).

Water use per well: Water use per well is intermediate compared with other areas, with a median of 2.0 and 1.6 million gal (7.6 and 6.1 million L) per well in the Williston Basin in North Dakota and Montana, respectively, according to the EPA's FracFocus 1.0 project database (Appendix Table B-5). The North Dakota State Water Commission reports similar volumes (2.2 million gal (8.3 million L) per well on average for North Dakota) in a summary fact sheet (North Dakota State Water Commission, 2014).[1] Scanlon et al. (2016) show that average water use per well in the Bakken play has increased over time, from 580,000 gal (2.2 million L) in 2005 to 3.7 million gal (14.1 million L) in 2014, due in part to the increasing lengths of laterals in horizontal drilling.

In addition to water for hydraulic fracturing, Bakken wells may require "maintenance water" (Scanlon et al., 2016; Scanlon et al., 2014a). This extra water is reportedly needed because of the relatively high salt content of Bakken brine, potentially leading to salt buildup, pumping problems, and restriction of oil flow. Based on estimates from the North Dakota Department of Mineral Resources, Scanlon et al. (2016) report that approximately 400 – 600 gal (1,500 – 2,300 L) per day per each well may be required for well maintenance. Assuming a 15-year lifetime for wells, this could add up to 3.3 million gal (12.5 million L) per well of additional water (Scanlon et al., 2016).

Water use/consumption at the county scale: Water use for fracturing in this region is greatest in the northwestern corner of North Dakota (Gadhamshetty et al., 2015). Hydraulic fracturing water use in 2011 and 2012 averaged approximately 123 million gal (466 million L) per county in the two-state area, with use in McKenzie and Williams Counties in North Dakota exceeding 500 million gal (1.9 billion L) (Appendix Table B-2). There were four counties where 2011 and 2012 average hydraulic fracturing water use was 10% or more of 2010 total water use. Mountrail and Dunn Counties showed the highest percentages (36% and 29%, respectively). Outside of North Dakota's northwest corner, hydraulic fracturing used much less water in the rest of the state and Montana (Table 4-3; Appendix Table B-2).

Potential for impacts: In this region, there are concerns about over-pumping groundwater resources, but the potential for impacts appears to be low provided the Missouri River is determined to be a sustainable and usable source. This finding of a low potential for impacts is also supported by the comparison of hydraulic fracturing water use to water availability at the county scale (Text Box 4-2; Figure 4-6a,b). This area is primarily rural, interspersed with small towns. Residents rely on a mixture of surface water and groundwater for domestic use depending on the county, with most water supplied by local municipalities (Appendix Table B-6).

[1] The fact sheet is a stand-alone piece, and it is not accompanied by an underlying report.

The state of North Dakota and the U.S. Army Corps of Engineers concluded that groundwater resources in western North Dakota are not sufficient to meet the needs of the oil and gas industry (U.S. Army Corps of Engineers, 2011; North Dakota State Water Commission, 2010). All users combined currently withdraw approximately 6.2 billion gal (23.5 billion L) of water annually in an 11-county region in western North Dakota, already stressing groundwater supplies (U.S. Army Corps of Engineers, 2011). By comparison, the total needs of the oil and gas industry are projected to range from approximately 2.2 and 8.8 billion gal (8.3 and 33.3 billion L) annually by the year 2020 (U.S. Army Corps of Engineers, 2011).

Due to concerns for already stressed groundwater supplies, the state of North Dakota limits industrial groundwater withdrawals, particularly from the Fox Hills-Hell Creek aquifer (Ceres, 2014; Plummer et al., 2013; EERC, 2011, 2010; North Dakota State Water Commission, 2010). Currently, the oil industry is the largest industrial user of water from the Fox Hills-Hell Creek aquifer (North Dakota State Water Commission, 2010). Many farms, ranches, and some communities in western North Dakota rely on flowing wells from this artesian aquifer, particularly in remote areas that lack electricity for pumping; however, low recharge rates and withdrawals throughout the last century have resulted in steady declines in the formation's hydraulic pressure (North Dakota State Water Commission, 2010). Declines in hydraulic pressure do not appear to be associated with impacts to groundwater quality; rather, the state is concerned with maintaining flows for users (North Dakota State Water Commission, 2010).

To reduce demand for groundwater, the state is encouraging the industry to seek surface water withdrawals from the Missouri River system. The North Dakota State Water Commission concluded the Missouri River and its dammed reservoir, Lake Sakakawea, are the only plentiful and dependable water supplies for the oil industry in western North Dakota (North Dakota State Water Commission, 2010). In 2011, North Dakota authorized the Western Area Supply Project, by which Missouri River water (via the water treatment plant in Williston, North Dakota) will be supplied to help meet water demands, including for oil and gas development, of the state's northwest counties (WAWSA, 2011). In July 2012, the U.S. Army Corps of Engineers made available approximately 32.6 billion gal (123 billion L) of water per year from Lake Sakakawea for municipal and industrial water demands over the next ten years (U.S. Army Corps of Engineers, 2011). The Army Corps estimated that the oil and gas industry could use up to 8.8 billion gal (33.3 billion L) annually during this time period in the 11-county surrounding area, and included this as part of the 32.6 billion gal total (123 billion L) to be made available (U.S. Army Corps of Engineers, 2011). For context, annual water use for hydraulic fracturing in all North Dakota counties combined was approximately 2.2 billion gal (8.3 billion L) per year in 2011 and 2012 according to EPA's FracFocus 1.0 project database (Appendix Table B-2). As such, Lake Sakakawea appears to be an adequate resource to meet the water demands of hydraulic fracturing in the region at least in the near term.

4.5.5 Arkansas and Louisiana

Arkansas and Louisiana were ranked seventh and tenth in the number of disclosures in the EPA FracFocus 1.0 project database, respectively (Appendix Table B-5). Hydraulic fracturing activity in Louisiana occurs primarily in the TX-LA-MS Salt Basin, which contains the Haynesville play; activity in Arkansas is dominated by the Arkoma Basin, which contains the Fayetteville play (Figure 4-12).

Figure 4-12. Major U.S. EIA shale plays and basins for Arkansas and Louisiana.
Source: EIA (2015).

Types of water used: Surface water is reported as the primary source of water for hydraulic fracturing operations in both Arkansas and Louisiana (ANRC, 2014; LA Ground Water Resources Commission, 2012; STRONGER, 2012). Quantitative information is lacking for Arkansas on the proportion of water sourced from surface versus groundwater. However, data are available for Louisiana, where an estimated 87% of water for hydraulic fracturing in the Haynesville Shale is from surface water (LA Ground Water Resources Commission, 2012) (Table 4-1). In 2008, during the early stages of development, hydraulic fracturing in Louisiana relied heavily on groundwater from the Carrizo-Wilcox aquifer, and concerns for the sustainability of groundwater resources prompted the state to encourage surface water withdrawals (LA Ground Water Resources Commission, 2012).

The EPA FracFocus report suggests that significant reuse of wastewater may occur in Arkansas to offset total fresh water used for hydraulic fracturing; 70% of all disclosures reporting a water source indicated a blend of "recycled/surface," whereas 3% of disclosures reporting a water source noted "fresh" as the exclusive water source (U.S. EPA, 2015b).[1] According to Veil (2011), Arkansas'

[1] Ninety-three percent of Arkansas disclosures included information related to water sources (U.S. EPA, 2015b).

Fayetteville Shale wastewater is of relatively good quality (i.e., low TDS), facilitating reuse.[1] Data are generally lacking on the extent to which hydraulic fracturing wastewater is reused in Louisiana.

Water use per well: Arkansas and Louisiana have the highest median water use per well of the states we considered from the EPA FracFocus 1.0 project database, at 5.3 million and 5.1 million gal (20.1 million and 19.3 million L), respectively (Appendix Table B-5).[2]

Water use/consumption at the county scale: On average, hydraulic fracturing uses 408 million gal (1.54 billion L) of water each year in Arkansas counties reporting activity, or 9.3% of 2010 total county water use (26.9% of total county consumption) (Appendix Table B-2). In 2011 and 2012, five counties dominated fracturing water use in Arkansas: Cleburne, Conway, Faulkner, Van Buren, and White Counties (Appendix Table B-2). Van Buren, which is sparsely populated and thus has relatively low total water use and consumption, is by far the Arkansas county highest in hydraulic fracturing water use and consumption relative to 2010 total water use and consumption (56% and 168%, respectively) (Table 4-3).

In Louisiana, hydraulic fracturing water use is concentrated in six parishes in the far northwestern corner of the state, associated with the Haynesville play.[3] On average in 2011 and 2012, hydraulic fracturing used 117 million gal (443 million L) of water annually per parish, representing approximately 3.6% and 10.8% of 2010 total water use and consumption, respectively (Appendix Table B-2). Operators in DeSoto Parish used the most water (over 1 billion gal (3.8 billion L) annually). Hydraulic fracturing water use and consumption was highest relative to 2010 total water use and consumption (35.5% and 83.2%, respectively) in Red River Parish (Table 4-3). These numbers may be low estimates, since Louisiana required disclosures to the state or FracFocus, and Arkansas required disclosures to the state but not FracFocus, during the time period analyzed (U.S. EPA, 2015b) (Appendix Table B-5).

Potential for impacts: Water availability is generally higher in Arkansas and Louisiana than in states farther west, reducing the potential for impacts to drinking water quantity and quality (Figure 4-6a, Figure 4-7a; Text Box 4-2). However, generally high water availability in this region does not preclude the potential for impacts at the local scale, particularly if surface water withdrawals occur during seasonal low flow periods. For instance, precipitation is highest in Arkansas in the late autumn and winter, with little rainfall occurring in the late spring and summer; thus, most small streams do not flow year round (Entrekin et al., 2015). Hydraulic fracturing surface water withdrawals from small streams during seasonal low flows have the potential to impact the quantity and quality of drinking water resources.

Additionally, in northwestern Louisiana, there are concerns about over-pumping of groundwater resources. Prior to 2008, most operators in the Louisiana portion of the Haynesville Shale used groundwater, withdrawing from the Carrizo-Wilcox, Upland Terrace, and Red River Alluvial aquifer

[1] Veil (2011) reports a range of 20,000-25,000 ppm TDS for Fayetteville Shale wastewater.

[2] According to STRONGER (2012) and STRONGER (2011a), both states require disclosure of information on water use per well, but this has not been synthesized into state level reports to date.

[3] Louisiana is divided into parishes, which are similar to counties in other states.

systems (LA Ground Water Resources Commission, 2012). To mitigate stress on groundwater, the state issued a water use advisory to the oil and gas industry that recommended Haynesville Shale operators seek alternative water sources to the Carrizo-Wilcox aquifer, which is predominantly used for public supply (LDEQ, 2008). Operators then transitioned to mostly surface water, with a smaller groundwater component (approximately 13% of all fracturing water used) (LA Ground Water Resources Commission, 2012). Of this groundwater component, the majority (approximately 74%) still came from the Carrizo-Wilcox aquifer (LA Ground Water Resources Commission, 2012).

Although the potential for hydraulic fracturing withdrawals to affect water supplies and water quality in the aquifer was reduced, it was not entirely eliminated. Despite Louisiana's water use advisory, a combination of drought conditions and higher than normal withdrawals (for all uses, not solely hydraulic fracturing) from the Carrizo-Wilcox and Upland Terrace aquifers caused several water wells to go dry in July 2011 (LA Ground Water Resources Commission, 2012). In August 2011, a groundwater emergency was declared for southern Caddo Parrish (LA Ground Water Resources Commission, 2012). Hydraulic fracturing withdrawals contributed to these conditions, alongside other users of water and the lack of precipitation.

4.6 Chapter Synthesis

In this chapter, we examined the potential for water acquisition for hydraulic fracturing to impact the quantity and quality of drinking water resources, and identified factors affecting the frequency or severity of impacts. Whether impacts occur from water acquisition for hydraulic fracturing depends on the local balance between water withdrawals and availability, and this balance can be modified by a combination of site or regional-specific factors. For this reason, information is needed at the local scale to determine whether impacts actually occur, yet this information is not available in many locations where hydraulic fracturing takes place; see Section 4.6.3 on Uncertainties below. Despite these limitations, our chapter used the scientific literature, county level assessments, and, where available, local case studies to point to areas with a higher potential for impacts; understand local dynamics, including example cases of impacts; and identify common factors that increase or decrease the frequency or severity of impacts. In this section, we summarize our major findings regarding hydraulic fracturing water acquisition activities, potential impacts, and these common factors (4.6.1 and 4.6.2). We then discuss uncertainities (4.6.3), and provide final conclusions (4.6.4).

4.6.1 Major Findings

The first half of this chapter focused on water acquisition activities, providing an overview of the types of water used (including sources, quality, and provisioning), water use per well, and water use and consumption at the national, state, and county scale. The three major types of water used for hydraulic fracturing are surface water, groundwater, and reused hydraulic fracturing wastewater. Because trucking can be a major expense, operators tend to use water sources as close to the well pad as possible. Operators usually self-supply surface water or groundwater directly, but may also obtain water from public water systems or other suppliers. Hydraulic fracturing operations in the eastern United States rely predominantly on surface water, whereas operations in more semi-arid to arid western states use either surface water or groundwater. There are areas of the country that rely entirely on groundwater supplies (e.g., western Texas).

Reuse of wastewater reduces the demand on fresh water sources, which currently supply the vast majority of water used for hydraulic fracturing. The proportion of the water used in hydraulic fracturing that comes from reused hydraulic fracturing wastewater is generally low; in a survey of literature values from 10 states, basins, or plays, we found a median value of 5%, with this percentage varying by location (Table 4-2).[1] Available data on reuse trends indicate increasing reuse of wastewater over time in both Pennsylvania and West Virginia, likely due to the lack of nearby disposal options in Class II wells. Reuse as a percentage of water injected is typically lower in other areas of the United States, likely in part because of the availability of disposal wells; see Chapter 8 for more information.

The median amount of water used nationally per hydraulically fractured well was approximately 1.5 million gal (5.7 million L) in 2011 through early 2013 based on the EPA analysis of FracFocus disclosures (U.S. EPA, 2015b, c). This increased to approximately 2.7 million gal (10.2 million L) in 2014, driven by a proportional increase in horizontal wells (estimated from data in Gallegos et al., 2015). These national estimates represent a variety of fractured well types, including types requiring much less water per well than horizontal shale gas wells. Thus, published estimates for horizontal shale gas wells are typically higher (e.g., approximately 4 million gal (15 million L) per well (Vengosh et al., 2014), and should not be applied to all fractured wells to derive national estimates. There was also wide variation within and among states and basins in the median per well water volumes reported in 2011 and 2012, from more than 5 million gal (19 million L) in Arkansas and Louisiana to less than 1 million gal (3.8 million L) in Colorado, Wyoming, Utah, New Mexico, and California (U.S. EPA, 2015c). This variation can result from several factors, including geologic formation, well length, and fracturing fluid formulation.

Hydraulic fracturing uses billions of gallons of water every year at the national and state scales, and even in some counties. When expressed relative to total water use or consumption at these scales, however, hydraulic fracturing generally accounts for only a small percentage, usually less than 1%. These percentages are higher though in specific counties. Annual hydraulic fracturing water use was 10% or more compared to 2010 total water use in 6.5% of counties with FracFocus disclosures in 2011 and 2012 in the EPA FracFocus 1.0 project database, 30% or more in 2.2% of counties, and 50% or more in 1.0% of counties (Appendix Table B-2). Consumption estimates follow the same pattern, with higher percentages in each category: hydraulic fracturing water consumption was 10%, 30%, and 50% or more of 2010 total water consumption in 13.5%, 6.2%, and 4.0% of counties with FracFocus disclosures in the EPA FracFocus 1.0 project database (Appendix Table B-2). Thus, hydraulic fracturing represents a relatively large user and consumer of water in these counties.

Whether water quantity or quality impacts occur from water acquisition for hydraulic fracturing depends on the local balance between water withdrawals and availability. From our survey of the literature and our county level assessments, southern and western Texas appear to have the

[1] Note that reused water as a percentage of total water injected differs from the percentage of wastewater that is reused. See Section 4.2 and Chapter 8 for more information.

highest potential for impacts of the areas assessed in this chapter, given the combination of high hydraulic fracturing water use, relatively low water availability, intense periods of drought, and reliance on declining groundwater resources; see Section 4.6.2 on Factors below. Importantly, our results do not preclude the possibility of local water impacts in areas with comparatively lower potential, nor do they necessarily mean impacts have occurred in the high potential areas. Our survey, however, provides an indicator of areas with higher *potential* for impacts, and could be used to target resources or future studies.

In two example cases, local impacts to drinking water resources occurred in areas with increased hydraulic fracturing activity. In a detailed case study, Scanlon et al. (2014b) observed generally adequate water supplies for hydraulic fracturing in the Eagle Ford play in southern Texas, except in specific locations. They found excessive drawdown of groundwater locally, with estimated declines of ~100-200 ft (30-60 m) in a small proportion of the play (~6% of the area) after hydraulic fracturing activity increased in 2009. In 2011, drinking water wells in an area overlapping with the Haynesville Shale ran out of water due to higher than normal groundwater withdrawals and drought (LA Ground Water Resources Commission, 2012). Hydraulic fracturing water withdrawals contributed to these conditions, along with other water users and the lack of precipitation. By contrast, two EPA case studies in the Upper Colorado and the Susquehanna River Basins found minimal impacts from hydraulic fracturing withdrawals currently (U.S. EPA, 2015e) (Sections 4.5.2, 4.5.3).

These site-specific findings emphasize the need to focus on regional and local dynamics when considering the impacts from hydraulic fracturing water withdrawals. The case studies and the scientific literature as a whole suggest some common factors that increase or decrease the frequency or severity of impacts. These are summarized in the section below.

4.6.2 Factors Affecting Frequency or Severity of Impacts

The potential for impacts depends on the combination of water withdrawals and water availability at a given withdrawal location. Where water withdrawals are relatively low compared to water availability, impacts are unlikely to occur. Where water withdrawals are relatively high compared to water availability, impacts are more likely.

Areas reliant on declining groundwater are particularly vulnerable to more frequent and severe impacts from cumulative water withdrawals, including withdrawals for hydraulic fracturing. Groundwater recharge rates can be extremely low, and groundwater pumping is exceeding recharge rates in many areas of the country (Konikow, 2013). When pumping exceeds recharge, the cumulative effects of withdrawals are manifested in declining water levels. For this reason, water levels in many aquifers in the United States have declined substantially over the last century (Konikow, 2013). Cumulative drawdowns can affect surface water bodies since groundwater can be the source of base flow in streams (Winter et al., 1998), and alter groundwater quality by mobilizing chemicals from geologic sources, among other means (DeSimone et al., 2014; Alley et al., 1999). Although in many of these areas (e.g., the Ogallala aquifer), irrigated agriculture is the dominant user of groundwater, hydraulic fracturing withdrawals now also contribute to declining groundwater levels. Hydraulic fracturing groundwater consumption, for example, exceeds estimated recharge rates in the seven most active hydraulic fracturing counties in the Eagle Ford Shale in southern Texas (Steadman et al., 2015). When necessary, state and local governments have encouraged or mandated industry to use surface water over groundwater, as evidenced in both Louisiana and North Dakota.

Among surface water sources, smaller streams, even in humid areas, are more vulnerable to frequent and severe impacts from withdrawals. A detailed EPA case study found that streams with the smallest contributing areas in northeastern Pennsylvania were particularly vulnerable to withdrawals (U.S. EPA, 2015e). Protecting smaller streams from excessive withdrawals is probably most important for aquatic life, but may also protect drinking water quantity and quality in certain instances.

Seasonal or long-term drought can also make impacts more frequent and severe for surface water and groundwater sources. Hot, dry weather depletes surface water bodies and reduces or prevents groundwater recharge, while water demand often increases simultaneously (e.g., for irrigation). The EPA case study in Pennsylvania found that even large streams could be vulnerable to withdrawals during times of low flows (U.S. EPA, 2015e). Much of the western United States has experienced prolonged periods of drought over the last decade (Figure 4-8). This dynamic will likely be magnified by future climate change in certain locations (Meixner et al., 2016).

By contrast to the above factors, consumption of water for hydraulic fracturing does not appear to substantially influence the frequency or severity of impacts. There are concerns that hydraulic fracturing permanently removes water from the hydrologic cycle, posing a threat to long-term water supplies. Since impacts occur locally and depend on the local water balance, impacts can occur regardless of whether the water is withdrawn and returned to the larger hydrologic cycle elsewhere or whether it is permanently sequestered underground. We acknowledge that whether the water is returned to the larger hydrologic cycle may make a difference for the water budget of a larger area, such as on the state, regional, or national scale. For example, water converted to steam during thermoelectric cooling in one location may condense and fall as precipitation in an adjacent state or region. At these larger scales, however, hydraulic fracturing water consumption is a very small fraction of total water availability.[1] Plus, at these scales, there are other larger factors that can affect regional water budgets, but which are out of scope for this assessment.[2] For these reasons, focusing on consumption distracts from the more salient issue that impacts depend upon the spatial and temporal balance between local water withdrawals and availability.

[1] For example, hydraulic fracturing used approximately 3.3 billion gal (12.5 billion L) of water on average annually in all Colorado counties with hydraulic fracturing activities combined according to FracFocus disclosures in 2011 and 2012 (Appendix B-1). Using the consumption rate of 82.5% yields a consumption estimate of approximately 2.7 billion gal (10.2 billion L). This would be approximately 0.1% of the fresh water and total water availability metrics used in Textbox 4-2 for all of those same counties combined (approximately 2.6 trillion gal (9.8 trillion L) of fresh water and total water available).

[2] The combustion of methane produced by hydraulic fracturing, for example, adds water molecules to the environment, and at large scales, this may affect regional water budgets. However, quantifying this is outside the scope of this assessment. Similarly, there are other larger factors (e.g., water used for cooling thermoelectric power plants) that can affect regional water budgets, but these are also outside the scope of this assessment.

There are also factors that can decrease the frequency and severity of any impacts from water withdrawals. The literature suggests that water management, particularly wastewater reuse, the use of brackish groundwater, the use of passby flows, and transitioning from limited groundwater sources to more abundant surface water sources can reduce impacts. Reuse is not a universal solution, since in many areas of the country wastewater volumes from one well are often a small percentage of the water needed to fracture the next well. In the Marcellus Shale, for instance, 100% reuse of the wastewater produced from one well means reducing fresh water demand by 10 or 30% for the next (Section 4.2.1; Chapter 7). Nevertheless, reuse can be an important local factor reducing fresh water demand.

Switching to brackish water is another means by which fresh water demand can be—and is in some locations—reduced. This is a source of alternative water in western and southern Texas, for example. In these areas, use of brackish water is currently reducing impacts to fresh water sources, and could with continued use reduce future impacts (Scanlon et al., 2014b; Nicot et al., 2012). Our county level estimates suggest that brackish water could readily meet the volume demanded by hydraulic fracturing in Texas.

Water management also includes passby flows, a low stream flow threshold below which withdrawals are not allowed. Evidence suggests passby flows can be effective in protecting streams from hydraulic fracturing water withdrawals (U.S. EPA, 2015e). Finally, as evidenced by examples in both North Dakota and Louisiana, water management may include transitioning from declining groundwater sources to surface water, if available.

4.6.3 Uncertainties

There are several uncertainties inherent in our assessment of the potential impacts of water acquisition for hydraulic fracturing. The largest uncertainties stem from the lack of literature and data on this subject at local scales. Because impacts occur at a given withdrawal point, our assessment could assess the potential for impacts, but often could not determine if potential impacts were realized in the absence of local data. The exceptions were local case studies from the Eagle Ford play in Texas, the Upper Colorado River Basin in Colorado, and the Susquehanna River Basin in Pennsylvania. Moreover, it is also not clear if local impacts, for example a drinking water well going dry, are likely to be documented in the scientific literature.

Other uncertainties arise from data limitations on the volume and types of water used or consumed for hydraulic fracturing, future water use projections, and water availability estimates. There are no nationally consistent data sources, and therefore, water use estimates must be based on multiple, individual pieces of information. For example, in their National Water Census, the USGS includes hydraulic fracturing in the broader category of "mining" water use, but hydraulic fracturing water use is not reported separately (Maupin et al., 2014). There are locations where average annual hydraulic fracturing water use in 2011 and 2012 in the EPA FracFocus 1.0 project database exceeded total mining water use in 2010, and one county where it exceeded all water use (U.S. EPA, 2015c; Maupin et al., 2014). This could be due to a rapid increase in hydraulic fracturing water use, differences in methodology between the two databases (i.e., the USGS 2010 National Water Census and the EPA FracFocus 1.0 project database), or both.

We used the EPA FracFocus 1.0 project database for water use estimates, which itself has limitations. Many states in the project database did not require disclosure to FracFocus during the time period analyzed (U.S. EPA, 2015b). We conclude that this likely does not change the overall hydraulic fracturing water use patterns observed across the United States (Text Box 4-1), but could affect particular county level estimates. Also, the database covered the time period of 2011 through early 2013. Thus, changes in the industry since then are not reflected in these data.

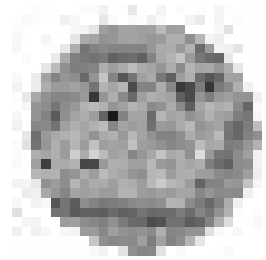
Hydraulic fracturing water use data that are often provided as water use associated with a particular well. While this is valuable information, the potential impacts of water acquisition for hydraulic fracturing could be better assessed if data were also available at the withdrawal point. If the total volume, date, location, and type (i.e., surface water or groundwater; and fresh, brackish, or reused wastewater) of each water withdrawal were documented, effects on availability could be better estimated. For example, surface withdrawal points could be aggregated by watershed or aquifer to estimate effects on downstream flow, groundwater levels, and water quality. Some of this information is available in disparate forms, but the lack of nationally consistent data on water withdrawal locations, timing, and amounts—data that are publicly available, easy to access and analyze—limits our assessment of potential impacts. The Susquehanna River Basin Commission collects this type of detailed data on hydraulic fracturing water withdrawals, but this type of information is not widely available across the nation.

Future hydraulic fracturing water use is also a source of uncertainty. Because water withdrawals and potential impacts are concentrated in certain localized areas, water use projections need to match this scale. Projections are available for Texas at the county scale, but more information at the county or sub-county scale is needed in other states with hydraulic fracturing activity and water availability concerns (e.g., northwest North Dakota, eastern Colorado). Due to a lack of data, we generally could not assess future water use and the potential for impacts in most areas of the country, nor could we examine these in combination with other relevant factors (e.g., climate change or population growth).

4.6.4 Conclusions

With notable exceptions, hydraulic fracturing uses and consumes a relatively small percentage of water when compared to total use, consumption, and availability at the national, state, and county scale. Despite this, impacts on drinking water resource quantity and quality from hydraulic fracturing water acquisition can occur at the local scale, because hydraulic fracturing water withdrawals are often concentrated in space and time, and impacts depend upon the local balance between withdrawals and availability. In two example cases, local impacts to drinking water resource quantity occurred in areas with increased hydraulic fracturing activity (e.g., in Texas's Eagle Ford play, and in Louisiana's Haynesville Shale). Declining groundwater resources, especially in the western United States, are particularly vulnerable to withdrawals, as are smaller streams, even in the more humid East. Finally, there are factors that increase or decrease the frequency and severity of impacts—included in this are times of low water availability, such as during drought, which can increase the frequency and severity of impacts, or conversely water management practices (e.g., shifting to brackish water, or passby flows), which can help protect drinking water resources.

Chapter 5. Chemical Mixing



Abstract

This chapter provides an analysis of the potential impacts on drinking water resources during the chemical mixing stage of the hydraulic fracturing water cycle and the factors governing the frequency and severity of these impacts. The chemical mixing stage includes the mixing of base fluid (90% to 97% by volume, typically water), proppant (2% to 10% by volume, typically sand), and additives (up to 2% by volume) on the well pad to make hydraulic fracturing fluid. This fluid is engineered to create and extend fractures in the targeted formation and to carry proppant into the fractures. Concentrated additives are delivered to the well pad and stored on site, often in multiple, closed containers, and moved around the well pad in hoses and tubing.

Changes in drinking water quality can occur if spilled fluids reach groundwater or surface water resources. In this assessment, a spill is considered to be any release of fluids. The EPA's analysis found that spills and releases of chemicals and fluids have occurred during the chemical mixing stage and have reached soil and surface water receptors. Spills of hydraulic fracturing fluids or additives included in the analysis had a median spill volume of 420 gal (1,590 L), with a range of 5 to 19,320 gal (9 to 72,130 L). Spills were caused most often by equipment failure or human error. The potential for spilled fluids to reach, and therefore impact, groundwater or surface water resources depends on the composition of the spilled fluid, spill characteristics, spill response activities, and the fate and transport of the spilled fluid.

The movement of spilled hydraulic fracturing fluids and chemicals through the environment is difficult to predict, because spills are site- and chemical-specific, and because hydraulic fracturing-related spills are typically complex mixtures of chemicals. Physicochemical properties, which depend on the molecular structure of a chemical, govern whether spilled chemicals volatilize, sorb, transform, and travel. Spill prevention practices and spill response activities can prevent spilled fluids from reaching ground or surface drinking water resources.

The severity of potential impacts on water quality from spills of additives or hydraulic fracturing fluids depends on the identity and amount of chemicals that reach ground or surface water resources, the hazards associated with the chemicals, and the characteristics of the receiving water body. The lack of monitoring following spills, along with the lack of publicly available information on the composition of additives and fracturing fluids, containment and mitigation measures in use, the proximity of chemical mixing to drinking water resources, and the fate and transport of spilled fluids limits the EPA's ability to fully assess potential impacts on drinking water resources and their frequency and severity. This chapter shows that spills of additives and hydraulic fracturing fluids during the chemical mixing stage of the hydraulic fracturing water cycle have occurred and have reached and impacted drinking water resources.

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5. Chemical Mixing

5.1 Introduction

This chapter provides an analysis of the potential impacts on drinking water resources during the chemical mixing stage of the hydraulic fracturing water cycle and the factors governing the frequency and severity of these impacts. Chemical mixing is a complex process that requires the use of specialized equipment and a range of different additives to produce the fluid that is injected into a well to fracture the formation. This fluid, the hydraulic fracturing fluid, generally consists of a base fluid (typically water), a proppant (typically sand), and additives (chemicals), although there is no standard or single composition of hydraulic fracturing fluid used. The number, type, and amount of chemicals used to create the hydraulic fracturing fluid vary from well to well based on site- and operator-specific factors. Spills may occur at any point in the chemical mixing process.[1] The potential for spilled fluids to reach, and therefore impact, ground or surface water resources depends on the composition of the spilled fluid, spill characteristics, spill response activities, and the fate and transport of the spilled fluid. This chapter is structured around these concepts.

The chapter starts by discussing the characteristics of hydraulic fracturing fluids (Sections 5.2 to 5.4). This includes an introductory overview of the chemical mixing process (Section 5.2), a description of the different components of the hydraulic fracturing fluid (Section 5.3), the range of different chemicals used and their classes, the most frequently used chemicals nationwide, and volumes used (Section 5.4).[2] (Appendix H provides a list of chemicals that the EPA identified as being used in hydraulic fracturing fluids.)

The chapter continues with a discussion on how chemicals are managed on the well pad, the characteristics of spills when they occur, and spill response activities (Sections 5.5 to 5.7). This includes a description on how potential impacts of a spill on drinking water resources depends upon chemical management practices, such as storage, on-site transfer, and equipment maintenance (Section 5.5). A summary analysis of reported spills and their common causes at hydraulic fracturing sites is then presented (Section 5.6). Then, there is a discussion on the different efforts of spill prevention, containment, and mitigation (Section 5.7).

Next, the fate and transport of spilled chemicals is discussed (Section 5.8). This section includes how a chemical can move through the environment and transform, and what governs exposure concentrations of chemicals in the environment. Due to the complexities of the processes and the site-specific and chemical-specific nature of spills, it is difficult to develop a full assessment of their fate and transport. This section provides a general overview and discusses how the fate and transport of a chemical depends on site conditions, environmental conditions, physicochemical

[1] In this assessment, a spill is considered to be any release of fluids. Spills can result from accidents, fluid management practices, or illegal dumping.

[2] Chemical classes are groupings of different chemicals based on similar features, such as chemical structure, use, or physical properties. Examples of chemical classes include hydrocarbons, alcohols, acids, and bases.

properties of the released chemicals, fluid composition, volume of the release, the proximity to a drinking water resource, and the characteristics of the drinking water resource that is the receptor.

Next is an overview of on-going changes in chemical use in hydraulic fracturing, with an emphasis on industry efforts to reduce potential impacts from surface spills by using fewer and safer chemicals (Section 5.9). The chapter concludes by providing a synthesis, including a summary of findings, factors that affect frequency and severity of potential impacts, and a discussion of uncertainties and data gaps (Section 5.10).

Due to the limitations of available data and the scope of this assessment, it is not possible to provide a detailed analysis of all of the factors listed above. Data limitations preclude a quantitative analysis of the likelihood or severity of chemical spills or impacts. Spills that occur off-site, such as those during transportation of chemicals to the site or storage of chemicals in staging areas, are out of the scope of this assessment. This chapter qualitatively characterizes the potential for impacts on drinking water resources given the current understanding of overall operations and specific components of the chemical mixing process.

5.2 Chemical Mixing Process

Understanding the chemical mixing process is necessary to understand how, why, and when spills might occur. This section provides a general overview of the chemical mixing stage of the hydraulic fracturing water cycle (Carter et al., 2013; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008). Figure 5-1 shows a hydraulic fracturing site during the chemical mixing process. In our discussion, we focus on the types of additives used at each phase of the process. While similar processes are used to fracture horizontal and vertical wells, a horizontal well treatment is described here. Horizontal well treatments are likely to be more complex and therefore illustrative of the variety of practices that have become more prevalent over time with advances in technology (Chapter 3). A water-based system is described, because water is the most commonly used base fluid, appearing in more than 93% of FracFocus 1.0 disclosures between January 1, 2011 and February 28, 2013 (U.S. EPA, 2015a).[1] While the number and types of additives may vary widely, the basic chemical mixing process and the on-site layout of hydraulic fracturing equipment are similar across sites (BJ Services Company, 2009). Equipment used in the chemical mixing process typically consists of chemical storage trucks, water supply tanks, proppant supply, slurry blenders, a number of high-pressure pumps, a manifold, surface lines and hoses, and a central control unit. Detailed descriptions of specific additives and the equipment used in the process are provided in Sections 5.3 and 5.5, respectively.

[1] FracFocus (www.fracfocus.org) is a registry of information of water and chemical use in wells in which hydraulic fracturing is conducted. More details are provided in Text Box 5-1.

Figure 5-1. Representative hydraulic fracturing site showing equipment used on-site during the chemical mixing process.
The frac well head is located in the center bottom (green), the manifold runs down the middle, and high pressure pumps lead into the manifold from either side. Source: Schlumberger.

At a newly-drilled production well, the chemical mixing process begins after the drilling, casing, and cementing processes are finished and hydraulic fracturing equipment has been set up and connected to the well. The process can generally be broken down into one or more sequential stages with specific chemicals added at different phases during each stage phase to achieve a specific purpose (Knappe and Fireline, 2012; Fink, 2003). The process for water-based hydraulic fracturing is described in Figure 5-2 below.

The first phase is the cleaning and preparation of the well. The fluid used in this phase is often referred to as the pre-pad fluid, pre-pad volume, or spearhead. Acid is typically the first chemical introduced. Acid, with a concentration of 3% to 28% (by volume, typically hydrochloric acid, HCl), is used to clean any cement left inside the well from cementing the casing and dissolve any pieces of rock that may remain in the well that could block the perforations.[1] Acid is typically pumped directly from acid storage tanks or tanker trucks, without being mixed with other additives. The first, or pre-pad, phase may also involve mixing and injection of additional chemicals to facilitate the flow of fracturing fluid introduced in the next phase of the process. These additives may include biocides, corrosion inhibitors, friction reducers, and scale inhibitors (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

[1] Prior to the injection of the pad fluid, for wells that are cased in the production zone, the well casing is typically perforated to provide openings through which the pad fluid can enter the formation. A perforating gun is typically used to create small holes in the section of the well being fractured. The perforating gun is lowered into position in the horizontal portion of the well. An electrical current is used to set off small explosive charges in the gun, which creates holes through the well casing and out a short distance into the formation (Gupta and Valkó, 2007).

Figure 5-2. Overview of a chemical mixing process of the hydraulic fracturing water cycle.
This figure outlines the chemical mixing process for a generic water-based hydraulic fracture of a horizontal well. The chemical mixing phases outline the steps taken at the surface in the overall fracturing job, while the hydraulic fracturing stages outline how each section of the horizontal well would be fractured beginning with the toe of the well, shown on left-side. The proppant gradient represents how the proppant size may change within each stage of fracturing as the fractures are elongated. The chemical mixing process is repeated depending on the number of stages used for a particular well. The number of stages is determined in part by the length of the horizontal leg. In this figure, four stages are represented, but typically, a horizontal fracturing treatment would consist of 10 to 20 stages per well (Lowe et al., 2013). Fracturing has been reported to be done in as many as 59 stages (Pearson et al., 2013).

In the second phase, a hydraulic fracturing fluid, typically referred to as the pad or pad volume, is mixed, blended, and pumped down the well under high pressure to create fractures in the formation.[1] The pad is a mixture of base fluid, typically water, and additives and is designed to create, elongate, and enlarge fractures in the targeted geologic formation when injected under high pressure (Gupta and Valkó, 2007) (see Section 6.3 for additional information on fracture growth following injection). A typical pad consists of, at minimum, a mixture of water and friction reducer. A typical pad consists of, at minimum, a mixture of water and friction reducer. Other additives (see U.S. EPA (2015a) and Table 5-1) may be used to facilitate flow and kill bacteria (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008). The pad is pumped into the formation through perforations or sliding sleeves in the well casing.

[1] In terms of chemical mixing, "pad" is a term used to describe hydraulic fracturing fluid without solid at the start of the fracturing of the formation. In terms of the entire hydraulic fracturing process, the "well pad" or "pad" is the area of land where drilling occurs.

In the third phase, proppant, typically sand, is mixed into the hydraulic fracturing fluid. The proppant volume, as a proportion of the injected fluid, is increased gradually until the desired concentration in the fractures is achieved. Gelling agents, if used, are also mixed with the proppant and base fluid in this phase to increase the viscosity to help carry the proppant. Additional chemicals may be added to gelled fluids, initially to maintain viscosity and later to break down the gel and decrease viscosity, so the hydraulic fracturing fluid can more readily flow back out of the formation and through the well to facilitate production from the fractured formation (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

A final flush or clean-up phase may be conducted after the stage is fractured, with the primary purpose of maximizing well productivity. The flush is a mixture of water and additives that work to aid the placement of the proppant, clean out the chemicals injected in previous phases, and prevent microbial growth in the fractures (Knappe and Fireline, 2012; Fink, 2003).

The second, third, and fourth phases are repeated multiple times in a well with multi-stage hydraulic fracturing. For each stage, the well is typically perforated and fractured beginning at the end, or toe, of the well and proceeding backwards toward the bend or heel of the well, near the vertical section. In vertical wells, stages typically begin in deeper portions of the well and proceed shallower. Each fractured stage is isolated before the next stage is fractured. The number of stages sets how many times the chemical mixing process is repeated at the site surface (Figure 5-2). The number of stages increases with longer intervals of the well subjected to hydraulic fracturing (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

The number of stages per well can vary, with several sources suggesting between 10 and 20 stages is typical (GNB, 2015; Lowe et al., 2013).[1] The full range reported in the literature is much wider, with one source documenting between 1 and 59 stages per well (Pearson et al., 2013) and others reporting values within this range (NETL, 2013; STO, 2013; Allison et al., 2009). The number of stages per well seems to have increased over time. One study reports that the average number of stages per horizontal well rose from approximately 10 in 2008 to 30 in 2012 (Pearson et al., 2013). As more stages are used, the total volume of hydraulic fracturing fluid and chemicals increase. This increases the potential, frequency, and severity of surface spills associated with chemical mixing and thus potential impacts on drinking water resources.

In each of these phases, water is usually the primary component of the hydraulic fracturing fluid, though the exact composition of the fluid injected into the well changes over the duration of each stage. In water-based hydraulic fracturing, the composition, by volume, of a typical hydraulic fracturing fluid is 90% to 97% water, 2% to 10% proppant, and 2% or less additives (Carter et al., 2013; Knappe and Fireline, 2012; SWN, 2011).[2]

[1] The number of stages has been reported to be 6 to 9 in the Huron in 2009 (Allison et al., 2009), 13 to 32 in the Marcellus (NETL, 2013), and up to 40 by STO (2013).

[2] This range is based on a compilation of sources. Sources present compositions as by mass, by volume, or without specificity. Because of non-additive volumes, the composition by volume can be different before and after mixing. By mass: 90% water, 8-9% proppant, 0.5 to 1.5% additives (Knappe and Fireline, 2012); 88% water, 11% proppant, <1%

5.3 Overview of Hydraulic Fracturing Fluids

Hydraulic fracturing fluids are formulated to perform specific functions: create and extend the fracture and transport and place the proppant in the fractures (Montgomery, 2013; Spellman, 2012; Gupta and Valkó, 2007).[1] The hydraulic fracturing fluid generally consists of three parts: (1) the base fluid, which is the largest constituent by volume, (2) the additives, and (3) the proppant. Additives, which can be a single chemical or a mixture of chemicals, are chosen to serve a specific purpose in the hydraulic fracturing fluid (e.g., friction reducer, gelling agent, crosslinker, biocide) (Spellman, 2012). Throughout this chapter, "chemical" is used to refer to an individual chemical substance (e.g., methanol, petroleum distillates).[2] Proppants are small particles, usually sand, mixed with fracturing fluid to hold fractures open so that the target hydrocarbons can flow from the formation through the fractures and up the wellbore. The combination of additives, and the mixing and injection process, varies based on a number of factors as discussed below. The additive combination determines the amount and type of equipment required for storage and, therefore, contributes to the determination of the potential for spills and impacts of those spills.

The particular composition of a hydraulic fracturing fluid is designed based on empirical experience, the geology and geochemistry of the production zone, economics, goals of the fracturing process, availability of the desired chemicals, and preference of the service company or operator (Montgomery, 2013; ALL Consulting, 2012; Klein et al., 2012; Ely, 1989).[3] No single set of specific chemicals is used at every site. Multiple types of fracturing fluids may be appropriate for a given site, and any given type of fluid may be appropriate at multiple sites. For the same type of fluid formulation, there can be differences in the additives, chemicals in those additives, and the concentrations selected. There are broad criteria for hydraulic fracturing fluid selection based on the targeted production zone temperature, pressure, water sensitivity, and permeability (Gupta and Valkó, 2007; Elbel and Britt, 2000). Figure 5-3 provides a general overview of the types of decisions to determine which fluid can be used for different situations. Similar fluids may be appropriate for different formations. For example, crosslinked fluids with 25% nitrogen foam (titanate or zirconate crosslink + 25% nitrogen) can be used in both gas and oil wells with high temperatures and

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additive (as median maximum concentration) (U.S. EPA, 2015a), 94% water, 6% proppant, <1% additive (Sjolander et al., 2011), 88% water, 11% proppant, <1% additive (OSHA, 2014a, b). By volume: 95% water, 5% proppant, <1% additive (before mixing), 97% water, 2% proppant, <1% additive (after mixing) (Sjolander et al., 2011), 90% water, 10% proppant, <1% additive (before mixing), 95% water, 5% proppant, <1% additive (after mixing) (OSHA, 2014a, b), 98-99.5%, water and sand 0.5 to 2% additives (Spellman, 2012). Not specified: 99.9% water and sand, 0.1% chemicals (SWN, 2011), 98-99% water and proppant, 1-2 % additives (Carter et al., 2013).

[1] We use "hydraulic fracturing fluid" to refer to the fluid that is injected into the well and used to create and hold open fractures the formation.

[2] In this chapter, because of the way many chemicals are reported, we use the word "chemical" to refer to any individual chemical or chemical substance that has been assigned a CASRN (Chemical Abstracts Service Registry Number). A CASRN is a unique identifier for a chemical substance, which can be a single chemical (e.g., hydrochloric acid, CASRN 7647-01-0) or a mixture of chemicals (e.g., hydrotreated light petroleum distillates (CASRN 64742-47-8), a complex mixtures of C9 to C16 hydrocarbons). For simplicity, we refer to both pure chemicals and chemical substances that are mixtures, which have a single CASRN, as "chemicals."

[3] Empirical experience tends to provide better result as operators gain experience at a new site or geology increases. When an operator moves to a new basin geology, there may be less than optimal results. With experience and understanding of the geology increases, the empirical evidence will inform what hydraulic fracturing fluid composition works better than others.

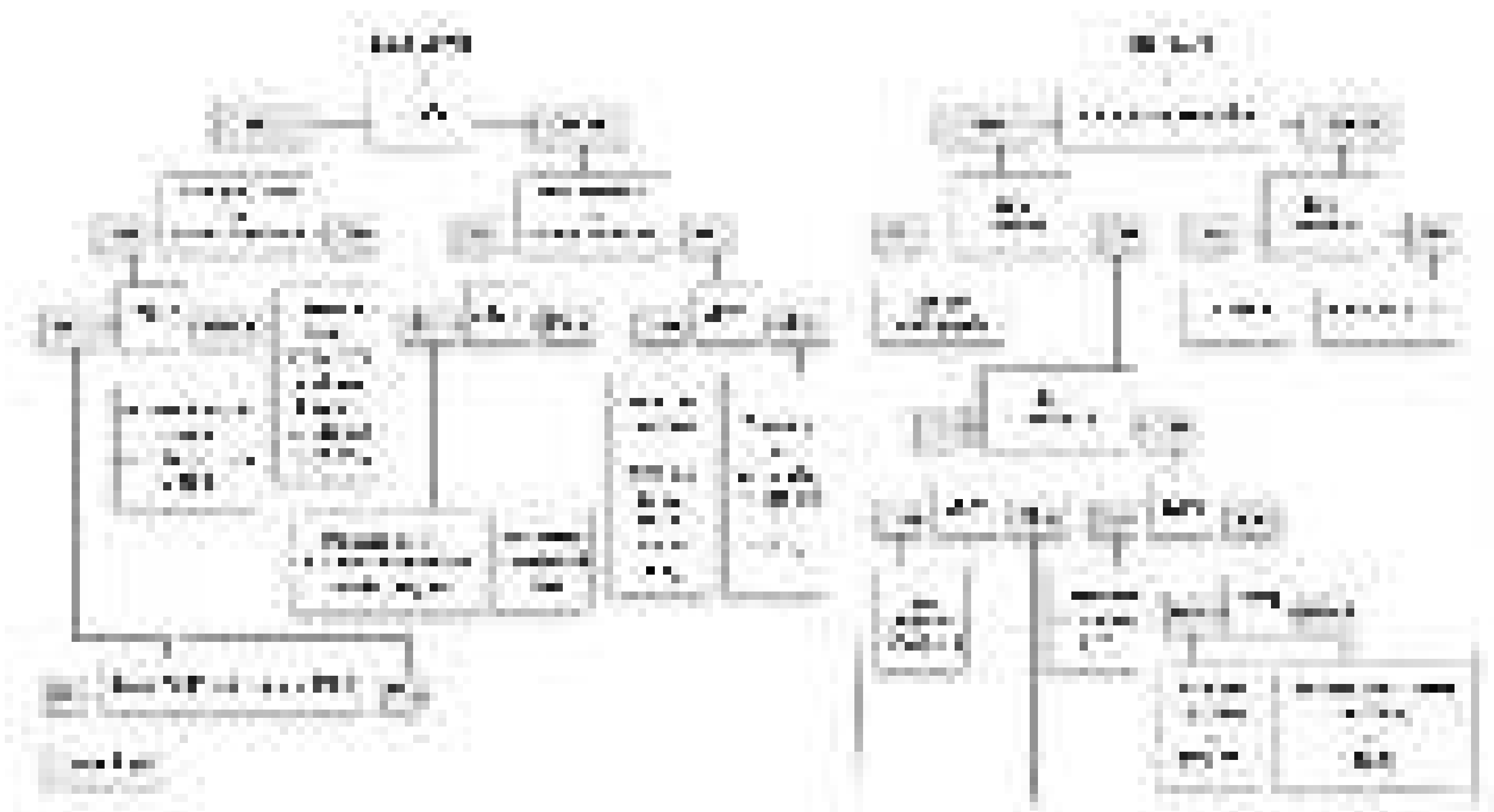

Figure 5-3. Example hydraulic fracturing fluid decision tree for gas and oil wells.
This decision tree figure serves as an example of the factors that determine the type of hydraulic fracturing fluid chosen to fracture a given formation, depending on whether the well will produce oil or gas. Factors include water sensitivity, formation temperature, and pressure. HPG is hydroxypropylguar, guar derivatized with propylene oxide. Parameters are: k_f, fracture permeability, w is the fracture width, and x_f is the fracture half-length. This figure was chosen to represent the differences between oil and gas wells and the types of decisions involved with choosing a fluid. This is adapted from Elbel and Britt (2000) and, as such, is dated to that time period. Since then, slickwater has become increasingly popular due to its simplicity and cheaper cost, and slickwater has often replaced linear and crosslinked gelled fluids, especially in shales. Other decision tree figures may exist.

variation in water sensitivity.[1,2] One of the most important properties in designing a hydraulic fracturing fluid is the viscosity (Montgomery, 2013).[3]

Table 5-1 provides a list of common types of additives, their functions, and the most frequently used chemicals for each purpose based on the EPA's analysis of disclosures to FracFocus 1.0 (U.S. EPA, 2015a, hereafter referred to as the EPA FracFocus 1.0 report), the EPA's project database of disclosures to FracFocus 1.0 (U.S. EPA, 2015c, hereafter referred to as the EPA FracFocus 1.0 project database), and other literature sources.[4] Additional information on more additives can be found in U.S. EPA (2015a).

A general description of typical hydraulic fracturing fluid formulations nationwide is difficult, because fracturing fluids vary from well to well. Based on the EPA FracFocus 1.0 report, the median number of chemicals reported for each disclosure was 14, with the 5th to 95th percentile ranging from four to 28 (see Appendix H for a list of hydraulic fracturing fluid chemicals). The median number of chemicals per disclosure was 16 for oil wells and 12 for gas wells (U.S. EPA, 2015a). Other sources have stated that between three and 12 additives and chemicals are used (Schlumberger, 2015; Carter et al., 2013; Spellman, 2012; GWPC and ALL Consulting, 2009).[5]

Water, the most commonly used base fluid for hydraulic fracturing, is inferred to be used as a base fluid in more than 93% of EPA FracFocus 1.0 disclosures (U.S. EPA, 2015c). Alternatives to water-based fluids, such as hydrocarbons and gases, including carbon dioxide and nitrogen-based foam, may also be used based on formation characteristics, cost, or preferences of the well operator or service company (ALL Consulting, 2012; GWPC and ALL Consulting, 2009). Non-aqueous base fluid ingredients were identified in 761 (2.2%) of EPA FracFocus 1.0 disclosures (U.S. EPA, 2015a). Gases and hydrocarbons may be used alone or blended with water; more than 96% of the disclosures identifying non-aqueous base fluids are blended (U.S. EPA, 2015a). There is no standard method to categorize the different fluid formulations (Patel et al., 2014; Montgomery, 2013; Spellman, 2012; Gupta and Valkó, 2007). Therefore, we broadly categorize the fluids as water-based or alternative fluids.

[1] A crosslinked fluid is a fluid that has polymers that have been linked together through a chemical bond. A crosslink chemical is added to have the polymer chains linked together to form larger chemical structures with higher viscosity. The increased fracturing fluid viscosity allows the fluid to carry more proppant into the fractures. The fracturing fluid remains viscous until a breaking agent is introduced to break the cross-linked polymer.

[2] Water sensitivity refers to when a formation's physicochemical properties are affected in the presence of water. An example of a water sensitive formation would be one where the soil particles swell when water is added, reducing the permeability of the formation.

[3] Viscosity is a measure of the internal friction of fluid that provides resistance to shear within the fluid, informally referred to as how "thick" a fluid is. For example, custard is thick and has a high viscosity, while water is runny with a low viscosity. Sufficient viscosity is needed to create a fracture and transport proppant (Gupta and Valkó, 2007). In lower-viscosity fluids, proppant is transported by turbulent flow and requires more hydraulic fracturing fluid. Higher-viscosity fluids allows the fluid to carry more proppant, requiring less fluid but necessitating the reduction of viscosity after the proppant is placed (Rickman et al., 2008; Gupta and Valkó, 2007).

[4] A disclosure refers to all data submitted for a specific oil and gas production well for a specific fracture date.

[5] Sources may differ based on whether they are referring to additives or chemicals.

Table 5-1. Examples of common additives, their function, and the most frequently used chemicals reported to FracFocus for these additives.

The list of examples of common additives was developed from information provided in multiple sources (U.S. EPA, 2015a, c; Stringfellow et al., 2014; Montgomery, 2013; Vidic et al., 2013; Spellman, 2012; GWPC and ALL Consulting, 2009; Arthur et al., 2008; Gupta and Valkó, 2007; Gidley et al., 1989). The additive functions are based on information the EPA received from service companies (U.S. EPA, 2013a).

Additives	Function	Chemicals reported in 20% or more of disclosures in the EPA FracFocus 1.0 project database for given additive[a,b]
Acid	Dissolves cement, minerals, and clays to reduce clogging of the pore space	Hydrochloric acid
Biocide	Controls or eliminates bacterial growth, which can be present in the base fluid and may have detrimental effects on the long term well productivity	Glutaraldehyde; 2,2-dibromo-3-nitrilopropionamide
Breaker	Reduces the designed increase in viscosity of specialized treatment fluids such as gels and foams after the proppant has been placed and flowback commences to clean up the well	Peroxydisulfuric acid diammonium salt
Clay control	Prevents the swelling and migration of formation clays that otherwise react to water-based fluids	Choline chloride
Corrosion inhibitor	Protects the iron and steel components in the wellbore and treating equipment from corrosive fluids	Methanol; propargyl alcohol; isopropanol
Crosslinker	Increases the viscosity of base gel fluids by connecting polymer molecules	Ethylene glycol; potassium hydroxide; sodium hydroxide
Emulsifier	Facilitates the dispersion of one immiscible fluid into another by reducing the interfacial tension between the two liquids to achieve stability	2-Butoxyethanol; polyoxyethylene(10)nonylphenyl ether; methanol; nonyl phenol ethoxylate
Foaming agent	Generates and stabilizes foam fracturing fluids	2-Butoxyethanol; Nitrogen, liquid; isopropanol; methanol; ethanol
Friction reducer	Reduces the friction pressures experienced when pumping fluids through tools and tubulars in the wellbore	Hydrotreated light petroleum distillates
Gelling agent	Increases fracturing fluid viscosity allowing the fluid to carry more proppant into the fractures and to reduce fluid loss to the reservoir	Guar gum; hydrotreated light petroleum distillates
Iron control agent	Controls the precipitation of iron compounds (e.g., Fe_2O_3) from solution	Citric acid

Additives	Function	Chemicals reported in 20% or more of disclosures in the EPA FracFocus 1.0 project database for given additive[a,b]
Nonemulsifier	Separates problematic emulsions generated within the formation	Methanol; isopropanol; nonyl phenol ethoxylate
pH control	Affects the pH of a solution by either inducing a change (pH adjuster) or stabilizing and resisting change (buffer) to achieve desired qualities and optimize performance	Carbonic acid, dipotassium salt; potassium hydroxide; sodium hydroxide; acetic acid
Resin curing agents	Lowers the curable resin coated proppant activation temperature when bottom hole temperatures are too low to thermally activate bonding	Methanol; nonyl phenol ethoxylate; isopropanol; alcohols, C12-14-secondary, ethoxylated
Scale inhibitor	Controls or prevents scale deposition in the production conduit or completion system	Ethylene glycol; methanol
Solvent	Controls the wettability of contact surfaces or prevents or breaks emulsions[1]	Hydrochloric acid

[a] Chemicals (excluding water and quartz) listed in the EPA FracFocus 1.0 project database in more than 20% of disclosures for a given purpose when that purpose was listed as used on a disclosure (U.S. EPA, 2015c). These are not necessarily the active ingredients for the purpose, but rather are listed as being commonly present for the given purpose. Chemicals may be disclosed for more than a single purpose (e.g., 2-butoxyethanol is listed as being used as an emulsifier and a foaming agent).

[b] Analysis considered 32,885 disclosures and 615,436 ingredient records that met selected quality assurance criteria, including: completely parsed (parsing is the process of analyzing a string of symbols to identify and separate various components); unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; valid concentrations; and valid purpose. Disclosures that did not meet quality assurance criteria (5,645) or other, query-specific criteria were excluded from analysis.

5.3.1 Water-Based Fracturing Fluids

The advantages of water-based fracturing fluids are low cost, ease of mixing, and ability to recover and reuse the water. The disadvantages are that they have low viscosity, they create narrow fractures, and they may not provide optimal performance in water-sensitive formations (Montgomery, 2013; Gupta and Valkó, 2007) (Section 5.3.2). Water-based fluids can be as simple as water with a few additives to reduce friction, such as "slickwater," or as complex as water with crosslinked polymers, clay control agents, biocides, and scale inhibitors (Spellman, 2012). (See Figure 5-4 for a slickwater example.)

Gels may be added to water-based fluids to increase viscosity, which assists with proppant transport and results in wider fractures. Gelling agents include natural polymers, such as guar, starches, and cellulose derivatives, which require the addition of biocide to minimize bacterial growth (Spellman, 2012; Gupta and Valkó, 2007). Gels may be linear or crosslinked. Crosslinking

[1] Wettability is the ability of a liquid to maintain contact with a solid surface. When wettability is high, a liquid droplet will lie flat across a surface, maximizing the area of contact between the liquid and the solid. When wettability is low, a liquid droplet will approach a spherical shape, minimizing the area of contact between the liquid and solid.

increases viscosity without adding more gel. Gelled fluids require the addition of a breaker, which breaks down the gel after it carries in the proppant, to reduce fluid viscosity to facilitate fluid flowing back after treatment (Spellman, 2012; Gupta and Valkó, 2007). The presence of residual breakers may make it difficult to reuse recovered water (Montgomery, 2013).

5.3.2 Alternative Fracturing Fluids

Alternative hydraulic fracturing fluids can be used for water sensitive formations (i.e., formations where permeability is reduced when water is added) or as dictated by production goals (Halliburton, 1988). Examples of alternative fracturing fluids include acid-based fluids; non-aqueous-based fluids; energized fluids, foams, or emulsions; viscoelastic surfactant fluids; gels; methanol; and other unconventional fluids (Montgomery, 2013; Saba et al., 2012; Gupta and Hlidek, 2009; Gupta and Valkó, 2007; Halliburton, 1988).

Acid fracturing is generally used in carbonate formations without the use of a proppant. Fractures are initiated with a hydraulic fracturing fluid, and acid (gelled, foamed, or emulsified) is added to irregularly etch the wall of the fracture. The etching serves to prop open the formation, for a high-conductivity fracture (Spellman, 2012; Gupta and Valkó, 2007).

Non-aqueous fluids, like petroleum distillates and propane, are used in water-sensitive formations. Non-aqueous fluids may also contain additives, such as gelling agents, to improve performance (Gupta and Valkó, 2007). The use of non-aqueous fluids has decreased due to safety concerns, and because water-based and emulsion fluid technologies have improved (Montgomery, 2013; Gupta and Valkó, 2007). Methanol, for example, was previously used as a base fluid in water-sensitive reservoirs beginning in the early 1990s, but was discontinued in 2001 for safety concerns and cost (Saba et al., 2012; Gupta and Hlidek, 2009; Gupta and Valkó, 2007). Methanol is still widely used as an additive or in additive mixtures in hydraulic fracturing fluid formulations.

Energized fluids, foams, and emulsions minimize fluid leakoff in low pressure targeted geologic formations, have high proppant-carrying capacity, improve fluid recovery, and are sometimes used in water-sensitive formations (Barati and Liang, 2014; Gu and Mohanty, 2014; Spellman, 2012; Gupta and Valkó, 2007; Martin and Valko, 2007).[1] However, these treatments tend to be expensive, can require high pressure, and pose potential health and safety concerns (Montgomery, 2013; Spellman, 2012; Gupta and Valkó, 2007). Energized fluids (see Figure 5-4 for an example of an energized fluid composition) are mixtures of liquid and gas (Patel et al., 2014; Montgomery, 2013). Nitrogen (N_2) or carbon dioxide (CO_2), the gases used, make up less than 53% of the fracturing fluid volume, typically ranging from 20% to 30% by volume (Montgomery, 2013; Gupta and Valkó, 2007; Mitchell, 1970). Energized foams are liquid-gas mixtures, with nitrogen or carbon dioxide gas comprising more than 53% of the fracturing fluid volume, with a typical range of 65% to 80% by volume (Montgomery, 2013; Mitchell, 1970). Emulsions are liquid-liquid mixtures, typically a

[1] Leakoff is the fraction of the injected fluid that infiltrates into the formation (e.g., through an existing natural fissure) and is not recovered during production (Economides et al., 2007). See Chapter 6, Section 6.3 for more discussion on leakoff.

hydrocarbon (e.g., condensate or diesel) with water.[1] Both water-based fluids, including gels, and non-aqueous fluids can be energized fluids or foams.

Foams and emulsions break easily using gravity separation and are stabilized by using additives such as foaming agents (Gupta and Valkó, 2007). Emulsions may be used to stabilize active chemical ingredients or to delay chemical reactions, such as the use of carbon dioxide-miscible, non-aqueous fracturing fluids to reduce fluid leakoff in water-sensitive formations (Taylor et al., 2006).

Other types of fluids not addressed above include viscoelastic surfactant fluids, viscoelastic surfactant foams, crosslinked foams, liquid carbon dioxide-based fluid, and liquid carbon dioxide-based foam fluid, and hybrids of other fluids (King, 2010; Brannon et al., 2009; Curtice et al., 2009; Tudor et al., 2009; Gupta and Valkó, 2007; Coulter et al., 2006; Boyer et al., 2005; Fredd et al., 2004; MacDonald et al., 2003).

5.3.3 Tracers

Some chemicals are added to the fluid to act as tracers. Tracers are added to hydraulic fracturing fluid to assess the efficiency of fracturing and proppant placement. As an example, the efficiency of oil production from multistage fracturing was assessed by using 17 oil soluble tracers. Each tracer was used to assess production from a specific interval of the well (Catlett et al., 2013), although the specific compounds used were not identified (Table 5-2). Chemical classes of tracers and individual examples show a range of compounds employed including both inorganic and organic, and including radioactive elements, although only a few specific chemicals have been revealed. Of these, examples are proppant tracers and fluorocarbons. Although radioactive fluids have also been used for proppant tracing, a commonly-used approach has the short-half-life elements Antimony^{124}, Iridium^{192}, and Scandium^{46} bound to the proppants and gamma emissions are subsequently measured by a neutron-logging device (Sonnenfield et al., 2016; Odegard et al., 2015; Lowe et al., 2013; Osborn and McIntosh, 2011; McDaniel et al., 2010).[2,3] Of the organic tracers, 14 fluorinated organics have been identified through an analysis of FracFocus 2.0 disclosures (Konschnik and Dayalu, 2016). Three fluorinated tracers and Antimony^{124} were identified in produced water (Maguire-Boyle and Barron, 2014) (Appendix Table H-4).

Table 5-2. Classes and specifically identified examples of tracers used in hydraulic fracturing fluids.

Class	Specific Chemical[a]	Reference
Thiocyanates (SCN^-)	ND	Dugstad (2007)
Fluorobenzoic acids	ND	Dugstad (2007)

[1] Diesel is a mixture typically of C8 to C21 hydrocarbons. The shorthand "C8" is used to represent a hydrocarbon with 8 carbons. Thus "C21" represents a hydrocarbon with 21 carbons. Octane has 8 carbons and is thus a C8, and is a component of gasoline.

[2] Antimony^{124}: 60.2 days, Iridium^{192}: 74 days, Scandium^{46}: 83.8 days.

[3] Gadolinium^{155} and Gadolinium^{157} have been suggested as bound proppant tracers because of their high-gamma-capture cross-sections (Liu et al., 2015).

Class	Specific Chemical[a]	Reference
Radioactive tracers	Tritiated Water	Dugstad (2007)
	Tritiated Methanol	Dugstad (2007)
	Antimony^{124}	Silber et al. (2003)
	Iridium^{192}	Silber et al. (2003)
	Scandium^{46}	Silber et al. (2003)
Fluorocarbons	2,2,3,3,4,4,4-heptafluorobutyl undecylate	Maguire-Boyle and Barron (2014)
	2,3,4-Trifluorobenzoic acid	Konschnik and Dayalu (2016)
	2,4,5-Trifluorobenzoic acid	Konschnik and Dayalu (2016)
	2,4-Difluorobenzoic acid	Konschnik and Dayalu (2016)
	2,6-Difluorobenzoic acid	Konschnik and Dayalu (2016)
	2-Chloro-4-fluorobenzoic acid	Konschnik and Dayalu (2016)
	2-Fluorobenzoic acid	Konschnik and Dayalu (2016)
	2-Trifluoromethylbenzoic acid	Konschnik and Dayalu (2016)
	3-Trifluoromethylbenzoate	Konschnik and Dayalu (2016)
	4-(Trifluoromethyl)benzoic acid	Konschnik and Dayalu (2016)
	4-Chloro-2-fluorobenzoic acid	Konschnik and Dayalu (2016)
	4-fluoro-2-(trifluoromethyl)benzoic acid	Konschnik and Dayalu (2016)
	4-Fluoro-3-(trifluomethyl)benzoic acid	Konschnik and Dayalu (2016)
	Benzoic acid, 3,5-difluoro-	Konschnik and Dayalu (2016)
	cis-4-ethyl-5-octyl-2,2-*bis*(trifluoromethyl)-1,3 dioxolane	Maguire-Boyle and Barron (2014)
	p-Fluorobenzoic acid	Konschnik and Dayalu (2016)
	tri-fluoromethyl tetradeculate	Maguire-Boyle and Barron (2014)
Oil soluble alkyl esters	ND	Deans (2007)
Unstable emulsions	ND	Catlett et al. (2013)
Controlled-release polymers and solid tracers	ND	Salman et al. (2014)

[a] ND = none disclosed.

A different set of tracers have been proposed for identifying environmental impacts from hydraulic fracturing fluids (Kurose, 2014). These tracers are designed so that the fluids from individual wells are identifiable while having no environmental impact themselves. DNA and nanoparticles with magnetic properties made specifically for each well have been proposed for this purpose (Kurose, 2014).

5.3.4 Proppants

Proppants are small particles carried down the well and into fractures by hydraulic fracturing fluid. They hold the fractures open after the injection pressure has been released and the hydraulic fracturing fluid has been removed (Brannon and Pearson, 2007). The propped fractures provide a path for the hydrocarbon to flow from the reservoir. The EPA's analysis of FracFocus 1.0 data showed that 98% of disclosures reported sand as the proppant, making sand (i.e., quartz) the most commonly reported proppant (U.S. EPA, 2015a). Other proppants include man-made or specially engineered particles, such as high-strength ceramic materials or sintered bauxite (Schlumberger, 2014; Brannon and Pearson, 2007). Proppant types can be used individually or in combinations.

5.3.5 Example Hydraulic Fracturing Fluids

There is no standard composition of hydraulic fracturing fluid used across the United States, and the literature does not present any typical hydraulic fluid composition. In Figure 5-4, we present two examples of hydraulic fracturing fluid mixtures based on analyses conducted on the EPA FracFocus 1.0 project database (U.S. EPA, 2015c). These examples represent two different types of fluids used at two different wells. The first is a slickwater, and the second is an energized fluid.[1] Details of each fluid are presented in the figure along with pie charts of their composition, as given by maximum percent by mass of the total hydraulic fracturing fluid.

The first hydraulic fracturing fluid (Figure 5-4a), the slickwater, is composed of 87% water, 13% sand, and 0.05% chemicals, by mass. The fluid is 71% fresh water and 16% reused produced water, with a total water volume of 4,763,000 gal (18,030,000 L). The chemical composition consists of six different additive types (acid, friction reducer, biocide, scale inhibitor iron control, and corrosion inhibitor) and a total of 13 chemicals.

The second hydraulic fracturing fluid (Figure 5-4b), the energized fluid, is more complex and consists of 58% water, 28% nitrogen gas, 13% sand, and 1.5% additives, by mass, with a total water volume of 105,000 gal (397,000 L). The hydraulic fracturing fluid composition consists of 10 additives (acid, surfactant, foamer, corrosion inhibitor, biocide, friction reducer, breaker, scale inhibitor, iron control, and clay stabilizer) and a total of 28 chemicals.

[1] A slickwater is a hydraulic fracturing fluid designed to have a low viscosity to allow pumping at high rates. The critical additive in a slickwater is the friction reducer, which makes the fluid "slick."

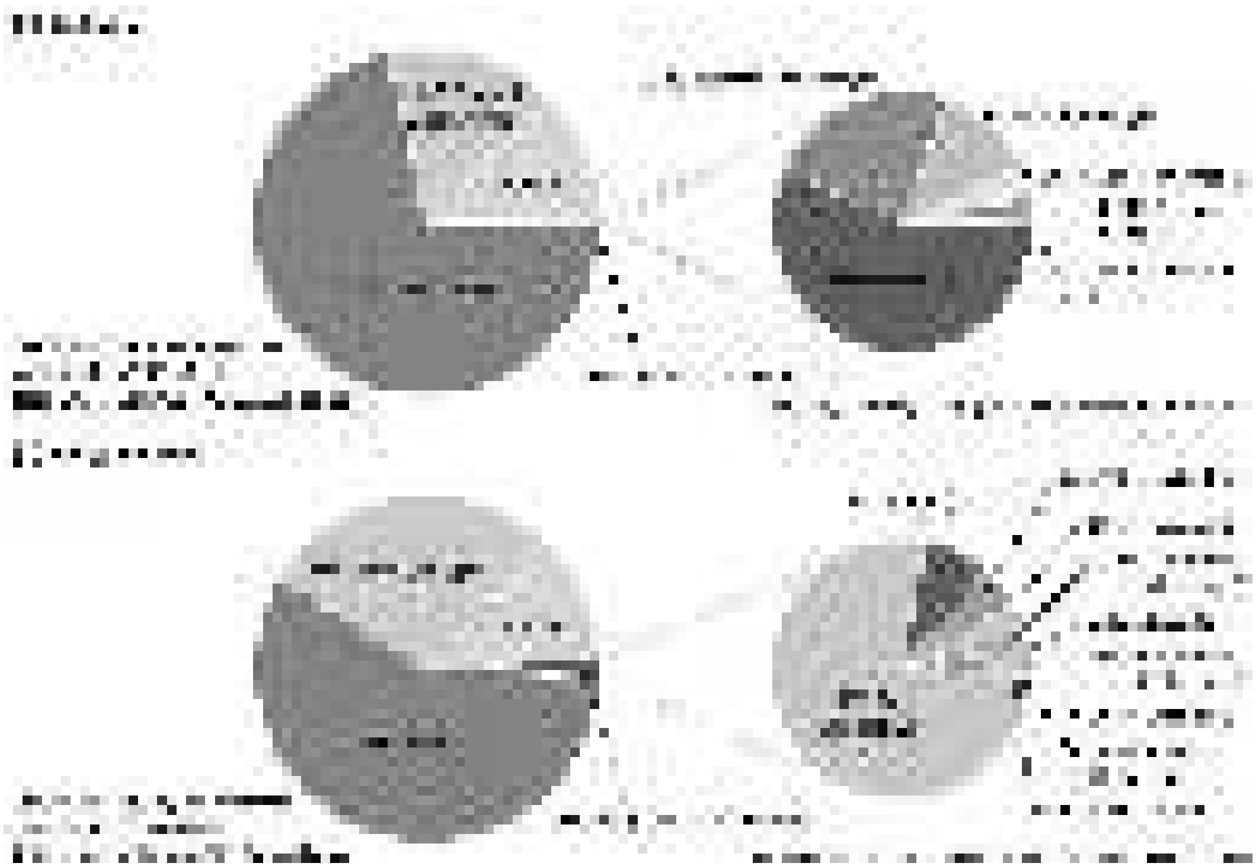

Figure 5-4. Example hydraulic fracturing fluids.

Example compositions of (a) slickwater and (b) energized fluid. The base fluid and proppants are on the left, and the additive breakdown is on the right. The number in parentheses represents the number of chemicals in that additive. See Table 5-1 for the function of different additives and the most common chemicals in those additives reported as based on the analysis of the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

These two examples give an idea of the difference in the compositions of two example hydraulic fracturing fluids. These compositions are the final mixture as if the entire fluid were mixed at once; they are generally not the actual composition at any given point in time. These compositions provide the potential composition of a spilled hydraulic fracturing fluid during the chemical mixing stage. Any of these ingredients (e.g., biocide) could be released by itself or mixed with the base fluid with other additives. The variability of hydraulic fracturing fluids from well to well and site to site makes it difficult to assess the potential of hydraulic fracturing additive or fluid release.

5.4 Frequency and Volume of Hydraulic Fracturing Chemical Use

This section highlights the different chemicals used in hydraulic fracturing fluids and discusses the frequency and volume of use. Using the EPA FracFocus 1.0 project database (Text Box 5-1), we focus our analysis on the individual chemicals that are used as ingredients in additive formulations, rather than on the complete mixture of chemicals that may be present in a hydraulic fracturing fluid. Operators can report information about well location, date of operations, and water and chemical use to the FracFocus registry. Chemicals are reported in FracFocus by using the chemical name and the Chemical Abstract Services Registration Number (CASRN), which is a unique number identifier for every chemical.[1] The information on specific chemicals, particularly those most commonly used, can be used to assess potential impacts on drinking water resources. The volume of chemicals stored on-site provides information on the potential volume of a chemical spill.

Text Box 5-1. The FracFocus Registry and EPA FracFocus Report.

The Ground Water Protection Council (GWPC) and the Interstate Oil and Gas Compact Commission (IOGCC) developed a national hydraulic fracturing chemical registry, FracFocus (www.fracfocus.org). Well operators can use the registry to disclose information about chemicals and water they use during hydraulic fracturing. As part of the EPA's Study of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking Water Resources, the EPA published the report titled *Analysis of Hydraulic Fracturing Fluid Data from the FracFocus Chemical Disclosure Record Registry 1.0* (U.S. EPA, 2015a). For this report, the EPA accessed data from FracFocus 1.0 from January 1, 2011 to February 28, 2013, which included more than 39,000 disclosures (records of well data) in 20 states that had been submitted by operators prior to March 1, 2013. Accompanying the U.S. EPA (2015a) report is the published EPA FracFocus 1.0 project database, which. It supported analyses of FracFocus chemical and water use data (U.S. EPA, 2015c), and a report describing the details of data management for development of the project database (U.S. EPA, 2015b).

Submission to FracFocus was initially voluntary and varied from state to state. During the timeframe covered in the EPA FracFocus 1.0 report (January 2011 to February 2013), six of the 20 states with data submitted to FracFocus and included in the EPA FracFocus 1.0 project database began requiring operators to disclose chemicals used in hydraulic fracturing fluids to FracFocus (Colorado, North Dakota, Oklahoma, Pennsylvania, Texas, and Utah). Three other states started requiring disclosure to either FracFocus or the state (Louisiana, Montana, and Ohio), and five states required or began requiring disclosure to the state (Arkansas, Michigan, New Mexico, West Virginia, and Wyoming). Alabama, Alaska, California, Kansas, Mississippi, and Virginia did not have reporting requirements during the period of the EPA's study.

The EPA's analysis may or may not be nationally representative. Disclosures from the five states reporting the most disclosures to FracFocus (Texas, Colorado, Pennsylvania, North Dakota, and Oklahoma) comprise over 78% of the disclosures in the database; nearly half (47%) of the disclosures are from Texas. Thus, data from these states are most heavily represented in the EPA's analyses.

A disclosure reports the total water volume (in gallons) and the chemicals used in the fluid (as maximum ingredient concentration by mass both in the additive and in the hydraulic fracturing fluid). The actual mass of the chemicals used in the fluid are not reported. The fluid composition reported in the disclosure does not necessarily reflect the actual composition of the fluid at any time. Rather, the disclosure represents what the total composition of the fluid would be if all chemicals used were mixed together at their maximum reported concentration.

The EPA summarized information on the locations of the wells in the disclosures, water volumes used, and the frequency of use and maximum ingredient concentrations of the chemicals in the additives and the hydraulic fracturing fluid. Additional information can be found in the EPA FracFocus 1.0 report (U.S. EPA, 2015a) and in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

[1] A CASRN and chemical name combination identify a chemical substance, which can be a single chemical (e.g., hydrochloric acid, CASRN 7647-01-0) or a mixture of chemicals (e.g., hydrotreated light petroleum distillates (CASRN 64742-47-8), a complex mixtures of C9 to C16 hydrocarbons). For simplicity, we refer to both pure chemicals and chemical substances that are mixtures, which have a single CASRN, as "chemicals."

The EPA compiled a list of 1,084 chemicals with unique CASRNs reported as used in the hydraulic fracturing process between 2005 and 2013 (full list, methodology, and details on sources in Appendix H).[1] These chemicals fall into different chemical classes and include 455 organic chemicals, 258 inorganic chemicals, and 361 organic mixtures or polymers. The chemical classes of commonly used hydraulic fracturing chemicals include but are not limited to:

- Acids (e.g., hydrochloric acid, peroxydisulfuric acid, acetic acid, citric acid);
- Alcohols (e.g., methanol, isopropanol, ethylene glycol, propargyl alcohol, ethanol);
- Aromatic hydrocarbons (e.g., benzene, naphthalene, heavy aromatic petroleum solvent naphtha);
- Bases (e.g., sodium hydroxide, potassium hydroxide);
- Hydrocarbon mixtures (e.g., petroleum distillates);
- Polysaccharides (e.g., guar gum);
- Surfactants (e.g., poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy, 2-butoxyethanol); and
- Salts (e.g., sodium chlorite, dipotassium carbonate).

Further details on these chemicals and their associated hazards are presented in Chapter 9.

All of the sources of information used to compile the list of chemicals found in hydraulic fracturing fluids (Appendix H) relied on reported use of those chemicals. In some cases, analysis of produced water samples by advanced analytical methods could provide information on suspected hydraulic fracturing additives, but other sources for the chemicals need careful consideration (Hoelzer et al., 2016). These sources include chemicals originating from components of the well, lab contamination, or subsurface reaction. We limit our discussion of hydraulic fracturing fluid chemicals to those directly reported as used.

An additional complication in providing an assessment on the use of chemicals in hydraulic fracturing is that companies can withhold reporting chemicals to the FracFocus registry by claiming that a chemical is Confidential Business Information (CBI). The use of CBI is to protect proprietary information, such as trade secrets. Details on CBI are provided in Text Box 5-2.

[1] The EPA used eight different sources to identify chemicals used in hydraulic fracturing fluids. This included the EPA FracFocus report (U.S. EPA, 2015a) and seven other sources (U.S. EPA, 2013a; Colborn et al., 2011; House of Representatives, 2011; NYSDEC, 2011; PA DEP, 2010a; U.S. EPA, 2004a; Material Safety Data Sheets).

Text Box 5-2. Confidential Business Information (CBI).

This assessment relies in large part upon information provided to the EPA or to other organizations. The submitters (e.g., businesses that operate wells or perform hydraulic fracturing services) may view some of the information as confidential business information (CBI) and accordingly asserted CBI claims to protect it. Information deemed to be CBI may include trade secrets or other proprietary business information entitled to confidential treatment under Exemption 4 of the Freedom of Information Act (FOIA) and other applicable laws. The FOIA and EPA's CBI regulations may allow for information claimed as CBI provided to the EPA to be withheld from the public, including in this document. In practical terms, when a well operator claims CBI for a specific chemical, they do not report the name or CASRN for that chemical in the disclosure submitted to the FracFocus registry (see Text Box 5-1 for information on FracFocus).

The EPA evaluated data from FracFocus, a national hydraulic fracturing chemical registry used and relied upon by some states, industry groups, and non-governmental organizations, as described in Text Box 5-1. A company submitting a disclosure to FracFocus may choose to not report the identity of a chemical it considers CBI. More than 70% of disclosures contained at least one chemical claimed as CBI and 11% of all chemicals were claimed as CBI. Of the disclosures containing CBI chemicals, there were an average of five CBI chemicals per disclosure (U.S. EPA, 2015a). Rates of withholding chemical information (designating a chemical as CBI) have increased from 11% in the 2011 to early 2013 time period of the EPA report, to 16.5% across the 2011 to early 2015 time period in another study using FracFocus data, with 92% of FracFocus 2.0 disclosures including at least one chemical claimed as CBI (Konschnik and Dayalu, 2016). When a chemical is claimed as CBI, there is no public means of accessing information on these chemicals (e.g., CASRN, name). Sometimes a CBI entry will provide the chemical family (Appendix H).

Consistent with the EPA's *Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2011d), data were submitted by nine service companies to the EPA regarding chemicals used in hydraulic fracturing from 2005 to 2009. These data were separate from the EPA FracFocus 1.0 project database. The data were submitted directly to the EPA, with the actual names and CASRNs of any chemicals the company considered CBI. This included a total of 381 CBI chemicals, with a mean of 42 CBI chemicals per company and a range of 7 to 213 (U.S. EPA, 2013a). Of these 381 chemicals, some companies only provided a generic chemical name and no CASRN, some provided neither a chemical name or CASRN, while others provided a CASRN and a specific chemical name. This resulted in 80 CASRNs/chemical names on this CBI list. Table H-3 lists generic chemical names, which may have been designed to mask CBI chemical names given to the EPA. The EPA does not know if the 381 chemicals represent 381 unique chemicals or if there are duplicates on this list.

The prevalence of CBI claims in the EPA FracFocus 1.0 project database limits completeness of the data set and introduces uncertainty. Ideally, all data would be available on all chemicals to do a full assessment.

5.4.1 National Frequency of Use of Hydraulic Fracturing Chemicals

A total of 692 chemicals were identified in the EPA FracFocus 1.0 project database that were reported as used in hydraulic fracturing from January 1, 2011, to February 28, 2013. This information comes from a total of 35,957 disclosures with chemical data in the database (U.S. EPA, 2015a).[1]

[1] Chemicals may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates), and they each have a single CASRN. Of these 692 chemicals, 598 had valid fluid and additive concentrations (34,675 disclosures). Sixteen chemicals were removed, because they were minerals listed as being used as proppants. This left a total of 582 chemicals (34,344 disclosures).

Table 5-3 presents the 35 chemicals (5% of all chemicals identified in the EPA's study) that were reported as ingredients in additives in at least 10% of the EPA FracFocus 1.0 project database disclosures for all states reporting to FracFocus 1.0 during this time (U.S. EPA, 2015c). This table also includes the top four additives in which the given chemical was reported as an ingredient.

Table 5-3. Chemicals identified in the EPA FracFocus 1.0 project database in 10% or more disclosures, with the percent of disclosures for which each chemical is reported as an ingredient in an additive and the top four reported additives for which the chemical is used. If a chemical is reported to be used in less than four additives, the table presents all additives (U.S. EPA, 2015c).

No.	Chemical name[a]	CASRN	Percent of disclosures[b]	Additives in which chemical is used (four most common, EPA FracFocus 1.0 project database)[c]
1	Methanol	67-56-1	72%	Corrosion inhibitors, surfactants, non-emulsifiers, scale control
2	Hydrotreated light petroleum distillates[d]	64742-47-8	65%	Friction reducers, gelling agents and gel stabilizers, crosslinkers and related additives, viscosifiers
3	Hydrochloric acid	7647-01-0	65%	Acids, solvents, scale control, clean perforations
4	Water[e]	7732-18-5	48%	Acids, biocides, clay control, scale control
5	Isopropanol	67-63-0	47%	Corrosion inhibitors, non-emulsifiers, surfactants, biocides
6	Ethylene glycol	107-21-1	46%	Crosslinkers and related additives, scale control, corrosion inhibitors, friction reducers
7	Peroxydisulfuric acid, diammonium salt	7727-54-0	44%	Breakers and breaker catalysts, oxidizer, stabilizers, clean perforations
8	Sodium hydroxide	1310-73-2	39%	Crosslinkers and related additives, biocides, pH control, scale control
9	Guar gum	9000-30-0	37%	Gelling agents and gel stabilizers, viscosifiers, clean perforations, breakers and breaker catalysts
10	Quartz[e]	14808-60-7	36%	Breakers and breaker catalysts, gelling agents and gel stabilizers, scale control, crosslinkers and related additives
11	Glutaraldehyde	111-30-8	34%	Biocides, surfactants, crosslinkers and related additives, sealers
12	Propargyl alcohol	107-19-7	33%	Corrosion inhibitors, inhibitors, acid inhibitors, base fluid
13	Potassium hydroxide	1310-58-3	29%	Crosslinkers and related additives, pH control, friction reducers, gelling agents and gel stabilizers

No.	Chemical name[a]	CASRN	Percent of disclosures[b]	Additives in which chemical is used (four most common, EPA FracFocus 1.0 project database)[c]
14	Ethanol	64-17-5	29%	Surfactants, biocides, corrosion inhibitors, fluid foaming agents and energizers
15	Acetic acid	64-19-7	24%	pH control, iron control agents, acids, gelling agents and stabilizers
16	Citric acid	77-92-9	24%	Iron control agents, scale control, gelling agents and gel stabilizers, pH control
17	2-Butoxyethanol	111-76-2	21%	Surfactants, corrosion inhibitors, non-emulsifiers, fluid foaming agents and energizers
18	Sodium chloride	7647-14-5	21%	Breakers/breaker catalysts, friction reducers, scale control, clay control
19	Solvent naphtha, petroleum, heavy arom.[f]	64742-94-5	21%	Surfactants, non-emulsifiers, inhibitors, corrosion inhibitors
20	Naphthalene	91-20-3	19%	Surfactants, non-emulsifiers, corrosion inhibitors, inhibitors
21	2,2-Dibromo-3-nitrilopropionamide	10222-01-2	16%	Biocides, clean perforations, breakers and breaker catalysts, non-emulsifiers
22	Phenolic resin	9003-35-4	14%	Proppants, biocides, clean perforations, base fluid
23	Choline chloride	67-48-1	14%	Clay control, clean perforations, base fluid, biocides
24	Methenamine	100-97-0	14%	Proppants, crosslinkers and related additives, biocides, base fluid
25	Carbonic acid, dipotassium salt	584-08-7	13%	pH control, proppants, acids, surfactants
26	1,2,4-Trimethylbenzene	95-63-6	13%	Surfactants, non-emulsifiers, corrosion inhibitors, inhibitors
27	Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides[g]	68424-85-1	12%	Biocides, non-emulsifiers, corrosion inhibitors, scale control
28	Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture)[h]	127087-87-0	12%	Surfactants, friction reducers, non-emulsifiers, inhibitors
29	Formic acid	64-18-6	12%	Corrosion inhibitors, acids, inhibitors, pH control

No.	Chemical name[a]	CASRN	Percent of disclosures[b]	Additives in which chemical is used (four most common, EPA FracFocus 1.0 project database)[c]
30	Sodium chlorite	7758-19-2	11%	Breakers/breaker catalysts, biocides, oxidizer, proppants
31	Nonyl phenol ethoxylate	9016-45-9	11%	Non-emulsifiers, resin curing agents, activators, friction reducers
32	Tetrakis(hydroxymethyl) phosphonium sulfate	55566-30-8	11%	biocides, scale control, clay control
33	Polyethylene glycol	25322-68-3	11%	Biocides, non-emulsifiers, surfactants, clay control
34	Ammonium chloride	12125-02-9	10%	Friction reducers, crosslinkers and related additives, scale control, clay control
35	Sodium persulfate	7775-27-1	10%	Breakers and breaker catalysts, oxidizer, pH control

[a] Chemical refers to chemical substances with a single CASRN; these may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates). Chemical names are sometimes different between FracFocus 1.0 and Appendix H, though they will have the same CASRN.

[b] Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.

[c] Analysis considered 32,885 disclosures and 615,436 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; valid concentrations; and valid purpose. Disclosures that did not meet quality assurance criteria (5,645) or other, query-specific criteria were excluded from analysis.

[d] Hydrotreated light petroleum distillates (CASRN 64742-47-8) is a mixture of hydrocarbons, in the C9 to C16 range.

[e] Quartz (CASRN 14808-60-7), the proppant most commonly reported, and water were also reported as an ingredient in other additives (U.S. EPA, 2015a).

[f] Heavy aromatic solvent naphtha (petroleum) (CASRN 64742-94-5) is mixture of aromatic hydrocarbons in the C9 to C16 range.

[g] Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides (CASRN 68424-85-1) is a mixture of benzalkonium chloride with carbon chains between 12 and 16.

[h] Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture) (CASRN 127087-87-0) is mixture with varying length ethoxy links.

There is no single chemical used in all hydraulic fracturing fluids across the United States. Methanol is the most commonly used chemical, reported at 72.1% of wells in the EPA FracFocus 1.0 project database and is associated with 33 types of additives, including corrosion inhibitors, surfactants, non-emulsifiers, and scale control (U.S. EPA, 2015c).[1] Table 5-3 also shows the variability in different chemicals included in the EPA FracFocus 1.0 project database. The percentage of disclosures reporting a given chemical suggests the likelihood of that chemical's use at a site. Only three chemicals (methanol, hydrotreated light petroleum distillates, and hydrochloric acid) were used at more than half of the sites nationwide, and only 12 were used at more than one-third.

[1] The number of additives may be an overestimate due to parsing issues. The true number of additives may be smaller.

In addition to providing information on frequency of use, the EPA FracFocus 1.0 project database provides the maximum concentration by mass of a given chemical in an additive. For example, methanol is the most frequently reported chemical. The median value for the maximum mass concentration reported for an additive in FracFocus disclosures is 30%, with a range of 0.44% to 100% (5th to 95th percentile).[1] Thus, methanol is generally used as part of a mixture of chemicals in the hydraulic fracturing fluid (typically at a concentration around 30% by mass). Other times, methanol is used as an additive in its pure form (concentration 100%). Therefore, methanol will sometimes be stored on-site in a mixture of chemicals and other times as pure methanol. This wide range of possible concentrations of methanol further complicates assessing the potential impact of spills, as the properties of the fluid will depend on the different chemicals present and on their concentrations. For all chemicals, spills of a highly concentrated chemical can have different potential impacts than spills of dilute mixtures. For more discussion on fluid and additive chemical composition, see Section 5.4.5.

A more recent study of FracFocus 2.0 data evaluated disclosures dating from March 9, 2011 to April 13, 2015 (96,449 disclosures) and reported 981 unique chemicals used in hydraulic fracturing (Dayalu and Konschnik, 2016; Konschnik and Dayalu, 2016). The earlier, EPA study (covering the 2011 to early 2013 time period) found 692 chemicals (U.S. EPA, 2015a). Konschnik and Dayalu (2016) identified 263 new CASRNs in addition to the 1,084 identified by the EPA (Appendix H), increasing the number of chemicals by approximately 24%. Of the new CASRNs, the only chemical reported in more than 1% of all disclosures was Alcohols, C9-11-iso-,C10-rich, ethoxylated propoxylated (CASRN 154518-36-2).

The 20 most common chemicals reported in Konschnik and Dayalu (2016) are similar to those listed in Table 5-3. There are three chemicals reported on their 20 most common list that are not included in Table 5-3. These chemicals are: sorbitan, mono-(9Z)-9-octadecenoate (CASRN 1338-43-8, reported in 29.6% disclosures (Konschnik and Dayalu, 2016) vs. 4% (U.S. EPA, 2015c), ethoxylated C12-16 alcohols (CASRN 68551-12-2, 27.9% vs. 4%), and thiourea polymer (CASRN 68527-49-1, 24.8% vs. 8%). Ammonium chloride was on each list, but disclosures increased from 10% to 30.5%. Four chemicals in Table 5-3 were not on their 20 most frequently used list: solvent naphtha, petroleum, heavy arom. (CASRN 64742-94-5), naphthalene (CASRN 91-20-3), 2,2-Dibromo-3-nitrilopropionamide (CASRN 10222-01-2), and phenolic resin (CASRN 9003-35-4).

5.4.2 Nationwide Oil versus Gas

Analyses based on the EPA FracFocus 1.0 project database also can elucidate the differences between the chemicals used during hydraulic fracturing for oil production and those used for gas production, providing a better understanding of potential spill impacts from each. Appendix Tables C-1 and C-2 present the chemicals reported in at least 10% of all gas (34 chemicals) and oil (39 chemicals) disclosures nationwide.

[1] For more information on how chemicals are reported to FracFocus see www.fracfocus.org and U.S. EPA (2015a).

Many of the same chemicals are used for oil and gas, but some chemicals are used more frequently in oil production and others more frequently in gas.[1] For example, hydrochloric acid is the most commonly reported chemical for gas wells (73% of disclosures); it is the fifth most frequently reported chemical for oil wells (58% of disclosures). However, both oil and gas operators each reports using methanol in 72% of disclosures. Methanol is the most common chemical used in hydraulic fracturing fluids at oil wells and the second most common chemical in hydraulic fracturing fluids at gas wells.

5.4.3 State-by-State Frequency of Use of Hydraulic Fracturing Chemicals

The composition of hydraulic fracturing fluids varies from site to site. Since the impacts of hydraulic fracturing occur locally, the potential impact depends on the chemicals used locally. We investigated geographic variation of chemical use based on the frequency of chemicals reported to FracFocus and included in the EPA FracFocus 1.0 project database by state (U.S. EPA, 2015c). Appendix Table C-3 presents and ranks chemicals reported most frequently for each state (U.S. EPA, 2015c). The list of the 20 most frequently reported chemicals used in each state together include 94 unique chemicals. A total of 94 chemicals indicates some level of similarity in chemical usage among states.[2]

Methanol is reported in 19 of the 20 states (95%). Alaska is the only state in which methanol is not reported (based on the state's 20 disclosures). The percentage of disclosures reporting use of methanol ranges from 38% (Wyoming) to 100% (Alabama, Arkansas).

Ten chemicals (excluding water) are among the 20 most frequently reported in 14 of the 20 states. These chemicals are: methanol; hydrotreated light petroleum distillates; ethylene glycol; isopropanol; quartz; sodium hydroxide; ethanol; guar gum; hydrochloric acid; and peroxydisulfuric acid, diammonium salt.[3] These 10 chemicals are also the most frequently reported chemicals nationwide.

This state analysis showed that methanol is used across the contiguous U.S. (not Alaska). There are 9 other chemicals that are frequently used across the United States. Beyond those, however, there are a number of different chemicals that are used in one state more commonly than others, and many chemicals may not be used at all in other states.

[1] This separation was done solely based on whether it was an oil or gas disclosure. The analysis did not separate out reservoir factors, such as temperature, pressure, or permeability, which may be important factors for which chemicals are used. There is no nationwide criterion to distinguish oil wells from gas wells. Production wells often produce some of both. A well identified as gas-producing in one place might be identified as oil-producing in another. This could affect the distribution of chemical use among these wells.

[2] The range of possible number of chemicals is from 20 to 400. If every state used the same 20 chemicals, there would be 20 different chemicals. If all 20 states each used 20 different chemicals, then there would be a total of 400 chemicals used.

[3] Quartz was the most commonly reported proppant and also reported as an ingredient in other additives (U.S. EPA, 2015a).

5.4.4 Volume of Chemical Use

Understanding the volume of chemicals used at each site is important for understanding potential impacts of chemicals as well as potential severity of impacts on drinking water resources. The chemical volume governs how much will be stored on-site, the types of containers required, the total amount that could spill, and how much could end up in a drinking water resource. While the on-site hydraulic fracturing service company has precise knowledge of the composition and volume of chemicals stored on-site, this information is not generally publicly available. We conducted a comprehensive review of publicly available sources and found two sources (OSHA, 2014a, b; Sjolander et al., 2011) that identify specific chemicals used at a hydraulic fracturing site and provide information on volumes. These are presented in Table 5-4. The volume of chemicals totaled 7,500 gal (28,000 L) and 14,700 gal (55,600 L) for the two sources, with a mean volume for an individual chemical of 1,900 gal (7,200 L) and 1,225 gal (4,637 L), respectively. The range of volumes for each chemical used is 30 to 3,690 gal (114 to 14,000 L).

Table 5-4. Example list of chemicals and chemical volumes used in hydraulic fracturing.
Volumes are for wells with an unknown number of stages and at least one perforation zone. Every well and fluid formulation is unique. Blank cells are data not reported.

		Sjolander et al. (2011)[a]		OSHA (2014a, 2014b)[b]	
Ingredient	**Examples**	**Volume (gal) or mass (lb)**	**Percent overall[c]**	**Volume (gal)**	**Percent by volume**
Water		4,000,000 gal	94.62	2,700,000 gal	90
Proppant	Sand	~ 1,500,000 lb[d]	5.26	285,300 gal	9.51
Acid	Hydrochloric acid or muriatic acid	1,338 gal	0.03	3,690 gal	0.123
Friction reducer	Polyacrylamide, mineral oil	2,040 gal	0.05	2,640 gal	0.088
Surfactant	Isopropanol			2,550 gal	0.085
Potassium chloride				1,800	0.06
Gelling agent	Guar gum or hydroxymethyl cellulose	-[e]	-[e]	1,680	0.056
Scale inhibitor	Ethylene glycol, alcohol, and sodium hydroxide	2,040 gal	0.05	1,290	0.043
pH buffer	Carbonate			330	0.011
Preservative	Ammonium persulfate			300	0.01

Ingredient	**Examples**	**Sjolander et al. (2011)[a]**		**OSHA (2014a, 2014b)[b]**	
		Volume (gal) or mass (lb)	**Percent overall[c]**	**Volume (gal)**	**Percent by volume**
Crosslinker	Borate salts	-[e]	-[e]	210	0.007
Iron control	Citric acid	-[e]	-[e]	120	0.004
Corrosion inhibitor	n,n-Dimethyl formamide	-[e]	-[e]	60	0.002
Biocide / antimicrobial agent	Glutaraldehyde, ethanol, methanol	2,040 gal	0.05	30	0.001
Gel-breaker	Ammonium persulfate	-[e]	-[e]		
Total volume of all chemicals		7,458 gal	0.18	14,700	0.49
Individual chemical volume: mean (full range)		1,864.5 gal (1,338 – 2,040 gal)		1,225 (30 – 3,690)	

[a] Adapted from Penn State "Water Facts" publication entitled *Introduction to Hydrofracturing* (Sjolander et al., 2011). Composite from two companies: Range Resources, LLC, and Chesapeake Energy, which released in July 2010 the chemistry and volume of materials typically used in their well completions and stimulations.

[b] Adapted from a table generated by the Occupational Safety and Health Administration (OSHA) for use in a training module (OSHA, 2014a, b).

[c] As presented in Sjolander et al. (2011); does not explicitly state percent by mass or by volume.

[d] Sjolander et al. (2011) presents proppant in pounds instead of gallons.

[e] Listed as an ingredient, but no information on volume or percentage.

Because of the limited information on chemical volumes publicly available, we estimated chemical volumes used across the nation based on the information provided in the EPA FracFocus 1.0 project database. Figure 5-5 plots median estimated chemical volumes, ranked from high to low, with the range of 5th to 95th percentiles. Estimated volumes used are presented for the 74 chemicals that were reported in at least 100 disclosures in the EPA FracFocus 1.0 project database and for which density data were available. The estimated median volumes vary widely among the different chemicals, covering a range of near zero to 27,000 gal (98,000 L). The mean of the estimated median volumes was 650 gal (2,500 L), and the mean of the estimated median mass was 3,200 lb (1,500 kg) (U.S. EPA, 2015c). Mass, volume, and density data are presented in Appendix C along with the estimation methodology and assumptions used.

With the median chemical volume, we can estimate total chemical volume for all chemicals used. Based on the above mean of median chemical volumes of 650 gal (2,500 L) per chemical, and given that the median number of chemicals used at a site is 14 (U.S. EPA, 2015a), an estimated 9,100 gal (34,000 L) of chemicals may be used per well. Given that the number of chemicals per well ranges from 4 to 28 (U.S. EPA, 2015a), the total volume of chemicals per well may range from 2,600 to 18,000 gal (9,800 to 69,000 L).

Another way to estimate total volume of chemicals per well is to use the estimated median volume of 1.5 million gal (5.7 million L) of fluid used to fracture a well (U.S. EPA, 2015a) (Chapter 4) and assume that up to 2% of that volume consists of chemicals added to base fluid (Carter et al., 2013; Knappe and Fireline, 2012), resulting in up to 30,000 gal (114,000 L) of chemicals used per well.

Using the estimated volume per chemical of 650 gal (2,500 L), we can also estimate volume per additive and extrapolate to estimate on-site chemical storage. If we assume three to five chemicals per additive, then total volume per additive stored on-site would be approximately 1,900 to 3,200 gal (7,400 to 12,000 L). On-site containers generally store 20% to 100% more additive volume than ultimately used (Houston et al., 2009; Malone and Ely, 2007). This would suggest that 2,300 to 6,500 gal (8,800 to 25,000 L) per additive are stored on site.

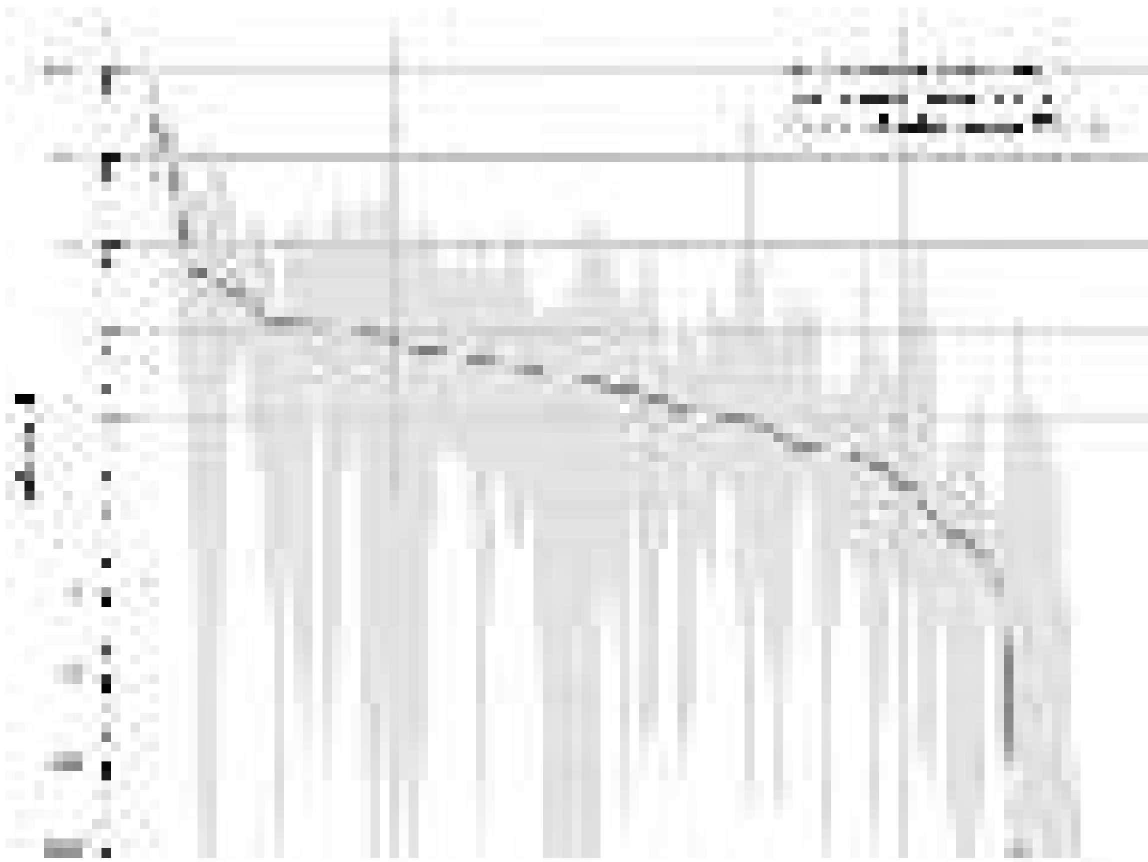

Figure 5-5. Estimated median volumes for 74 chemicals reported in at least 100 disclosures in the FracFocus 1.0 project database for use in hydraulic fracturing from January 1, 2011 to February 28, 2013.

Chemicals are plotted in order of largest to smallest median volume. Shaded area represents the zone of 5% and 95% confidence limits. Derived from U.S. EPA (2015c).

5.4.5 Chemical Composition of Hydraulic Fracturing Fluids and Additives

As the hydraulic fracturing process proceeds, the composition of the fluid injected changes over time. The overall composition of additives and hydraulic fracturing fluid may be reported by well operators to the FracFocus national registry, depending on the local disclosure requirements and operator preference. For each chemical that is injected into a well (excluding CBI chemicals), the maximum concentration in the resulting overall fluid and in each additive is given as maximum percent by mass. Based on this information, we calculated the median chemical composition reported in at least 10% of the disclosures in the EPA FracFocus 1.0 project database (Table 5-3) and a range based on the 5th and 95th percentile. Table 5-5 shows that some chemicals may be used in their pure form (100% of mass in a given additive). These chemicals include: methanol, hydrochloric acid, water, isopropanol, guar gum, citric acid, 2,2-Dibromo-3-nitrilopropionamide, tetrakis(hydroxymethyl) phosphonium sulfate, and sodium persulfate.

Chemicals may be stored in their concentrated, pure form, resulting in the potential for spills of concentrated volumes of these chemicals, which may increase the severity of impacts if they reach a drinking water resource. Once chemicals are mixed with the base fluid to form the hydraulic fracturing fluid, the chemical is diluted to much lower concentrations, which has the potential for a less severe impact. However, a larger volume of spill could occur with smaller concentrations. The larger volume may increase the potential for a spill reaching a drinking water resource, albeit at a lower concentration. There is the further complication of the hazard of the associated chemicals, since a smaller mass of a more hazardous chemical may be of more concern than a larger mass of a less hazardous chemical (as discussed in Chapter 9). It is therefore impossible to make a general statement without more detail on the spill characteristics, including the hazard, concentration, and volume.

Appendix Table C-6 provides mean, median, 5th and 95th percentile mass (kg) estimates for all reported chemicals in 100 or more disclosures in the EPA FracFocus 1.0 project database where density information was available.

Table 5-5. Fluid and additive composition by maximum mass percent.

Median, 5th and 95th percentile maximum concentration in hydraulic fracturing fluid and in additive (percent by mass) for the chemicals identified in the EPA FracFocus 1.0 project database in 10% or more disclosures. See Table 5-3 for percentage of disclosures and the common additives for which these chemicals are used. Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.

EPA-standardized chemical name	CASRN	Maximum concentration in hydraulic fracturing fluid (percent by mass)			Maximum concentration in additive (percent by mass)		
		Median	5th Percentile	95th Percentile	Median	5th Percentile	95th Percentile
Methanol	67-56-1	0.0092	0.00011	0.12	30	0.44	100
Distillates, petroleum, hydrotreated light	64742-47-8	0.025	0.0013	0.35	30	0.70	70
Hydrochloric acid	7647-01-0	0.15	0.0083	1.3	15	2.8	60

EPA-standardized chemical name	CASRN	Maximum concentration in hydraulic fracturing fluid (percent by mass)			Maximum concentration in additive (percent by mass)		
		Median	5th Percentile	95th Percentile	Median	5th Percentile	95th Percentile
Water	7732-18-5	0.53	0.00065	82	65	5.0	100
Isopropanol	67-63-0	0.0038	0.000020	0.15	20	0.30	100
Ethylene glycol	107-21-1	0.016	0.00027	0.11	30	0.59	60
Peroxydisulfuric acid, diammonium salt	7727-54-0	0.0069	0.00010	0.064	100	0.11	100
Sodium hydroxide	1310-73-2	0.0092	0.000040	0.077	10	0.085	52
Guar gum	9000-30-0	0.16	0.0019	0.42	50	1.6	100
Quartz	14808-60-7	0.0033	0.000030	12	2.0	0.10	97
Glutaraldehyde	111-30-8	0.0072	0.00039	0.023	27	0.040	60
Propargyl alcohol	107-19-7	0.00015	0.000010	0.0028	8.0	0.0032	30
Potassium hydroxide	1310-58-3	0.0070	0	0.053	15	0.14	50
Ethanol	64-17-5	0.0034	0.000060	0.098	30	1.0	60
Acetic acid	64-19-7	0.0033	0	0.037	50	1.0	90
Citric acid	77-92-9	0.0027	0.000060	0.017	60	7.0	100
2-Butoxyethanol	111-76-2	0.0047	0	0.14	10	0.29	60
Sodium chloride	7647-14-5	0.0083	0	0.14	30	0.020	50
Solvent naphtha, petroleum, heavy arom.	64742-94-5	0.0051	0.000020	0.035	10	0.00052	30
Naphthalene	91-20-3	0.0014	0	0.0055	5.0	0.0023	5.0
2,2-Dibromo-3-nitrilopropionamide	10222-01-2	0.0018	0.000010	0.022	98	10	100
Phenolic resin	9003-35-4	0.12	0.0046	1.1	5.0	0.80	20
Choline chloride	67-48-1	0.062	0.00068	0.14	75	0.75	80
Methenamine	100-97-0	0.010	0	0.21	1.0	0	2.0
Carbonic acid, dipotassium salt	584-08-7	0.039	0	0.15	60	30	60
1,2,4-Trimethylbenzene	95-63-6	0.00067	0	0.0068	1.0	0.010	20

EPA-standardized chemical name	CASRN	Maximum concentration in hydraulic fracturing fluid (percent by mass)			Maximum concentration in additive (percent by mass)		
		Median	5th Percentile	95th Percentile	Median	5th Percentile	95th Percentile
Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides	68424-85-1	0.0019	0	0.0041	7.0	3.0	10
Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture)	127087-87-0	0.0025	0.000010	0.0089	5.0	5.0	10
Formic acid	64-18-6	0.0021	0	0.030	60	0.11	98
Sodium chlorite	7758-19-2	0.0040	0.00018	0.037	10	5.0	30
Nonyl phenol ethoxylate	9016-45-9	0.0088	0.000030	0.085	10	5.0	54
Tetrakis(hydroxymethyl) phosphonium sulfate	55566-30-8	0.011	0.00025	0.065	60	0.029	100
Polyethylene glycol	25322-68-3	0.0035	0.000010	0.038	20	0.0071	70
Ammonium chloride	12125-02-9	0.0025	0.00029	0.022	10	1.5	30
Sodium persulfate	7775-27-1	0.0017	0.000020	0.022	100	100	100

5.5 Chemical Management and Spill Potential

This section provides a description of the primary equipment used in the chemical mixing and well injection processes, along with a discussion of the spill vulnerabilities specific to each piece of equipment. Equipment breakdown or failure can trigger a spill itself, and it can also lead to a suspension of activity and the disconnection and reconnection of various pipes, hoses, and containers. Each manipulation of equipment poses additional potential for a spill. The EPA found that 31% of chemical spills on or near the well pad related to hydraulic fracturing resulted from equipment failure (U.S. EPA, 2015m). When possible, we describe documented spills, associated with or attributed to specific pieces of equipment, in text boxes in the relevant subsections.

Equipment used in hydraulic fracturing operations typically consists of chemical storage trucks, oil storage tanks/tanker trucks; a slurry blender; one or more high-pressure, high-volume fracturing pumps; the main manifold; surface lines and hoses; and a central control unit (Table 5-6). There are many potential sources for leaks and spills in this interconnected system. Furthermore, hydraulic fracturing operations are mobile and must be assembled at each well site, and each assembly and disassembly presents a potential for spills.

Equipment varies in age and technological advancement depending upon service company standards and costs associated with purchase and maintenance. Older equipment may have experienced wear and tear, which may be a factor in spills caused by equipment failure. New equipment may be more automated, potentially reducing opportunities for human error. Information detailing the extent of technological and age differences in fracturing equipment across sites and operators is limited.

Table 5-6. Examples of typical hydraulic fracturing equipment and its function.

Equipment	Function
Acid transport truck	Transports acids to job sites; the truck has separate compartments for multiple acids or additives.
Chemical storage truck	Transports chemicals to the site in separate containment units or totes. Chemicals are typically stored on and pumped from the chemical storage truck.
Base fluid tanks	Stores the required volume of base fluid to be used in the hydraulic fracturing process.
Proppant storage units	Holds proppant and feeds it to the blender via a large conveyor belt.
Blender	Takes fluid (e.g., water) from the fracturing tanks and proppant (e.g., sand) from the proppant storage unit and combines them with additives before transferring the mixture to the fracturing pumps
High-pressure fracturing pumps	Pressurizes mixed fluids received from the blender and injected into the well.
Manifold trailer with hoses and pipes	Serves as a transfer station for all fluids. Includes a trailer with a system of hoses and pipes connecting the blender, the high-pressure pumps, and the fracturing wellhead.
Fracturing wellhead or frac head	Allows fracture equipment to be attached to the well; located at the wellhead.
Central control unit or frac van	Monitors the hydraulic fracturing job using pressure and rate data supplied from around the job site.

While the primary equipment and layout are generally the same across well sites, the type, size, and number of pieces of equipment may vary depending on a number of factors (Malone and Ely, 2007):

- Size and type of the fracture treatment;
    - Length of well and number of stages;
    - Number of wells drilled per well pad;
    - Geographic location;
    - Depth below surface;
    - Length of the fractures;

- Volumes and types of additives, proppants, and fluids used; and
- Operating procedures of the well operator and service company (e.g., some companies require backup systems in case of mechanical failure, while others do not).

Figure 5-6 provides a schematic diagram of a typical layout of hydraulic fracturing equipment.

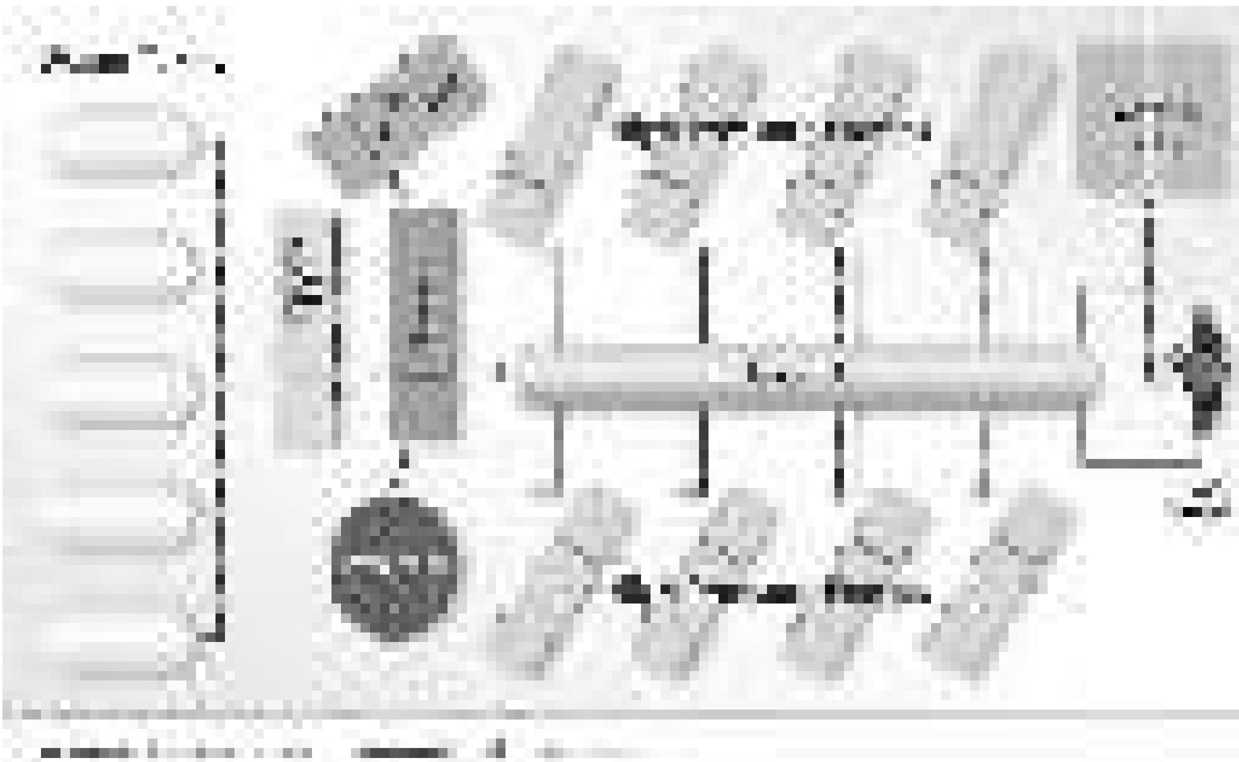

Figure 5-6. Typical hydraulic fracturing equipment layout.
This illustration shows how the various components of a typical hydraulic fracturing site fit together. The numbers of pumps and tanks vary from site to site. Some sites do not use a hydration unit as the gel is batch mixed prior to the treatment (Olson, 2011; BJ Services Company, 2009).

5.5.1 Storage

This section provides an overview of publicly available information on storage and containment of chemicals used in the hydraulic fracturing process. Most public sources provide general information on the types and sizes of containment units. While operators maintain a precise inventory of volumes of chemicals stored and used for each site, this information is typically not made public.

The volumes of each chemical used are based on the size and site-specific characteristics of each fracture treatment. Sites often store an excess of the design volume of chemicals for contingency purposes, typically 20% to 100% beyond what is necessary (Houston et al., 2009; Malone and Ely, 2007). See Text Box 5-3 for documented spills from storage units.

Text Box 5-3. Spills from Storage Units.

Of the 151 spills of chemicals, additives, or fracturing fluid discussed and evaluated in (U.S. EPA, 2015m) (see Text Box 5-10 for more information), 54 spills were from storage units. Storage units include totes or tanks used for storing individual chemicals or additives and larger tanks containing hydraulic fracturing fluid. These spills resulted from equipment failure, failure of storage integrity, or human error. Sixteen of these spills were due to failure of container integrity, which includes holes and cracks in containers, demonstrating the importance of properly constructed and maintained storage units. The remaining spills from storage containers resulted from human error or equipment malfunctions or had an unknown cause.

5.5.1.1 Hydraulic Fracturing Base Fluid Storage

Base fluids used in hydraulic fracturing are typically stored on-site in large volume tanks. Non-water-based fluids may be stored in specialized containment units designed to prevent or minimize releases. For example, nitrogen and carbon dioxide must be stored in compressed gas or cryogenic liquid cylinders, as required by U.S. Department of Transportation (DOT) and OSHA regulations. Due to the large volume of base fluid storage tanks (about 21,000 gal or 80,000 L) (Halliburton, 1988), uncontrolled spills could damage other storage units and equipment, which could result in additional spills. Fresh water used as a base fluid is generally not a source of concern for spills. Reused wastewater, brine, and non-aqueous base fluids have the potential to adversely impact drinking water resources in the event of a spill. Chapter 7 discusses reusing hydraulic fracturing wastewater as a base fluid and the spill/release potential on-site from pits and impoundments.

5.5.1.2 Additive Storage

Additives are typically stored on-site in the containers in which they were transported and delivered. The additive trailer typically consists of a flatbed truck or van enclosure that holds a number of chemical totes, described below, and is equipped with metering pumps that feed chemicals to the blender. Depending on the size and type of the fracturing operation, there may be one or more additive trailers per site (NYSDEC, 2015; ALL Consulting, 2012). While additives constitute a relatively small portion of fluids used in a hydraulic fracturing fluid, additive volumes can range from the tens to tens of thousands of gallons.

The storage totes generally remain on the transportation trailers, but they also may be unloaded from the trailers and transferred to alternative storage areas before use. Our investigation did not find much information on how often, when, or why these transfers occur. Additional transfers and movement can increase the likelihood of a spill. See Text Box 5-4 for a documented spill from an additive storage unit.

Text Box 5-4. Spill from Additive (Crosslinker) Storage Tote.

On Sept 19, 2009, during a tote transfer in Pennsylvania, a tote of crosslinker fell off a forklift spilling approximately 15 – 20 gal (60 – 80 L) onto the well pad. The area was scraped clean with a backhoe, and the waste was placed in a lined containment area (PA DEP, 2012, ID# 1845178).

The most commonly used chemical totes are 200 – 375 gal (760 – 1,420 L) capacity polyethylene containers that may be reinforced with steel or aluminum mesh (NYSDEC, 2015). Metal containers may also be used. The totes are typically equipped with bottom release ports, which enable direct feed of the additives to the blending equipment (NYSDEC, 2015). Spills may occur if lines are improperly connected to these ports or if the connection equipment is faulty.

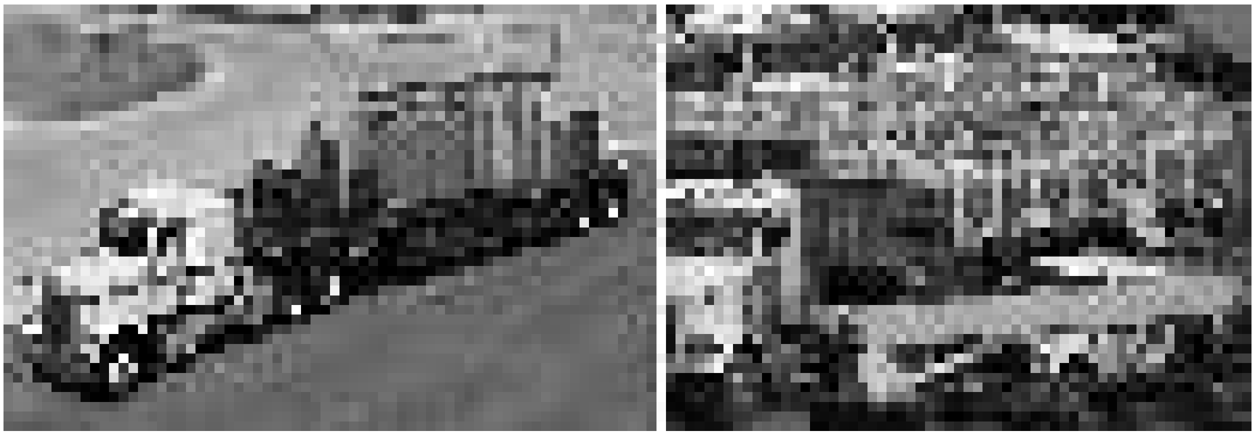

Figure 5-7. Metal and high-density polyethylene (HDPE) additive units.
The image on the left depicts metal totes (industry source). The image on the right depicts plastic totes. Source: NYSDEC (2011).

Certain additives require specialized containment units with added spill prevention measures. For example, additives containing methanol may be subject to federal safety standards, and industry has developed guidance on methanol's safe storage and handling (Methanol Institute, 2013).

Dry additives are typically transported and stored on flatbed trucks in 50 or 55 lb (23 or 25 kg) bags, which are set on pallets containing 40 bags each (NYSDEC, 2015; UWS, 2008; Halliburton, 1988). Proppants are stored on-site in large tanks or bins with typical capacities of 350,000 to 450,000 lb (150,000 to 200,000 kg) (ALL Consulting, 2012; BJ Services Company, 2009; Halliburton, 1988).

5.5.1.3 Acid Storage

Acids are generally stored on-site in the containment units in which they are transported and delivered. A typical acid transport truck holds up to 5,000 gal (19,000 L) of acid and can have multiple compartments to hold different kinds of acid (Arthur et al., 2009b). Acids such as hydrochloric acid and formic acid are corrosive and can be extremely hazardous in concentrated form. Therefore, acid transport trailers and fracture tanks must be lined with chemical-resistant coating designed to prevent leakage and must meet applicable DOT regulatory standards (pursuant to 40 Code of Federal Regulations (CFR) 173) designed to prevent or minimize spills.

Acid fracture treatments may use thousands of gallons of acid per treatment (Spellman, 2012). Given the large volumes used, failure of containment vessels during storage or failure of connections and hoses during pumping could result in high-volume acid spills. Details of a documented acid spill are presented in Text Box 5-5.

Text Box 5-5. Spill of Acid from Storage Container.

In July 2014, in Oklahoma, 20,000 gal (76,000 L) of hydrochloric acid spilled from a storage container when a flange malfunctioned. The acid spilled into a nearby alfalfa field, where it was contained with an emergency berm (Phillips, 2014; Wertz, 2014). There is no information on how much leached into soils or if the spill reached drinking water resources.

5.5.1.4 Gel Storage

Gels can be added to hydraulic fracturing fluid using either batch or continuous (also called "on-the-fly") mixing systems. Gelling agents and gel slurries are stored differently on-site and can pose different potential spill scenarios depending on whether the site is using batch or continuous mixing processes (BJ Services Company, 2009).

In a typical batch mixing process, powdered gelling agents and related additives (e.g., buffers, surfactants, biocides) are mixed on-site with base fluid water and proppant in large tanks, typically 20,000 gal (80,000 L)(BJ Services Company, 2009; Halliburton, 1988). The number of gel slurry tanks used varies based on site-specific conditions and the size of the fracture job. These tanks can be subject to leaks or overflow during the batch mixing process and during storage prior to injection. One of the disadvantages of batch mixing is the need for multiple suction hoses to draw pre-gelled fluids from storage tanks into the blender, if used, which can increase the potential for spills. Yeager and Bailey (2013) state that a drawback of batch mixing is the "fluid spillage and location mess encountered when pre-mixing tanks," suggesting that small spills are not uncommon during batch mixing. Details of a documented gel slurry spill are presented in Text Box 5-6. Details of a documented gel slurry spill are presented in Text Box 5-6.

Text Box 5-6. Spill of Gel Slurry during Mixing.

On April 9, 2010, in Louisiana, a company was mixing a gel slurry for an upcoming fracture job. The tank had developed a crack, which allowed approximately 10,000 gal (38,000 L) of water mixed with 60 gal (230 L) of gel to leak out. The mixture did not reach a water receptor, and absorbents were used to clean up the gel (LDEQ, 2013).

In continuous mixing operations, powdered gels are typically replaced with liquid gel concentrates (Allen, 2013; BJ Services Company, 2009). Operators prepare dilute gelling agents as needed using specialized hydration units (BJ Services Company, 2009). Liquid gel concentrates may be stored on-site in single-purpose tanker trucks (Harms and Yeager, 1987) but are more often stored in specialized mixing and hydration units (Ayala et al., 2006). Continuous mixing requires less preparation than batch mixing but typically requires more equipment (BJ Services Company, 2009; Browne and Lukocs, 1999). This can increase the possibility for spills resulting from equipment malfunctions or human error.

5.5.2 Hoses and Lines

High- and low-pressure hoses and lines are used to transfer hydraulic fracturing fluids from storage units to specialized mixing and pumping equipment and ultimately to the wellhead. A discussion of the different types of hoses and lines and possible points of failure is provided below. Figure 5-8 shows an example of hoses and lines at a hydraulic fracturing site.

Figure 5-8. Hoses and lines at a site in Arkansas.
Photo credit: Christopher Knightes (U.S. EPA).

Suction pumps and hoses move large volumes of base fluid to the blender. Incomplete or damaged seals in inlet or outlet connections can cause fluid leaks at the connection points. Improperly fitted seals also severely limit or eliminate suction lift, which can impair the suction pump and increase spill potential. Suction hoses themselves are susceptible to leaks due to wear and tear. Equipment providers recommend hoses be closely inspected to ensure they are in good operating condition prior to use (Upstream Pumping, 2015; BJ Services Company, 2009; Malone and Ely, 2007).

Discharge hoses transfer additives from containment vessels or totes to the blender. Given the potential for concentrated chemicals to spill during transfer from storage totes to the blender, it is particularly important that these hoses are in good condition and that connector seals or washers fit properly and are undamaged. Discharge hoses are also used to carry fracturing fluid pumped from the blender via the low-pressure manifold to the high-pressure pumps. Proppant-heavy fluids are pumped through discharge hoses at relatively low rates. If a sufficient flow rate is not maintained, proppant may settle out, damaging pumps and creating a potential for spills and leaks (Upstream Pumping, 2015; BJ Services Company, 2009; Malone and Ely, 2007).

High-pressure flow lines convey pressurized fluids from the high-pressure pumps into the high-pressure manifold and from the manifold into the wellbore. High-pressure flow lines are subject to erosion caused by the high-velocity movement of abrasive, proppant-laden fluid. Curved sections of flow lines (e.g., swivel joints) where abrasive fluids are forced to turn corners are particularly subject to erosion and are more likely to develop stress cracks or other defects that can result in a leak or spill. Safety restraints are typically used to prevent movement of flow lines such as in the event of failure and to help control spills. High-pressure flow lines are pressure-tested to detect fatigue or stress cracks prior to the fracturing treatment (OSHA, 2015; BJ Services Company, 2009; Arthur et al., 2008; Malone and Ely, 2007; Halliburton, 1988).

Nineteen spills of chemicals or fracturing fluids associated with leaks from hoses or lines had a total spill volume of 12,756 gal (48,287 L), with a median volume of 420 gal (1,600 L) (U.S. EPA, 2015m).

5.5.3 Blender

The blender is the central piece of equipment used to create the fracturing fluid for injection. It moves, meters, and mixes precise amounts of the base fluid, additives, and proppant and pumps the mixed slurry to high-pressure pumping equipment (BJ Services Company, 2009; Malone and Ely, 2007; Halliburton, 1988) (Figure 5-6). A typical blender consists of a centrifugal suction pump for pulling base fluid, one or more chemical metering pumps to apportion the additives, one or more proportioners to measure and feed proppant, and a central agitator tank where fluid components are mixed together.

The blending process is monitored to ensure that a uniform mixture is maintained regardless of injection rates and volumes. Excessive or reduced rates of flow during treatment can cause the blender to malfunction or to shut down, which can result in spills (Malone and Ely, 2007; Halliburton, 1988). For aqueous hydraulic fracturing fluid blends, spills that occur downstream of the blender will be a dilute mixture (less than or equal to 2%) of chemicals. Details of a spill from a blender are presented in Text Box 5-7.

Text Box 5-7. Spill of Hydraulic Fracturing Fluid from Blender.

In May 2006, a blender malfunctioned during a fracture job in Oklahoma. Approximately 294 gal (1,110 L) of fluid spilled into a nearby wheat field. The fluid consisted of hydrochloric acid, clay stabilizer, diesel, and friction reducer. Contaminated soil was removed by the operator (OCC, 2013, ID#137000).

5.5.4 Manifold

A trailer-mounted manifold and pump system functions as a central transfer station for all fluids used in the hydraulic fracturing operation. The manifold is a collection of low- and high-pressure pipes equipped with multiple fittings for connector hoses. Fluids are pumped from the blender through the low-pressure manifold hoses, which distribute fluids to high-pressure pump trucks. Pressurized slurry is sent from the pump trucks through high-pressure manifold lines and into additional high pressure lines that lead to the wellhead (Malone and Ely, 2007).

Manifold and pump system components require varying amounts of manual assembly and undergo varying amounts of pre-testing (Malone and Ely, 2007). Improperly tested parts may be more likely to break or lose functionality, leading to a spill. In manifolds requiring more manual assembly, there may be more opportunities for human error.

5.5.5 High-Pressure Fracturing Pumps

High-pressure fracturing pumps take the fracturing fluid mixture from the blender, pressurize it, and propel it down the well. Typically, multiple high-pressure, high-volume fracturing pumps are needed for hydraulic fracturing (Upstream Pumping, 2015). Such pumps come in a variety of sizes. Bigger pumps move greater volumes of fluid at higher pressures; therefore, spills from these pumps can be larger. Smaller pumps can require more operators and more maintenance (BJ Services Company, 2009), and therefore have the potential for an increased frequency of spills.

The “fluid ends” of hydraulic fracturing pumps are the pump components through which fluids are moved and pressurized. Pump fluid ends must withstand high pressure and move a large volume of abrasive fluid high in solids content. These pumps have multiple parts (e.g., seals, valves, seats and springs, plungers, stay rods, studs) that can wear out under the stress of high-pressure pumping (Upstream Pumping, 2015). Given the sustained pressures involved, careful maintenance of fluid ends is necessary to prevent equipment failure (Upstream Pumping, 2015; API, 2011). Details of a documented spill from a fracture pump are presented in Text Box 5-8.

Text Box 5-8. Spill of Fluid from Fracture Pump.

On December 19, 2011, in Arkansas, a fluid end on a fracture pump developed a leak, spilling approximately 840 gal (3,200 L) of fracturing fluid. A vacuum truck was used to recover the spilled fluid, and all affected soils were neutralized and taken to a landfill at the end of the job, after removal of the equipment (Arkansas DEQ, 2012, ID#063012).

5.5.6 Surface Wellhead for Fracture Stimulation

A wellhead assembly, often referred to as a frac head or frac stack, is temporarily installed on the wellhead during the fracture treatment. The frac head assembly allows high volumes of high-pressure proppant-laden fluid to be injected into the formation (OSHA, 2015; Halliburton, 2014; Stinger Wellhead Protection, 2010). The temporary frac head is equipped with specialized isolation tools so that the wellhead is protected from the effects of pressure and abrasion.

Figure 5-9. Multiple fracture heads.
Source: DOE/NETL.

As with all components of hydraulic fracturing operations, repeated and prolonged stress from highly pressurized, abrasive fluids may lead to equipment damage. The presence of minute holes or cracks in the frac head may result in leaks when pressurized fluids are pumped. In addition, surface blowouts or uncontrolled fluid releases may occur at the frac head because of valve failure or failure of other components of the assembly.[1] Details of a documented frac head failure are presented in Text Box 5-9.

Text Box 5-9. Spill from Frac Head Failure.

On March 2, 2011, in Colorado, a frac head failed during fracturing operations. Approximately 8,400 gal (32,000 L) of slickwater fracturing fluid leaked. The majority of the spill was contained on-site, though a small amount ran off into a nearby cornrow. There were 5,460 gal (20,700 L) of the fluid recovered, and saturated soils were scraped and stockpiled on the well pad. There was a net loss of 2,940 gal (11,100 L) (COGCC, 2012, ID#2524586).

[1] A well blowout is when there is uncontrolled flow of fluids out of a well.

5.6 Overview of Chemical Spills Data

Spills of hydraulic fracturing fluids have occurred across the country and have affected the quality of drinking water resources (U.S. EPA, 2015m; Brantley et al., 2014; COGCC, 2014; Gradient, 2013).[1] Spills may infiltrate drinking water resources by reaching surface water or by leaching into the groundwater. Potential impacts depend upon a variety of factors including the chemical spilled, environmental conditions, and actions taken in response to the spill.

5.6.1 EPA Analysis of Spills Associated with Hydraulic Fracturing

The EPA used data gathered from state and industry sources to characterize hydraulic fracturing-related spills between January 2006 and April 2012 (2015m) (see Text Box 5-10 for additional information). In this study, the sources had data on over 36,000 spills. Of these spills, the EPA identified 457 spills that occurred on or near the well pad and definitively related to hydraulic fracturing. Of these 457 spills, 151 were related to the chemical mixing process – spills that consisted of chemicals, additives, or fracturing fluids. Information in the spill reports included: spill causes (e.g., human error, equipment failure), sources (e.g., storage tank, hose or line), volumes, and environmental receptors. Spill reports contain little information on chemical-specific spill composition. Spilled fluids were often described by their additive type (e.g., acids, biocides, friction reducers, cross-linkers, gels,) or as a blended hydraulic fracturing fluid. Specific chemicals mentioned in spill reports included hydrochloric acid and potassium chloride.

Text Box 5-10. EPA Review of State and Industry Spill Data: Characterization of Hydraulic Fracturing-Related Spills.

As part of the EPA's Study of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking Water Resources, the EPA published the report titled *Review of State and Industry Spill Data: Characterization of Hydraulic Fracturing-Related Spills* (U.S. EPA, 2015m). In this document, hereafter referred to as the EPA spills report, the EPA used data gathered from state and industry sources to characterize hydraulic fracturing-related spills with respect to volumes spilled, materials spilled, sources, causes, environmental receptors, containment, and responses. For the purposes of the study, hydraulic fracturing-related spills were defined as those occurring on or near the well pad before or during the injection of hydraulic fracturing fluids or during the post-injection recovery of fluids. Because the main focus of this study is to identify hydraulic fracturing-related spills on the well pad that may reach surface or groundwater resources, the following topics were not included in the scope of this project: transportation-related spills, drilling mud spills, and spills associated with disposal through underground injection control wells.

Data on spills that occurred between January 2006 and April 2012 were obtained from nine state agencies with online spill databases or other data sources, nine hydraulic fracturing service companies, and nine oil and gas production well operators. The data sources used in this study contained over 36,000 spills. The EPA searched each spill report for keywords related to hydraulic fracturing (e.g., frac, glycol, flowback). Spill records from approximately 12,000 spills (33 percent of the total number of spills reviewed) contained insufficient information to determine whether the event was related to hydraulic fracturing.

(Text Box 5-10 is continued on the following page.)

[1] In this assessment, a spill is considered to be any release of fluids. Spills can result from accidents, fluid management practices, or illegal dumping.

Text Box 5-10 (continued). EPA Review of State and Industry Spill Data: Characterization of Hydraulic Fracturing-Related Spills.

Of the spills with sufficient information, the EPA identified approximately 24,000 spills (66%) as not related to hydraulic fracturing on or near the well pad. The remaining 457 spills (approximately 1%) occurred on or near the well pad and were definitively related to hydraulic fracturing. These 457 spills occurred in 11 different states over six years (January 2006 to April 2012). Of these 457 spills, 151 spills were chemical mixing-related and included spills of chemicals, additives, and hydraulic fracturing fluid, and 225 releases were of produced water (Chapter 7).

The EPA categorized spills according to the following causes: equipment failure, human error, failure of container integrity, other (e.g., well communication, weather, vandalism), and unknown.[1] Figure 5-10 presents the percent distribution of causes of hydraulic fracturing-related spills and for spills associated specifically with chemicals or fracturing fluid. The distributions for causes of hydraulic fracturing- and chemical mixing-related spills are similar.[2]

Spills in the EPA spills report were also categorized by the following sources: storage, equipment, well or wellhead, hose or line, and unknown. Figure 5-11 presents the percent distribution for all hydraulic fracturing- and chemical mixing-related spills associated with each source.

Figure 5-10. Percent distribution of the causes of spills.

Percent distribution by spill type for (a) 457 hydraulic fracturing-related spills (all spills) and (b) 151 chemical mixing-related spills. Data from U.S. EPA (2015m). Legend shows categories in clockwise order, from the top left of each pie chart.

[1] Well communication is when hydraulic fracturing fluids or displaced subsurface fluids move through newly created fractures into an offset well or its fracture network (See Section 6.3.2.3 for more details),

[2] Hydraulic fracturing-related spills are spills that occur at any phase within the hydraulic fracturing water cycle. These include chemicals, additives, hydraulic fracturing fluids (chemical mixing phase); produced water; and wastewater.

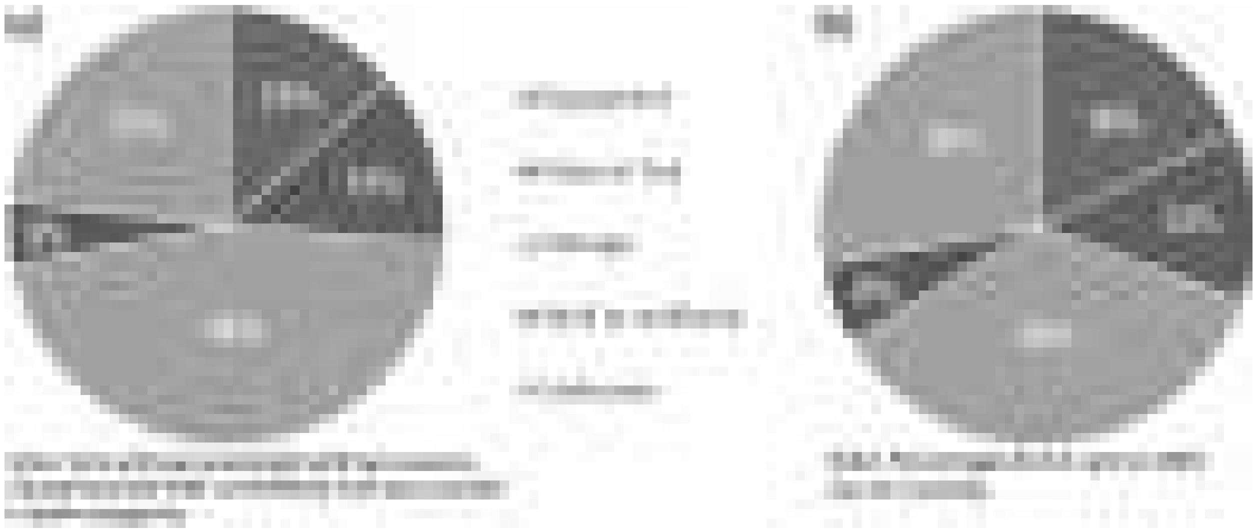

Figure 5-11. Percent distribution of the sources of spills.
Percent distribution of spill source of (a) 457 hydraulic fracturing-related spills (all spills) and (b) 151 chemical mixing-related spills. Data from U.S. EPA (2015m). Legend shows categories in clockwise order, from the top left of each pie chart.

Figure 5-12 presents the distribution of the number of spills for different volumes for hydraulic fracturing- and chemical mixing-related spills. The spills associated with chemical mixing ranged in volume from 5 to 19,320 gal (19 to 73,130 L), with a median volume of 420 gal (1,600 L). The source of largest spills was storage containers, which released approximately 83,000 gal (314,000 L) of spilled fluid (Figure 5-13b). Spills from wells or wellheads are often associated with high spill volumes. There were no reported chemical mixing-related spills greater than 100,000 gal (380,000 L) (Figure 5-15b).

Figure 5-12. Distribution of the number of spills for different ranges of spill volumes.
Number of spills due to Hydraulic Fracturing related activities and distribution of spill volumes for (a) 457 hydraulic fracturing-related spills (all spills) and (b) 151 chemical mixing-related spills. A value of 0% means that there were no spills in that category. Data from U.S. EPA (2015m).

Figure 5-13 presents the total volume of spills for different sources for all hydraulic fracturing-related activity and those associated with chemicals and fracturing fluid. The reported total volume of 125 of 151 chemical or hydraulic fracturing fluid spills was approximately 184,000 gal (697,000 L). The volume was unknown for 26 of these spills.

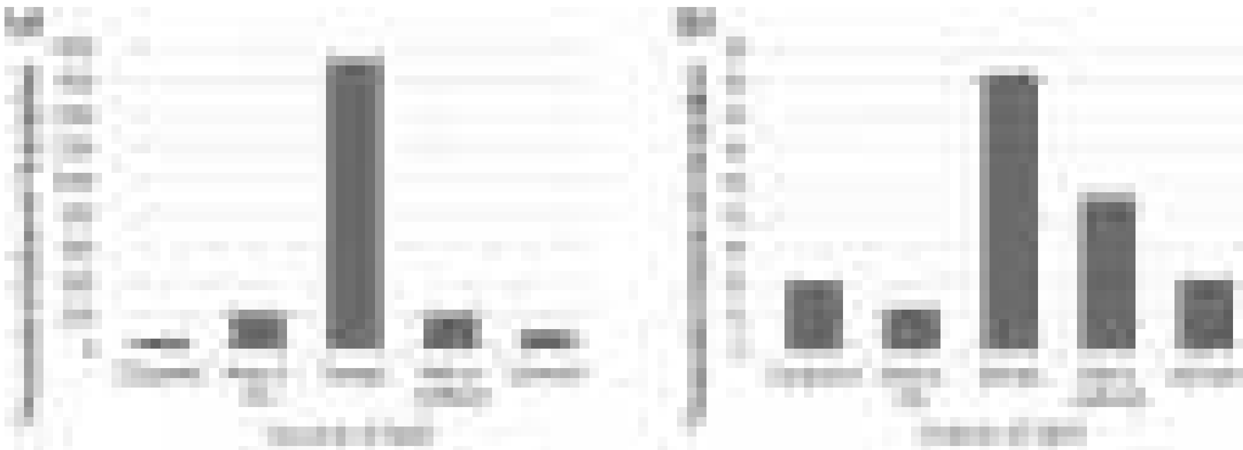

Figure 5-13. Total volume of fluids spilled from different sources.
Total volume of fluids spilled for (a) 457 hydraulic fracturing-related spills (all spills) and (b) 151 chemical mixing-related spills. Data from U.S. EPA (2015m).

Figure 5-14 presents the number of spills that reached environmental receptors, by receptor type, for all hydraulic fracturing-related activity (Figure 5-14a) and those associated with chemicals and fracturing fluid (Figure 5-14b). Environmental receptors (i.e., surface water, groundwater, soil) were identified in 101 of the 151 chemical mixing-related spills, or 67% of all chemical and fracturing fluid spills in the EPA's analysis (U.S. EPA, 2015m). Soil was by far the dominant environmental receptor, with 97 spills reaching soil; reported spill volumes ranged from 5 gal to 8,300 gal (19 L to 31,000 L). Thirteen spill reports indicated that the spilled fluid had reached surface water; reported spill volumes ranged from 28 gal to 7,350 gal (105 L to 27,800 L). Nine spill reports identified both soil and surface water as a receptor; spill volumes ranged from 28 gal to 2,856 gal (106 L to 10,800 L). Groundwater was not identified as a receptor from spills of chemicals or hydraulic fracturing fluid in any of the spill reports. Due to the lack of observations, it is often unclear if there was impact on groundwater. Movement through the subsurface is generally slow.[1] It may take years for a spilled fluid to reach groundwater or to reach a drinking water well. Thus, even if there is a pre-drilling characterization of groundwater chemistry in private/public wells, the time period of transport to actually detect a release at these private/public wells for contaminants that are transported at the rates of groundwater flow (see Section 5.8 for discussion on fate and transport of spilled chemicals).

[1] For example, a groundwater flow rate of 1 foot per day (not uncommon) would mean it could take approximately 1,000 days (~3 years) to travel 1,000 ft (305 m) from the well pad. Likewise, for a groundwater travel rate of 0.1 ft (0.03m) per day, impact would not be observed for at least 10,000 days (~27 years). For a travel rate of 10 ft (3 m) per day, the time for impact would be at least 100 days (~0.3 years).

Figure 5-14. Number of spills by environmental receptor.

Number of hydraulic fracturing-related spills and chemical mixing-related spills that reported whether an environmental receptor was reached for (a) 457 hydraulic fracturing-related spills (all spills) and (b) 151 chemical mixing related spills. "Yes" means that the spill was reported to reach this receptor. "Unknown" refers to hydraulic fracturing related spill events for which environmental receptors were specified as unknown or not identified (positively or negatively). "No" means the spill was reported to not meet this receptor. Data from U.S. EPA (2015m).

Storage units were the predominant sources of spills that reached an environmental receptor. Six spills from storage containers reached a surface water receptor. Thirty-eight of the spills from storage units reached a soil receptor. If a spill was confined to a lined well pad, for example, it might not have reached the soil, but most incident reports did not include whether the well pad was lined or unlined. Regarding spills of hydraulic fluids and chemicals from storage containers, 16 spills were due to failure of container integrity, which includes holes and cracks in containers, and overflowing containers as a result of human error or equipment malfunctions.

5.6.2 Estimated Spill Rate and Other Spill Reports and Data

The rate of reported spills during the hydraulic fracturing water cycle is estimated to range from 0.4 to 12.2 reported spills for every 100 wells, based on spills data from Brantley et al. (2014), Gradient (2013), Rahm et al. (2015), U.S. EPA (2013a), and North Dakota Department of Health (2015) (Appendix E) with a median rate of 2.6 reported spills for every 100 wells. (See Appendix Section C.4 and Appendix Table C-8 for details.) The estimated rates provide an approximate estimate of the potential frequency of the number of spills at a site. It is uncertain how representative these rates are of national spill rates or rates in other states. These numbers are not specific to the chemical mixing stage.

There are an estimated 2.6 reported spills of injected fluids and chemicals per 100 wells hydraulically fractured in North Dakota, based on an analysis of the North Dakota spills database for 2015, separate from the EPA spills report. The median spill volume of injection fluid was 1750 gal (6620 L), with a range of 2.9 to 17,600 gal (11 to 66,600 L). The median spill volume of injection chemical was 44 gal (167 L), with a range of 2.1 to 126 gal (7.9 to 477 L) (see Appendix E for more information).

A study of spills reported to the Colorado Oil and Gas Conservation Commission identified 125 spills during well stimulation (i.e., a part of the life of an oil and gas well that often, but not always, includes hydraulic fracturing) between January 2010 and August 2013. Of these spills, 51% were caused by human error and 46% were due to equipment failure (COGCC, 2014).

Considine et al. (2012) identified spills related to oil and gas development in the Marcellus Shale that occurred between January 2008 and August 2011 from Notices of Violations issued by the Pennsylvania Department of Environmental Protection. The authors identified spills greater than 400 gal (1,500 L) and spills less than 400 gal (1,500 L). Among these spills, spilled fluids included hydrochloric acid, gel friction reducer, and blended hydraulic fracturing fluid. Brantley et al. (2014) identified fewer than 10 instances of spills of additives and/or hydraulic fracturing fluids greater than 400 gal (1,500 L) that reached surface waters in Pennsylvania between January 2008 and September 2013. Reported spill volumes, among these spills, ranged from 3,400 gal to 227,000 gal (13,000 L to 859,000 L).

Surface spills related to hydraulic fracturing activities are not well documented in the scientific literature. There is some evidence of spills and impacts on environmental media (e.g., U.S. EPA, 2015i; Brantley et al., 2014; Gross et al., 2013; Papoulias and Velasco, 2013). Papoulias and Velasco (2013) stated that fluid overflowed a retention pit into surface water and likely contributed to the distress and deaths of threatened blackside dace fish in Kentucky. A variety of chemicals entered the creek and significantly reduced the stream's pH and increased stream conductivity. Using data from post-spill sampling reports in Colorado, Gross et al. (2013) identified concentrations of benzene, toluene, ethylbenzene, and xylene (BTEX) in groundwater samples. They attributed this to numerous hydraulic fracturing-related spills, although not necessarily specifically related to the chemical mixing process. This work, however, demonstrate that surface spills impacted groundwater, with a frequency of < 0.5% of active wells. Drollette et al. (2015) reported that organic compounds detected in shallow aquifers were consistent with surface spills, and that diesel range compounds had elevated concentrations compared to gasoline range compounds, further suggesting evidence of feasible groundwater impact.

5.7 Spill Prevention, Containment, and Mitigation

Spill prevention, containment, and mitigation affect the frequency and severity of the impacts of spills. Several factors influence spill prevention, containment, and mitigation, including federal, state, and local regulations and company practices. State regulations governing spill prevention, containment, and mitigation at hydraulic fracturing facilities vary in scope and stringency (Powell, 2013; GWPC, 2009). Employee training and equipment maintenance are also factors in effective spill prevention, containment, and mitigation. Analysis of these factors was outside the scope of this assessment.

The province of New Brunswick, Canada released rules for industry on responsible environmental management of oil and natural gas activities (GNB, 2013). Hydraulic fracturing service companies themselves may develop and implement spill prevention and containment procedures. It was beyond the scope of this assessment to evaluate the efficacy of these practices or the extent to which they are implemented.

Spill containment systems include primary, secondary, and emergency containment systems. Primary containment systems are the storage units, such as tanks or totes, in which fluids are intentionally kept. Secondary containment systems, such as liners and berms installed during site set-up, are intended to contain spilled fluids until they can be cleaned up. Emergency containment systems, such as berms, dikes, and booms, can be implemented temporarily in response to a spill.

The EPA investigated spill containment and mitigation measures in an analysis of spills related to hydraulic fracturing activities (U.S. EPA, 2015m). Of the approximately 25% of reports that included information on containment, the most common types of containment systems referenced in the hydraulic fracturing-related spill records included berms, booms, dikes, liners, and pits, though many of the spill reports did not indicate specific containment measures. Some spills were reported to breach the secondary containment systems. Breaches of berms and dikes were most commonly reported.

In cases where secondary containment systems were not present or were inadequate, operators sometimes built emergency containment systems. The most common were berms, dikes, and booms, but there were also instances where ditches, pits, or absorbent materials were used to contain the spilled fluid. Absorbent materials were generally used when small volumes (10 – 200 gal or 40 – 800 L) of additives or chemicals were spilled (U.S. EPA, 2015m). There was not enough information to detail the use of emergency containment systems or their effectiveness.

Remediation is the action taken to clean up a spill and its affected environmental media. The most commonly reported remediation activity, mentioned in approximately half of the hydraulic fracturing-related spill records evaluated by the EPA, was removal of spilled fluid and/or affected media, typically soil. Other remediation methods reported in U.S. EPA (2015m) included the use of absorbent material, vacuum trucks, flushing the affected area with water, and neutralizing the spilled material. Removal activities were found to occur in various combinations. For example, a spill of approximately 4,200 gal (16,000 L) of acid was cleaned up by first spreading soda ash to neutralize the acid and then removing the affected soil (U.S. EPA, 2015m).

5.8 Fate and Transport of Spilled Chemicals

The fate and transport of chemicals in the environment is complex. Due to the complexities of the processes and the site-specific and chemical-specific nature of spills, it is difficult to develop a full assessment of their fate and transport. The potential for hydraulic fracturing chemicals and fluids to reach drinking water resources is further complicated by the fact that these chemicals are typically present as mixtures, and unlike many organic contaminant mixtures (e.g., gasoline, diesels, PCBs, PAHs), hydraulic fracturing fluid chemicals are present as complex mixtures of chemicals covering a range of chemical classes with varying properties, often in aqueous solutions.

In this section, we provide a general overview of fate and transport of hydraulic fracturing-related chemicals spilled in the environment to give the reader a general understanding of the potential pathways and processes with which these chemicals can impact drinking water resources (Figure 5-15). We also include a discussion of the physicochemical properties of the organic chemicals used in hydraulic fracturing fluids, because these properties directly affect the transport of chemicals in the environment. This presentation is not meant to be exhaustive.

A chemical spill has the potential to migrate to and have an impact on drinking water resources. Once spilled, there are different paths that chemicals can travel and different processes they can undergo. Chemicals can react and transform into other chemicals, volatilize, travel to surface water, leach into and partition to soils, and/or reach groundwater. The potential path and the severity of the impact of a spill depend on different factors, including site conditions; the length of the path to a drinking water resource; the type and characteristics of the drinking water resource (stream, lake, aquifer); environmental conditions; climate; weather; chemical properties, constituents, and concentrations; and the volume of the release. The point in the chemical mixing stage where the spill occurs affects potential impact. If the spill occurs before chemicals are mixed into the base fluid, the chemicals will be in a more concentrated form. If the hydraulic fracturing fluid spills, then the chemicals will be diluted by the base fluid and can feasibly be present in lower concentrations. There can also be effects on persistence and mobility due to interactions among the chemicals present. The total mass of spilled chemical can therefore be dependent on what stage in the process a spill occurs.

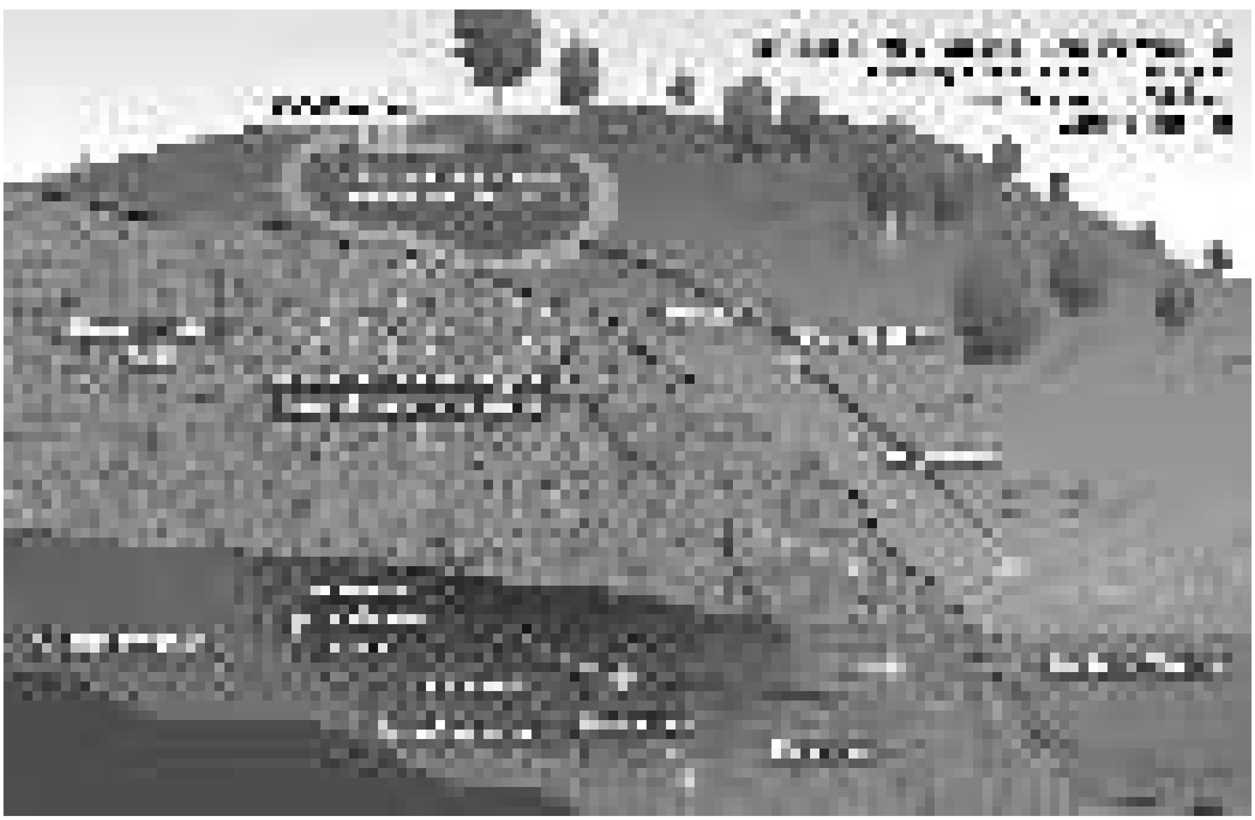

Figure 5-15. Fate and transport schematic for a spilled hydraulic fracturing fluid.
Schematic shows the potential paths and governing processes by which spilled chemicals can lead to potential impacts on drinking water resources.

For inorganic chemicals, the properties and processes governing fate and transport depend on pH, oxidation state, presence of iron oxides, soil organic matter, cation exchange capacity, and major ion chemistry (U.S. EPA, 1996).[1] Transport of these chemicals into groundwater depends on the nature of groundwater flow and flow through the unsaturated zone above the water table.[2] Potential transformations of inorganic chemicals differ from those of organic chemicals. Some inorganic anions (i.e., nitrate, chloride, and bromide) move with their carrier liquid and are affected mostly by physical transport mechanisms. For many inorganic chemicals, transport is driven by the physical flow processes (advection and dispersion), sorption, and precipitation. The relative role of each of these depends on both chemical and environmental characteristics.[3,4]

Determining the fate and transport of organic chemicals and mixtures is a complex problem, because of the many processes and different environmental media (air, soil, water). Unlike inorganic chemicals, organic chemicals degrade, which can affect their movement and potential impact. Schwarzenbach et al. (2002) formalized a general framework for organic chemical transport, where transport and transformation depend on both the nature of the chemical and the properties of the environment. The fate and transport of organic chemicals in soils has been presented in the literature (e.g., Bouchard et al., 2011; Rivett et al., 2011; Abriola and Pinder, 1985a, b) and in textbooks (e.g., Domenico and Schwartz, 1997; Schnoor, 1996; Freeze and Cherry, 1979b).

5.8.1 Potential Paths

Chemicals and hydraulic fracturing fluids that are released into the environment may travel along different potential paths, as detailed in Figure 5-15. Liquids can flow overland to nearby surface water or infiltrate the subsurface, where they may eventually reach the underlying groundwater or travel laterally to reach surface water. Movement can occur quickly or be delayed and have a later or longer-term impact. Surface and groundwater gain or lose flow to each other (Chapter 2), and can transport chemicals in the process. A dry chemical (e.g., gelling agents, biocides, friction reducers) released to the environment can remain where it is spilled. Any spill that is not removed could act as a long-term source of contamination. Wind could cause the chemical to disperse and rain could mobilize soluble chemicals. Dissolved chemicals can infiltrate into soil or flow overland. Insoluble chemicals and those sorbed to soil particles could be mobilized by rain events via runoff and erosion.

5.8.1.1 Movement across the Land Surface

In low permeability soils, there may be little infiltration and greater overland flow. Higher permeability soils will allow fluid to penetrate into the soil layer. In either case, some of the

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[1] Cation exchange capacity is the total amount of cations (positively charged ions) that a soil can hold. For example, when metal ions like Ca^{2+} and Na^{+} pass through the soil, they adhere and remain attached to the soil.

[2] The unsaturated zone is also referred to as the vadose zone. Meaning "dry," the vadose zone is the soil zone above the water table that is only partially filled by water.

[3] Advection is a mechanism for moving chemicals in flowing water, where a chemical moves along with the flow of the water itself.

[4] Sorption is the general term used to describe the partitioning of a chemical between soil and water and depends on the nature of the solids and the properties of the chemical.

chemicals in the fluid can sorb to the soil particles and the vegetation, and then these chemicals can be mobilized during precipitation, runoff, or erosion. As precipitation percolates through the soil, it can dissolve stored chemicals, which can then migrate toward groundwater. The type of release is also important. If the spill is a slow leak, then the liquid may pond and the affected area will expand slowly with greater potential for infiltration. If a more rapid release occurs, like a blowout or tank failure, then momentum can result in greater overland movement and less soil infiltration during the event, with greater potential to reach a nearby surface water.

5.8.1.2 Movement through the Subsurface

The unsaturated and saturated zones are the two zones of soils below the ground surface. Movement through the unsaturated zone is driven by the depth of ponding of the spilled fluid, gravity, and capillary properties of the subsurface.[1] In fractured rock or highly permeable soils, fluids can move quickly through the subsurface. In low permeability soil, the movement of the fluid may be slower. However, the presence of preferential pathways (e.g., fractures, heterogeneities, root holes, and burrows) can result in faster movement than the overall permeability would suggest.

As chemicals pass through the subsurface, some can sorb to soil or remain in the open spaces between soil particles, effectively slowing their movement. Chemicals can be mobilized during future precipitation events, resulting in infiltration towards groundwater or movement through the unsaturated zone towards surface water.

Fluids that move through the subsurface into the saturated zone will move in the direction of the flowing groundwater. Generally, fluids travel farther in systems with high groundwater flow rates and high recharge (e.g., sandy aquifers in humid climates) than in systems with low flow and low recharge. Chemicals can sorb to suspended soil particles, complex with naturally occurring chemicals (e.g., dissolved organic carbon), or associate with colloids and be transported with the flowing water.[2] These mechanisms can mobilize sparingly soluble chemicals that would otherwise be immobile.

5.8.2 Physicochemical Properties of Organic Hydraulic Fracturing Chemicals

Three physicochemical properties are useful to describe the movement of organic chemicals in the environment: (1) K_{ow}, the octanol-water partition coefficient, (2) the aqueous solubility, and (3) the Henry's law constant.[3] These properties describe whether a chemical will sorb to soil and organic

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[1] Capillarity occurs because of the forces of attraction of water molecules to themselves (cohesion) and to other solid substances such as soils (adhesion).

[2] Complexation is a reaction between two chemicals that form a new complex, either through covalent bonding or ionic forces. This often results in one chemical solubilizing the other.

[3] The octanol-water partition coefficient (K_{ow}) represents the ratio of the solubility of a compound in octanol (a nonpolar solvent) to its solubility in water (a polar solvent) in a mixture of the two. The higher the K_{ow}, the more nonpolar the compound.

matter or stay in water (K_{ow}), how much of a chemical may dissolve in water (aqueous solubility), and whether a chemical will tend to remain in the water or volatilize (Henry's law constant).[1]

The K_{ow} measures the relative hydrophobicity (chemicals that prefer to be in oil, log K_{ow} >0) and hydrophilicity (chemicals that prefer to be in water, log K_{ow} <0) of a chemical. Aqueous solubility is the maximum amount of a chemical that will dissolve in water in the presence of a pure chemical; solubility generally serves as an upper bound on possible concentrations. The Henry's law constant is the ratio of the concentration of a chemical in air (or vapor pressure) to the concentration of that chemical in water.

Estimates and measured values for physicochemical properties were obtained by using the Estimation Program Interface (EPI) Suite 4.1, as described in Appendix C.[2] Of the 1,084 chemicals the EPA listed as used in hydraulic fracturing (Appendix H), EPI Suite™ has estimated properties for 455 organic chemicals (42% of all chemicals) with structures that are considered suitably representative of the substance to compute properties within the constraints of EPI Suite™ software. Only uniquely defined organic desalted structures were submitted for property calculation. Figure 5-16 presents histograms of all 455 of the organic chemicals, sorted by four physicochemical parameters: measured log K_{ow} (n = 195), estimated log K_{ow} (n=455), estimated log of the aqueous solubility (n = 455), and estimated log of Henry's law constant (at 77°F or 25°C, n = 449). Property estimation methods are limited in their ability to predict physicochemical properties. Chemicals that are different than the chemicals used to develop the estimation techniques may have more error associated with their predictions. These figures enable comparison of physicochemical properties across the organic chemicals for which we have values. These figures show how the physicochemical properties are distributed and which chemicals have higher values compared to others with lower values. Limitations in knowing what chemicals are present (e.g., CBI) further hinders our ability to know the physicochemical properties of these chemicals and their potential to move through the environment and impact drinking water resources. These estimates are solely for the organic chemicals for which EPI Suite™ could be used. This does not provide information on the 258 inorganic chemicals or the 361 organic mixtures or polymers. This limits our ability to make a full assessment on the physicochemical properties of all chemicals, yet provides insight into the properties of the organic chemicals used.

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[1] We present the physicochemical parameter values using $\log_{10}$ because of the wide range of values that these parameters cover.

[2] EPI Suite™, version 4.1, http://www.epa.gov/opptintr/exposure/pubs/episuite.htm (U.S. EPA, 2012c). The EPI (Estimation Programs Interface) Suite™ is a Windows®-based suite of physicochemical property and environmental fate estimation programs developed by the EPA Office of Pollution Prevention and Toxics and Syracuse Research Corporation. EPI Suite™ provides estimates of physicochemical properties for organic chemicals and has a database of measured values for physicochemical properties when available. EPI Suite™ cannot estimate parameters for inorganic chemicals.

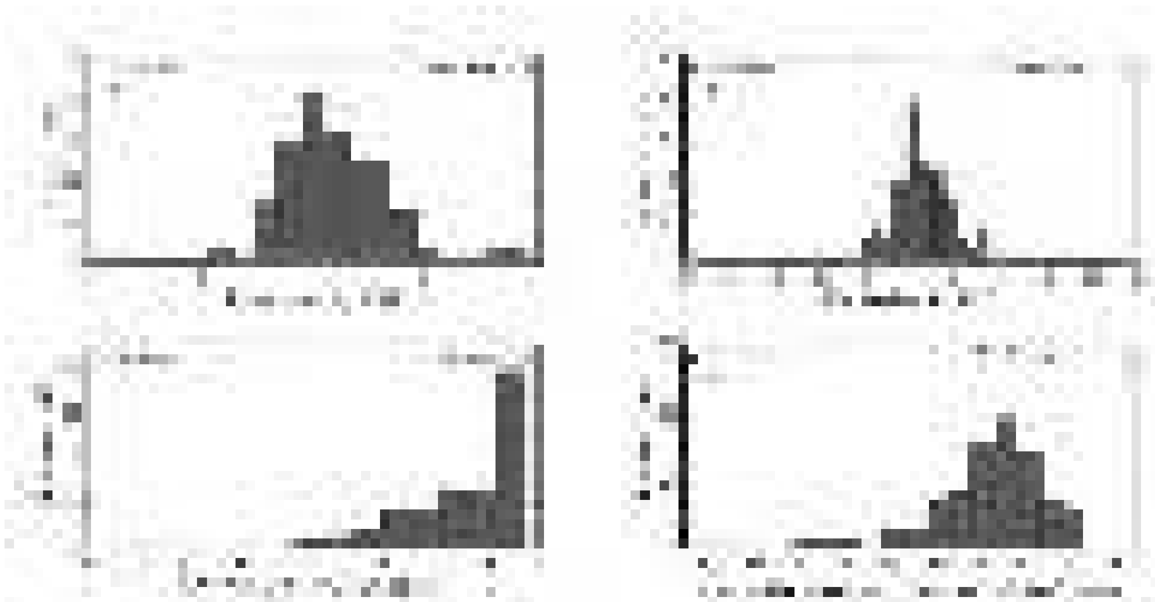

Figure 5-16. Histograms of physicochemical properties of organic chemicals used in the hydraulic fracturing process.

Physicochemical properties as given by EPI Suite™ (a) measured values of log K_{ow}, (b) estimated log K_{ow}, (c) estimated log Solubility, and (d) estimated log Henry's law constant.

We used EPI Suite™ to determine the physicochemical properties for 19 CBI chemicals used in hydraulic fracturing fluids. These chemicals were submitted to the EPA by nine service companies from 2005 to 2009 (see Text Box 5-3 for discussion on CBI).[1] The CBI chemical physicochemical properties are plotted as histograms in Appendix Figure C-1. The values of the physicochemical properties of known and CBI chemicals are similar, covering similar ranges and centered on similar values, suggesting that even though these chemicals are not publicly known, their physicochemical properties are not appreciably different from the known chemicals. This suggests that their fate and transport would not be appreciably different than the chemicals that are publicly known.

5.8.3 Mobility of Organic Hydraulic Fracturing Chemicals

Figure 5-16 shows the distribution of log K_{ow}, solubility, and Henry's Law constant for organic chemicals used in hydraulic fracturing fluids. These figures suggest that the organic chemicals used in hydraulic fracturing cover a wide range of physicochemical properties. For example, many chemicals are centered around log $K_{ow} = 0$, which indicates that these chemicals are likely to associate roughly equally with organic or aqueous phases. Many chemicals have log $K_{ow} > 0$, indicating less mobility, which may cause these chemicals to serve as later-term or long-term sources of impact on drinking water. Solubilities range from fully miscible to sparingly soluble. Many chemicals have log Henry's law constants less than 0, indicating that most are not highly volatile. Volatilization may not serve as a dominant loss process for hydraulic fracturing chemicals.

[1] Well operators may specify certain ingredients as confidential business information (CBI) and not disclose the chemicals used to FracFocus. The CASRNs of a range of CBI chemicals were provided to the EPA by nine service companies.

The 20 chemicals with the smallest K_{ow} (most mobile) may have greater potential to cause immediate impacts on drinking water resources (Appendix Table C-10). Most of these chemicals were infrequently reported in disclosures (≤2% of wells) in the EPA FracFocus 1.0 project database (U.S. EPA, 2015a). Choline chloride (14% of wells), used for clay control, and tetrakis(hydroxymethyl)-phosphonium sulfate (11% of wells), a biocide, were more commonly reported. The 20 chemicals with the largest K_{ow} (least mobile) may have a greater potential to serve as long-term sources of contamination (Appendix Table C-11). The estimated aqueous solubilities for some of these chemicals are extremely low, with highest solubilities of less than 10 μg/L. Seven low mobility chemicals were reported in disclosures in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c). Five were reported infrequently (<1% of wells). Tri-n-butyltetradecylphosphonium chloride (6% of wells), used as a biocide, and C>10-alpha-alkenes (8% of wells), a mixture of alpha-olefins with carbon numbers greater than 10 used as a corrosion inhibitor, were more commonly reported. Sorbitan, tri-(9Z)-9-octadecenoate, a mineral oil co-emulsifier (0.05% of wells) had the highest estimated log K_{ow} of 22.56.[1]

Table 5-7 shows the EPI Suite™ estimated physicochemical property values of the 20 chemicals most frequently reported nationwide in disclosures along with estimated mean and median volumes based on disclosures in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c). Most have log K_{ow} < 1, meaning that they are generally hydrophilic and will associate with water. These chemicals also have very high solubilities, so they will be mobile in the environment, transport with water, and can occur at high concentrations. These chemicals have the potential for faster impacts on drinking water resources.

Naphthalene (CASRN 91-20-3) has a measured log K_{ow} = 3.3 with an estimated solubility of 142.1 mg/L, which means it will be less mobile in the environment. Naphthalene will sorb to particles and move slowly through the environment and has the potential to act as a long-term source of contamination.[2] All of these chemicals have low Henry's law constants, so they tend not to volatilize. We also include ranges of similar physicochemical properties for two chemicals that are organic mixtures: distillates, petroleum, hydrotreated light (CASRN 64742-47-8) and solvent naphtha, petroleum, heavy arom. (CASRN 64742-94-5). Both of these are complex organic mixtures, and thus EPI Suite™ cannot estimate their properties. However, the Total Petroleum Hydrocarbon Work Group has provided regressions to relate physicochemical properties to the number of carbons for aliphatic and aromatic hydrocarbons (Gustafson et al., 1997), which shows that they have low solubilities and large log K_{ow}.

[1] Sorbitan, tri-(9Z)-9-octadecenoate, CASRN 26266-58-0, is soluble in hydrocarbons and insoluble in water, listed as an effective coupling agent and co-emulsifier for mineral oil (Santa Cruz Biotechnology, 2015; ChemicalBook, 2010).

[2] Chemicals can have the potential to be long-term sources of contamination when they move slowly through the environment. In this discussion, we are not accounting for biodegradation or other transformation processes, which may reduce the persistence of certain chemicals in the environment. Under certain conditions, for example, naphthalene is biodegradable, which can reduce or remove it from the environment, and thus may not be a long-term source of contamination.

Table 5-7. The 20 chemicals reported most frequently nationwide for hydraulic fracturing based on the EPA FracFocus 1.0 project database, with EPI Suite™ physicochemical parameters where available, and estimated mean and median volumes of those chemicals where density was available.

Excludes water, sodium chloride, and quartz. NA means that the physicochemical parameter is not provided by EPI Suite™ or the volume could not be estimated due to missing data. For organic salts, parameters are estimated using the desalted form. Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.

				Log K_{ow} (unitless)			Henry's Law Constant (atm m^3/mole @ 25°C)			Estimated volume, per disclosure (gal)	
Rank	**Chemical name**	**CASRN**	**Number of wells using chemical (% of wells)**	**Estimated**	**Measured**	Water solubility estimate from log K_{ow} (mg/L @ 25°C)	Estimated, bond method	Estimated, group method 25	Measured	Mean	Median
1	Methanol	67-56-1	24,753 (72%)	-0.63	-0.77	1.00×10^6	4.27×10^{-6}	3.62×10^{-6}	4.55×10^{-6}	1,218	110
2	Distillates, petroleum, hydrotreated light[a,b]	64742-47-8	22,463 (65%)	log K_{oc} = 4.5 to 6.7	NA	0.00035 to 0.12	55 to 69 cm^3/cm^3	NA	NA	NA	NA
3	Hydrochloric acid	7647-01-0	22,380 (65%)	NA	NA	NA	NA	NA	NA	28,320	3,110
4	Isopropanol	67-63-0	16,039 (47%)	0.28	0.05	4.024×10^5	7.52×10^{-6}	1.14×10^{-5}	8.10×10^{-6}	2,095	55
5	Ethylene glycol	107-21-1	15,800 (46%)	-1.2	-1.36	1.00×10^6	1.31×10^{-7}	5.60×10^{-11}	6.00×10^{-8}	614	184
6	Peroxydisulfuric acid, diammonium salt	7727-54-0	14,968 (44%)	NA	NA	NA	NA	NA	NA	NA	NA
7	Sodium hydroxide	1310-73-2	13,265 (39%)	NA	NA	NA	NA	NA	NA	551	38
8	Guar gum	9000-30-0	12,696 (37%)	NA	NA	NA	NA	NA	NA	NA	NA
9	Glutaraldehyde	111-30-8	11,562 (34%)	-0.18	NA	1.672×10^5	1.10×10^{-7}	2.39×10^{-8}	NA	1,313	122
10	Propargyl alcohol	107-19-7	11,410 (33%)	-0.42	-0.38	9.355×10^5	5.88×10^{-7}	NA	1.15×10^{-6}	183	2
11	Potassium hydroxide	1310-58-3	10,049 (29%)	NA	NA	NA	NA	NA	NA	NA	NA
12	Ethanol	64-17-5	9,861 (29%)	-0.14	-0.31	7.921×10^5	5.67×10^{-6}	4.88×10^{-6}	5.00E-06	831	121
13	Acetic acid	64-19-7	8,186 (24%)	0.09	-0.17	4.759×10^5	5.48×10^{-7}	2.94×10^{-7}	1.00×10^{-7}	646	47
14	Citric acid	77-92-9	8,142 (24%)	-1.67	-1.64	1.00×10^6	8.33×10^{-18}	NA	4.33×10^{-14}	163	20

Rank	Chemical name	CASRN	Number of wells using chemical (% of wells)	Log K_{ow} (unitless)		Water solubility estimate from log K_{ow} (mg/L @ 25°C)	Henry's Law Constant (atm m^3/mole @ 25°C)			Estimated volume, per disclosure (gal)	
				Estimated	Measured		Estimated, bond method	Estimated, group method 25	Measured	Mean	Median
15	2-Butoxyethanol	111-76-2	7,347 (21%)	0.57	0.83	6.447×10^4	9.79×10^{-8}	2.08×10^{-8}	1.60×10^{-6}	385	26
16	Solvent naphtha, petroleum, heavy arom.[b,c]	64742-94-5	7,108 (21%)	log K_{oc} = 3.2 to 2.7	NA	5.8 to 65	0.028 to 0.39 cm^3/cm^3	NA	NA	NA	NA
17	Naphthalene	91-20-3	6,354 (19%)	3.17	3.3	1.421×10^2	5.26×10^{-4}	3.7×10^{-4}	4.4×10^{-4}	72	12
18	2,2-Dibromo-3-nitrilopropionamide	10222-01-2	5,656 (16%)	1.01	0.82	2.841×10^3	6.16×10^{-14}	NA	1.91×10^{-8}	183	5
19	Phenolic resin	9003-35-4	4,961 (14%)	NA	NA	NA	NA	NA	NA	NA	NA
20	Choline chloride	67-48-1	4,741 (14%)	-5.16	NA	1.00×10^6	2.03×10^{-16}	NA	NA	2,131	290

[a] Hydrotreated light petroleum distillates (CASRN 64742-47-8) is a mixture of hydrocarbons in the C9 to C16 range.

[b] Physicochemical parameters are estimated using Gustafson et al. (1997). Parameters are presented as log K_{oc} (soil organic carbon-water partition coefficient), solubility (mg/L), and Henry's Law Constant (cm^3/cm^3).

[c] Heavy aromatic solvent naphtha (petroleum) (CASRN 64742-94-5) is mixture of aromatic hydrocarbons in the C9 to C16 range.

For the top 20 chemicals, many chemicals have high solubilities and negative or almost zero log K_{ow} (e.g., methanol, isopropanol, ethylene glycol). These chemicals are likely to travel quickly through the environment and could result in an immediate impact. Three chemicals, with larger log K_{ow} and smaller solubilities (distillates, petroleum, hydrotreated light; solvent naphtha, petroleum, heavy arom.; and naphthalene) may result in more severe impacts. These chemicals could associate with the soil particles, releasing into the groundwater at low concentrations slowly over time, and thus serve as long-term sources of contamination.

Mobility of a chemical is complex, and these numbers solely represent how a chemical behaves in an infinitely dilute aqueous solution, a simplifying approximation of the real world. Many factors can affect the fate and transport of a chemical, such as the transformation process (e.g., biodegradation), the presence of other chemicals, and site and environmental conditions. We discuss these factors in the next sections.

5.8.4 Transformation Processes

Once a chemical is released into the environment, it can transform or degrade. Understanding the processes governing these reactions in the environment is important to assessing potential impacts. The transformation of a chemical may reduce its concentration over time. Chemicals may completely degrade before reaching a drinking water resource. Transformation processes can be biotic or abiotic and may transform a chemical into a less or more harmful chemical.

One important transformation process is biodegradation. Biodegradation is a biotic process where microorganisms transform a chemical from its original form into another chemical. For example, the general biodegradation pathway of methanol is $CH_3OH \rightarrow CH_2O \rightarrow CHOOH \rightarrow CO_2$ or methanol → formaldehyde → formic acid → carbon dioxide (Methanol Institute, 2013).[1] This pathway shows how the original chemical transforms through a series of steps until it becomes the final product, carbon dioxide. Some chemicals are readily biodegraded, while others break down slowly over time. Biodegradation is a highly site-specific process, requiring nutrients, a carbon source, water, and an energy source. A highly biodegradable chemical could be persistent if the conditions for biodegradability are not met. Conversely, a chemical could biodegrade quickly under the right conditions, affecting its potential to impact a drinking water resource. The relationship between mobility and biodegradability is complex, and a variety of factors can influence a particular chemical's movement through the environment.

Abiotic processes, such as oxidation, reduction, photochemical reactions, and hydrolysis, can transform or break apart chemicals. The typical results are products that are more polar than the

[1] In methanol biodegradation, PQQ (pyrroloquinoline quinone) is a redox cofactor that goes from PQQ to $PQQH_2$ removing two hydrogen from methanol in the first step to form formaldehyde. Water is added to formaldehyde to provide the second oxygen to form formic acid. Nicotinamide adenine dinucleotide (NAD) is a coenzyme that takes up a hydrogen, going from NAD to $NADH^+$. This removes the hydrogen in the second and third steps, to result in carbon dioxide.

original compounds, and thus have different physicochemical properties (Schwarzenbach et al., 2002).[1]

5.8.5 Fate and Transport of Chemical Mixtures

Spills during the chemical mixing stage are often present as mixtures of chemicals. Additives are often mixtures of a few to several chemicals, possibly highly concentrated, and hydraulic fracturing fluids are often dilute mixtures of several additives. Chemical mixtures can act differently in the environment than individual chemicals. Individual chemicals can affect the fate and transport of other chemicals in a mixture primarily by changing their physicochemical properties and transformation rates.

Chemical mixtures can be more mobile than individual chemicals due to cosolvency, which increases solubility in the aqueous phase. Methanol and ethanol are examples of cosolvent alcohols used frequently in hydraulic fracturing fluids (U.S. EPA, 2015a). The presence of either greatly increases BTEX solubility (Rasa et al., 2013; Corseuil et al., 2011; Heermann and Powers, 1998).[2] By increasing solubility, ethanol can affect the fate and transport of other compounds. For example, BTEX has been observed to travel farther in the subsurface in the presence of ethanol (Rasa et al., 2013; Corseuil et al., 2011; Corseuil et al., 2004; Powers et al., 2001; Heermann and Powers, 1998).

The presence of surfactants lowers fluid surface tension and increases solubility of organic chemicals. Surfactants can mobilize less soluble/less mobile organic chemicals. Two common surfactants reported in disclosures in the EPA FracFocus 1.0 project database were 2-butoxyethanol (CASRN 111-76-2, 21% of disclosures) and poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture) (CASRN 127087-87-0, 20% of disclosures). Additionally, surfactants can mobilize bacteria in the subsurface, which can increase the impact of pathogens on drinking water resources (Brown and Jaffé, 2001).

When chemicals are present as mixtures, one chemical can decrease or enhance the biodegradability of another through inhibition or co-metabolism. The process of inhibition can slow biodegradation of each of the chemicals present. For example, the biodegradation of ethanol and methanol can slow the biodegradation rate of BTEX or other organic chemicals present (Rasa et al., 2013; Powers et al., 2001). Co-metabolism can increase the biodegradation rate of other chemicals. For example, when methane or propane is present with tetrachloroethylene, the enzyme produced by bacteria to degrade methane also degrades tetrachloroethylene (e.g., Alvarez-Cohen and Speitel, 2001 and references therein). For the purposes of chemicals used in hydraulic fracturing, the presence of other chemicals in additives and hydraulic fracturing fluids could result in increased or decreased biodegradation if the chemicals are spilled.

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[1] A polar molecule is a molecule with a slightly positive charge at one part of the molecule and a slightly negative charge on another. The water molecule, H_2O, is an example of a polar molecule, where the molecule is slightly positive around the hydrogen atoms and negative around the oxygen atom.

[2] BTEX is an acronym for benzene, toluene, ethylbenzene, and xylenes. These chemicals are a group of single ringed aromatic hydrocarbons based on the benzene structure. These compounds are found in petroleum and are of specific importance because of their potential health effects.

5.8.6 Site and Environmental Conditions

Environmental conditions at and around the spill site affect the movement and transformation of chemicals. This section discusses the following: site conditions (e.g., proximity, land cover, and slope), soil conditions (e.g., permeability and porosity), and weather and climate.

The proximity of a spill to a drinking water resource, either laterally in the case of a surface water body or downward for groundwater, affects the potential for impact and its severity. Land cover will affect how readily a fluid moves over land. For example, more rugged land cover such as forest can impede flow, and an asphalt road can facilitate flow. A spill that occurs on or near a sloped site can move overland faster, increasing the potential to reach nearby surface water. Flatter surfaces result in a greater chance for infiltration to the subsurface, which could increase the potential for groundwater impact.

Soil characteristics that affect the transport and transformation of spill chemicals include soil texture (e.g., clay, silt, sand), permeability, porosity, and organic content.[1,2] Fluids will move more quickly through permeable soil (e.g., sand) than through less permeable soil (e.g., clay). A soil with a high porosity provides more volume to hold water and spilled chemicals. Another important factor for a site is the organic content, of which there are two competing types: soil organic carbon and dissolved organic carbon. Each type of carbon acts as a strong substance for chemicals to associate with. Soil organic carbon present in a solid phase, such as dead and decaying leaves and roots, is not mobile and slows the movement of chemicals through the soil. Dissolved organic carbon (DOC) moves with the water and can act as a shuttling mechanism to mobilize less soluble chemicals across the surface and through the subsurface. Chemicals may also associate and move with particulates and colloids.

Weather and climate conditions affect the fate and transport of a spilled chemical. After a spilled chemical stops moving, precipitation can remobilize the chemical. The amount, frequency, and intensity of precipitation will impact the volume, distance, and speed of chemical movement. Precipitation can carry chemicals downward or overland, and it can cause erosion, which can move sorbed chemicals overland.

5.8.7 Peer-Reviewed Literature on the Fate and Transport of Hydraulic Fracturing Fluid Spills

There has been limited peer-reviewed research investigating the fate and transport of chemicals spilled at hydraulic fracturing sites. Aminto and Olson (2012) modeled a hypothetical spill of 1,000 gal (3,800 L) of hydraulic fracturing fluid using equilibrium partitioning. The authors evaluated how 12 chemicals typically used for hydraulic fracturing in the Marcellus Shale would partition among different phases: air, water, soil, and biota.[3] They presented a ranking of

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[1] Permeability of a soil describes how easily a fluid can move through the soil. Under a constant pressure, a fluid will move faster in a high permeability soil than the same fluid in a low permeability soil.

[2] Porosity of a soil describes the amount of empty space for a given volume of soil. The porosity describes how much air, water, or hydraulic fluid a given volume of soil can hold.

[3] The chemicals they investigated included: sodium hydroxide, ethylene glycol, 4,4-dimethyl oxazolidine, 3,4,4-trimethyl oxazolodine, 2-amino-2-methyl-1-propanol, formamide, glutaraldehyde, benzalkonium chloride, ethanol, hydrochloric acid, methanol, and propargyl alcohol.

concentrations for each phase. In water, they showed that sodium hydroxide (a pH buffer), 4,4-dimethyl oxazolidine (a biocide), hydrochloric acid (a perforation clean-up additive), and 3,4,4-trimethyl oxazolidine (a biocide) had the highest simulated water concentrations; however, these concentrations depended on the chemicals included in the simulated mixture and the concentrations of each. Their analysis suggested that after a spill, a large fraction of the spill would volatilize and leave the soil; however, some constituents would be left behind in the water, soil, and biota compartments, which could act as long-term contamination sources. Aminto and Olson (2012) only studied this one scenario. Other scenarios could be constructed with different chemicals in different concentrations. These scenarios may result in different outcomes and impacts. Any spill would require site- and spill-specific modeling on a case-by-case basis. For this reason, we cannot make any general statement about fate and transport of hydraulic fracturing chemicals and fluids. For this reason, we cannot make any general statement about fate and transport of hydraulic fracturing chemicals and fluids.

Drollette et al. (2015) suggested a link between surface spills and groundwater contamination, possibly from hydraulic fracturing activity, because the chemicals detected were hydraulic fracturing additives. This work demonstrates the pathway for surface spills to impact groundwater sources. They detected low levels of gasoline related organic chemicals with elevated diesel range organic chemicals, which suggests that the former were degraded or volatilized, while the latter were more persistent and penetrated into the subsurface and into groundwater.

5.8.8 Potential and Documented Fate and Transport of Documented Spills

There is limited information on the fate and transport of hydraulic fracturing fluids and chemicals. This section highlights both potential and documented impacts for three reported spills (U.S. EPA, 2015m). In each case, we provide the documented and potential paths (surface, subsurface, or combination) and the associated fate and transport governing processes by which the spill has been documented or has the potential to have an impact on drinking water resources. The three cases involve a tank overflow with a reported surface water impact, a human error blender spill with a reported soil impact, and an equipment failure that had no reported impact. We specifically chose these three spills to highlight three different cases. One demonstrates a documented impact with a demonstrated pathway that had an observed effect on a nearby drinking water resource. The second case shows how a release can impact an environmental receptor with a pathway for potential impact on a drinking water resource, but there was no observed impact. The third example is a spill that was contained and cleaned up resulting in likely no impact. None of these chemical releases have any documented pre- or post-sampling. No information on the specific chemicals spilled or the concentrations or total mass of any chemical is provided. We cannot provide any quantitative assessment from these observed cases.

In the first documented spill, shown in Figure 5-17, a tank overflowed twice, releasing a total of 7,350 gal (980 ft^3, 28 m^3, or 27,800 L) of friction reducer and gel (PA DEP, 2012, ID#1830163).[1] The spill traveled across the land surface, crossed a road, and then continued to a nearby stream. The

[1] We provide the total volume of the spill in gallons as well as cubic length (cubic feet and cubic meters), because it may be a little harder to visualize how far a volume of 7,300 gal (28,000 L) might travel.

spill affected wetlands and a stream, where fish were reported to have been killed. The fish kill indicates an observable impact. This represents a good example for how environmental conditions can affect the severity and timing of impact, due to the slope of the lands surface, the permeability of the soil, and the proximity to surface water. We are not aware of any measurements performed for soils, groundwater, surface water, sediments, or fish tissue. Based on the publicly available information, we do not know what chemicals were in the friction reducer and gel, which limits further assessment.

Figure 5-17. Fate and Transport Spill Example: Case 1.
Spills information from PA DEP (2012, ID#1830163).

For this first spill, the documented path was overland flow from the tank to the stream with a documented, immediate impact. There are also other potential paths for potential impacts on drinking water resources. The spilled chemicals could have penetrated into the soils or sorbed to soils and vegetation as the fluid moved across the ground towards the stream. Chemicals could then be mobilized during later precipitation, runoff, or erosion events. Chemicals that infiltrated the subsurface could serve as long-term sources, travel laterally across the unsaturated zone, or continue downwards to groundwater. Some chemicals could be lost to transformation processes. The absence of reported soil or groundwater sampling data prevents the ability to know if these potential paths occurred or not.

The second documented spill, shown in Figure 5-18, occurred when a cap was left off the blender, and 504 gal (70 ft^3 or 2 m^3) of biocide and hydraulic fracturing fluid were released (COGCC, 2012, ID#2608900). In addition, 294 gal (39 ft^3 or 1.1 m^3) were retained by a dike with a lined secondary containment measure, demonstrating the partial effectiveness of this containment mechanism. The remaining 210 gal (28 ft^3 or 0.8 m^3) of fluid (biocide and water) ran off-site. Of this, 126 gal were vacuumed, leaving 84 gal. There was no documented impact on surface or groundwater. However, potential impacts potentially could have occurred.

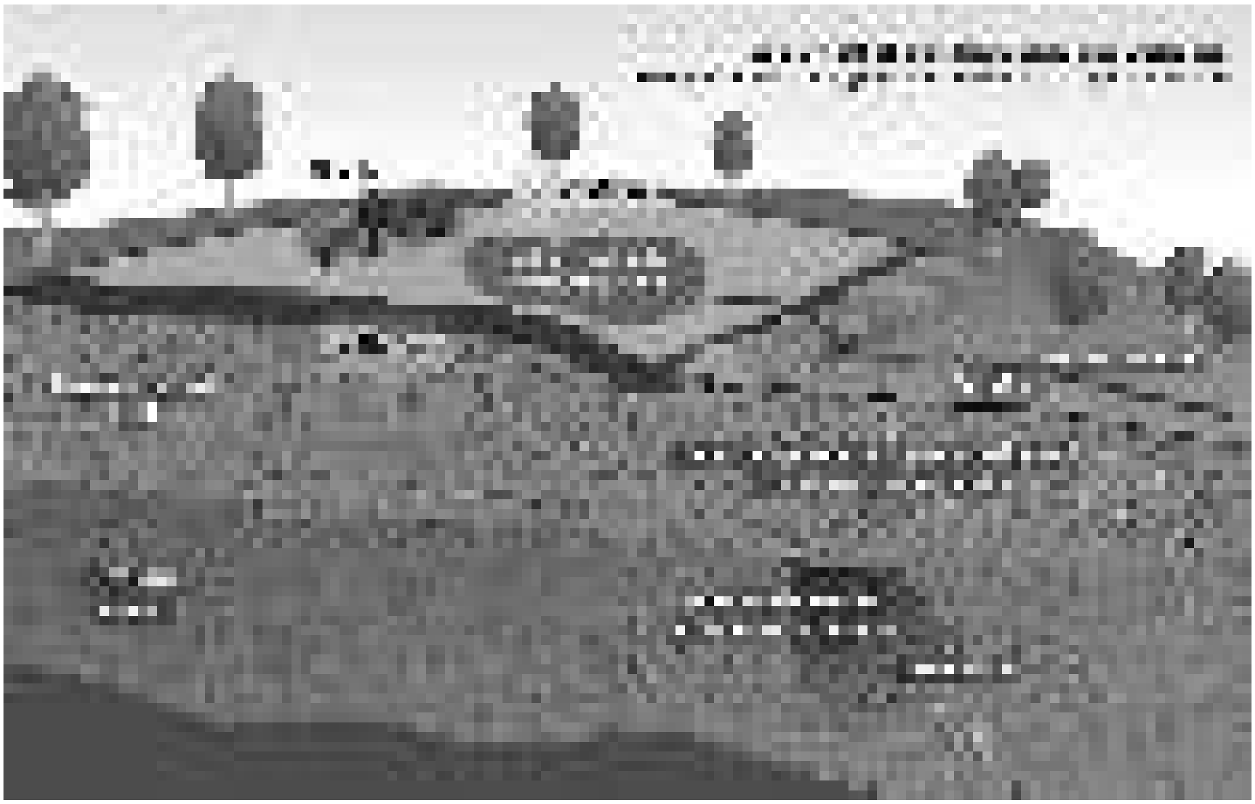

Figure 5-18. Fate and Transport Spill Example: Case 2.
Spills information from COGCC (2012, ID#2608900).

In this second case, the uncontained 84 gal could have infiltrated the subsurface, creating a potential path to groundwater. Highly mobile chemicals could have penetrated the soil more quickly than less mobile chemicals, which would have sorbed to soil particles. As the chemicals penetrated into the soil, some could have moved laterally in the unsaturated zone, or traveled downward to the groundwater table and moved with direction of groundwater flow. These chemicals could have served as a long-term contamination source. The chemicals also could have transformed into other chemicals with different physicochemical properties, and any volatile chemicals could have moved to the air as a loss process. As in the first case, there was no reported sampling of soil or groundwater, so there is no way to know if chemicals did or did not follow any of these pathways. We do not have any more information on the types of chemicals present or the concentrations with which they were present, which limits further assessment.

In the third documented spill, shown in Figure 5-19, 630 gal (84 ft^3 or 2.4 m^3) of crosslinker spilled onto the well pad when a hose wore off at the cuff (COGCC, 2012, ID#1395827). The spill was contained in the berm and an on-site vacuum truck was used to clean up the spill. No impact on soil or water was reported.

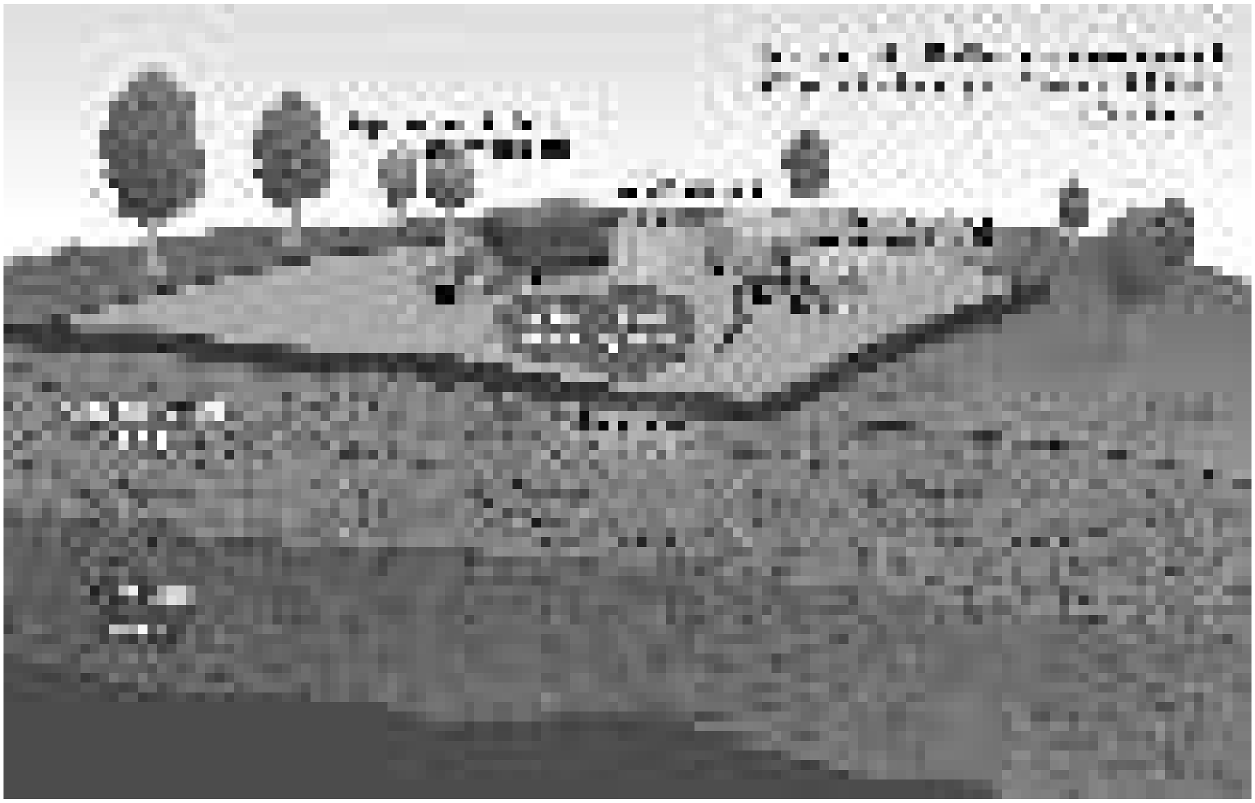

Figure 5-19 Fate and Transport Spill Example: Case 3.
The pad may or may not have had a liner. Spills information from COGCC (2012, ID#1395827).

For this third case, we do not have any information on whether the well pad was lined or not. If the site had a liner, the spill could have been fully contained and cleaned up. Without a liner or if the integrity of the liner was compromised (e.g., had a tear), any residual chemical that was not effectively cleaned up could have remained in the soil. This would create potential paths similar to those above in the second case, where the chemicals could have sorbed to the soils and penetrated into the subsurface and possibly reach groundwater. There was no reported sampling of soil or groundwater to determine whether or not chemicals migrated into the soil, and we do know the types of chemicals or the concentrations of the released chemicals.

5.8.9 Challenges with Unmonitored and Undetected Chemicals

One of the challenges confronting a thorough assessment of the fate and transport of spilled hydraulic fracturing chemicals lies in the lack of documented observations. It is difficult to prove absence of impact, and absence of observations does not necessarily imply lack of impact. Also, we know there are over 1,000 different chemicals reported used in hydraulic fracturing (Section 5.4), and this number is increasing. For many chemicals, there is not an analytical technique available to detect them in samples taken to a laboratory. Due to the lack of information on the chemicals used on site (some of which are claimed as CBI), one would not know what chemicals to include in the lab analysis. Hydraulic fracturing chemicals are typically present as complex mixtures, which also complicates sample analysis. Chemicals can transform upon release, which can result in different chemicals in the environment than those originally released. Even if chemicals are detected on-site, it can be difficult to demonstrate a direct linkage to hydraulic fracturing operations, since many of the chemicals used in hydraulic fracturing are also used for other purposes (such as gasoline or diesel from vehicles). Since there are currently no requirements for a detection–monitoring network to assess the occurrence and extent of chemical releases from the well pad, it is not possible to conclusively assess the frequency and impact of fluid releases during the chemical mixing process.

5.9 Trends in the Use of Hydraulic Fracturing Chemicals

Hydraulic fracturing science and engineering continues to advance. A part of this research includes using different chemicals. This section provides an overview of the changes in chemical use, with an emphasis on efforts to reduce potential impacts from surface spills by using fewer and safer chemicals. Reasons for changing the types of chemicals used can include: improving the fracturing process, using greener/safer chemicals, and reducing overall cost.

Representatives from oil and gas companies, chemical companies, and non-profits are working on strategies to reduce the number and volume of chemicals used and to identify safer chemicals (Waldron, 2014). Southwestern Energy Company, for example, is developing an internal chemical ranking tool (SWN, 2014), and Baker Hughes is working on a hazard ranking system designed for wide-scale external use (Baker Hughes, 2014; Brannon et al., 2012; Daulton et al., 2012; Brannon et al., 2011). Environmental groups, such as the Environmental Defense Fund, are also developing hazard rating systems (Penttila et al., 2013). Typical criteria used to rank chemicals include mobility, persistence, biodegradation, bioaccumulation, toxicity, and hazard characteristics. In this assessment, toxicity and a methodology to rank chemical hazards of hydraulic fracturing chemicals is discussed in Chapter 9.

Given that human error is the cause of 25% of chemical mixing related spills and spill prevention can never be 100% effective, changes to the types of chemicals used could reduce the frequency or the severity of potential impacts. Using chemicals with specific physicochemical properties that affect the fate and transport of chemicals could reduce their potential impacts. Less mobile chemicals could make cleanup of spills easier. For example, using dry chemicals that are hydrated on-site could minimize impacts if there were a container failure. Using chemicals with lower persistence and higher biodegradability, if spill prevention and cleanup are not fully effective, would lessen the severity of potential impact. Use of less hazardous chemicals could lessen impact in cases where a spill reaches a drinking water resource.

The EPA has not conducted a comprehensive review of efforts to develop safer hydraulic fracturing chemicals. However, the following are some specific examples of efforts that companies cite as part of their efforts toward safer chemical use:

- A renewable citrus-based replacement for conventional surfactants (Fisher, 2012);

- A crosslinked gel system comprised of chemicals designated as safe food additives by the U.S. Food and Drug Administration (Holtsclaw et al., 2011);
- A polymer-free gel additive (Al-Ghazal et al., 2013);
- A dry, hydrocarbon-free powder to replace liquid gel concentrate (Weinstein et al., 2009);
- Biodegradable polymers (Irwin, 2013);
- The use of ultraviolet light to control bacteria (Rodvelt et al., 2013);
- New chelating agents that reduce the use of strong acids (LePage et al., 2013);
- Eco-friendly viscoelastic surfactant (VES) polymer-free fluid reduces fracture cleanup time with 95% retrieved fluids compared to 40 – 60% and is less toxic than polymer-based fluids (AlKhowaildi et al., 2016); and
- The recovery and reuse of produced water as hydraulic fracturing fluids, which can reduce the need to add additional chemicals (Horn et al., 2013).

A review of the EPA's new chemicals program found that, from 2009 to April 2015, the Agency received pre-manufacturing notices (PMN) for about 110 chemicals that have the potential for use as additives. Examples include chemicals intended for use as clay control agents, corrosion inhibitors, gel crosslinkers, emulsifiers, foaming agents, hydrate inhibitors, scale inhibitors, and surfactants. At the time of PMN submission, these chemicals were not in commercial use in the United States. As of April 2015, the EPA had received 30 notices of commencement, indicating that some of the chemicals are now used commercially.

As different hydraulic fracturing fluids are developed, they have corresponding effects on different stages of the hydraulic fracturing water cycle. For example, in Figure 5-4(b) an example of an energized fluid uses a total water volume of 105,000 gal (397,000 L), which means less water is required in the water acquisition stage and less produced water results in less wastewater. Figure 5-4(a) shows slickwater with 4,763,000 gal (18,030,000 L) of water, yet a larger fraction of slickwater may be reused, reducing the need for more water for another frac job and requiring the treatment of less wastewater.

5.10 Synthesis

The chemical mixing stage includes the mixing of base fluid, proppant, and additives on the well pad to make hydraulic fracturing fluid. This chapter provided an analysis of the factors affecting potential impacts on drinking water resources during the chemical mixing stage of the hydraulic fracturing water cycle and the factors governing the frequency and severity of these impacts.

5.10.1 Summary of Findings

Reports have demonstrated that spills and releases of chemicals and fluids have occurred during the chemical mixing stage and have reached soils and surface water receptors. Spill reports have not documented impacts on groundwater related to the chemical mixing stage. Spill reports have little information on post-spill testing and sampling. Impacts on groundwater may remain undocumented. The potential pathway for impact on groundwater has been demonstrated and documented for chemicals spilled during other parts of the hydraulic fracture water cycle. (Evidence of groundwater impact from produced water spills is discussed Chapter 7.)

The hydraulic fracturing fluid generally consists of a base fluid (typically water), a proppant (typically sand), and additives (chemicals), although there is no standard or single composition of hydraulic fracturing fluid used. According to the analysis of the EPA FracFocus 1.0 project database, based on FracFocus disclosure data from January 2011 to February 2013, approximately 93% of hydraulic fracturing fluids use water as a base fluid. Non-aqueous fluids, such as nitrogen, carbon dioxide, and hydrocarbons, are also used as base fluids or used in combination with water as base fluids. The number of chemicals injected into a well typically ranges from 4 to 28, with a median of 14 (U.S. EPA, 2015a). In water-based hydraulic fracturing, the composition, by volume, of a typical hydraulic fracturing fluid is 90% to 97% water, 2% to 10% proppant, and 2% or less additives (Carter et al., 2013; Knappe and Fireline, 2012).

The EPA has identified 1,084 different chemicals used in chemical mixing. A recent study of FracFocus disclosure data, covering January 2011 to April 2015, has reported 263 new CASRNs, increasing the number of chemicals identified for use by approximately 24% (Konschnik and Dayalu, 2016). Hydraulic fracturing chemicals cover a wide range of chemical classes and a wide range of physicochemical properties. The chemicals include acids, aromatic hydrocarbons, bases, hydrocarbon mixtures, polymers, and surfactants. The use of 32 chemicals, excluding water, quartz, and sodium chloride, is reported in 10% or more of disclosures in the EPA FracFocus 1.0 project database. The ten most common chemicals (excluding quartz) are methanol, hydrotreated light petroleum distillates, hydrochloric acid, isopropanol, ethylene glycol, peroxydisulfuric acid diammonium salt, sodium hydroxide, guar gum, glutaraldehyde, and propargyl alcohol (U.S. EPA, 2015c). These chemicals can be present in multiple additives. Methanol, hydrotreated light petroleum distillates, and hydrochloric acid are the three chemicals reported to be used in more than half of all hydraulic fracturing jobs, with methanol being used at 72% of all sites.

An EPA analysis of spills data (January 2006 to April 2012, from nine states, nine service companies, and nine operators) identified over 36,000 spills, with 457 spills (~1%) that were on or near the well pad and definitively associated with hydraulic fracturing. Of these spills, 151 were of chemicals or hydraulic fracturing fluid and thus assumed to be associated with the chemical mixing stage. Chemical spills during the chemical mixing stage were primarily caused by equipment failure (34%), followed by human error (25%), although 26% spills had an unknown source. The remaining spills were caused by a failure of container integrity, weather, vandalism, and well communication. Reported spills covered a large range of volumes, from 5 to 19,320 gal (19 to 73,130 L), with a median of 420 gal (1,600 L) (U.S. EPA, 2015m).

The rate of reported spills during the hydraulic fracturing water cycle is estimated to range from 0.4 to 12.2 reported spills for every 100 wells, based on spills data from North Dakota, Pennsylvania, and Colorado, with a median rate of 2.6 reported spills for every 100 wells (See Appendix C). The estimated rates provide an approximate estimate of the potential frequency of the number of spills at a site. It is uncertain how representative these rates are of national spill rates or rates in other states. These numbers are not specific to the chemical mixing stage. In 2015, there are 2.6 reported spills occurring during the chemical mixing stage per 100 wells hydraulically fractured in North Dakota.

The total volume of chemicals used on site are estimated to range from 2,600 to 30,000 gal (9,800 to 114,000L). An estimate for the mean volume for any chemical used on-site is 650 gal (2,500 L) with a mean mass of 1500 kg (3,200 lb). An estimate of 2,300 to 6,500 gal (8,800 to 25,000 L) of additives are stored on site, typically in multiple totes of 200 to 375 gal (760 to 1,420 L). These volumes provide insight on how much potentially could spill at any given hydraulic fracturing site and what the volume of a spill might be depending on where/when it occurs during the chemical mixing process.

The potential of spills to reach drinking water resources depends on site and chemical properties. The fate and transport of spilled hydraulic fracturing chemicals is complex, particularly because chemicals are generally present as diverse, complex mixtures. There are different pathways for a spill to reach ground and surface water and to serve as a long term source. Roughly 40% of hydraulic fracturing chemicals are organic chemicals, which have physicochemical properties that cover the parameter space, from fully miscible to insoluble and from highly hydrophobic to highly hydrophilic. Of the 20 most frequently used chemicals used at hydraulic fracturing sites, three chemicals have low mobility: hydrotreated light petroleum distillates, heavy aromatic petroleum solvent naphtha, and naphthalene. These chemicals have the potential to act as long term sources of contamination if spilled on-site.

5.10.2 Factors Affecting the Frequency or Severity of Impacts

The specific factors that have the potential to affect the frequency and severity of impacts include the size and type of the fracturing operation; volume, mass, and concentration of chemicals spilled; type of chemicals and their properties; combination of chemicals spilled; environmental conditions; proximity to drinking water resources; employee training and experience; quality and maintenance of equipment; and spill containment and mitigation.

The size and type of a fracturing operation, including the number of wellheads, the depth of the well, the length of the leg(s), and the number of stages and phases, affect the potential frequency and severity spills. Larger operations can require larger volumes of chemicals, more storage containers, more equipment, and additional transfers between different pieces of equipment. Larger storage containers increase the maximum volume of a spill or leak from a storage container. Additional transfers between equipment increase the possibility of human error and potential frequency of spills.

The volume, mass, and concentration of spilled chemicals affect the frequency and severity of impacts. A larger volume increases the potential for a spill to travel a longer distance and reach a drinking water resource. The severity of the spill will be affected by the spill volume, the total mass of chemicals released, and the concentration with which it reaches the drinking water resource.

The type of chemicals spilled affects how the chemicals will move and transform in the environment and the type of impact it will have on a drinking water resource. More mobile chemicals move faster through the environment, which can increase the frequency of impact. More soluble chemicals can reach a drinking water resource at higher concentrations, thereby increasing the potential severity of an impact. Less mobile chemicals will move more slowly, and can have delayed and longer-term impacts at lower concentrations. The potential severity of impact is affected by how the chemical adversely impacts water quality. Some chemicals can have severe impacts at low concentrations, while some chemicals can have minimal impacts even at high concentrations. Water quality impacts can range from aesthetic effects (e.g., taste, smell) to adverse health effects.

The environmental conditions at and around the spill site affect the fate and transport of a given chemical and thus affect the frequency of impacts as well as potential severity. Conditions include soil properties, climate, weather, and terrain. Permeable soils allow for rapid transport of the spilled fluid through the subsurface and to groundwater. The presence of preferential flow paths (e.g., fractures, animal burrows) may provide rapid transport through the subsurface in what might appear to have low permeability. The presence of complexing agents and colloids may further increase transport of less soluble chemicals. Precipitation can re-mobilize trapped chemicals and move them over land or through the subsurface.

The proximity of a spill to drinking water resources affects the frequency and severity of impact. The closer a spill is to a drinking water resource, the higher the potential to reach it. As a fluid moves toward a drinking water resource, it can decrease in concentration, which can reduce the severity of an impact. The characteristics of the drinking water resource will also influence the severity of the impact of a spill. For example, a slow release into a fast moving stream will result in large dilution and lower concentrations of chemicals (less severe impact). The transport of a chemical to groundwater may have a more severe impact, as there may be less dispersion of the chemical (higher concentrations in the groundwater, more severe impact) and the chemical could serve as a long-term source of contamination (resulting in a chronic exposure versus an acute exposure).

Effective spill containment and mitigation measures can prevent or reduce the frequency and severity of impacts. Spill containment measures include well pad containment liners, diversion ditches, berms, dikes, overflow prevention devices, drip pans, and secondary containers. These may prevent a spill from reaching soil and water receptors. Spill mitigation, including removing contaminated soils, vacuuming up spilled fluids, and using sorbent materials can limit the severity of a spill. It is unclear how effective these practices are and to what extent they are implemented.

5.10.3 Uncertainties

The lack of information and the uncertainty around information having to do with the composition of additives and fracturing fluids, containment and mitigation measures in use, the proximity of chemical mixing to drinking water resources, and the fate and transport of spilled fluids limits our ability to fully assess potential impacts on drinking water resources and the factors affecting their frequency and severity.

There is no standard design for hydraulic fracturing fluids. Detailed information on the chemicals used is limited. Volumes, concentrations, and mass, as well as the identity of some of chemicals stored on-site, are generally not publicly available. The FracFocus national registry, which currently holds the most comprehensive information on water and chemicals used in hydraulic fracturing fluids, is structured so as to input chemical information as a maximum percentage of the mass of fracturing fluid and the given additive. This does not provide exact information on the volume of a chemical, the mass of a chemical, or the actual composition of an additive. The accuracy and completeness of original FracFocus disclosure information has not been verified. In applying the EPA-standardized chemical list to the ingredient records in the EPA FracFocus 1.0 project database, standardized chemical names were assigned to only 65% of the ingredient records from the more than 36,000 unique, fully parsed disclosures. The remaining ingredient records could not be assigned a standardized chemical name and were excluded from analyses (U.S. EPA, 2015a).

Operators may specify certain ingredients as confidential business information (CBI) and not disclose the chemical used. More than 70% of disclosures in the EPA FracFocus 1.0 project database contained at least one CBI chemical. Of disclosures with at least one CBI chemical, the average number of CBI chemicals per disclosure was five. Approximately 11% of all chemicals reported in the disclosures in the EPA FracFocus 1.0 project database were reported as CBI (U.S. EPA, 2015a). The rate of withholding in FracFocus 2.0 data has increased to 16.5% (Konschnik and Dayalu, 2016). No data are available in FracFocus disclosures for any chemical listed as CBI. Therefore, chemicals identified as CBI in FracFocus disclosures are not included in any of the analyses in this assessment including estimates of chemical volume, physicochemical properties, or frequency of use. It is feasible that the same chemicals are repeatedly reported as CBI. Each reported CBI chemical could also be unique, which would mean there is a very large number of chemicals that we know nothing about. This results in an unknown amount of uncertainty regarding CBI chemicals and their potential impact on drinking water resources.

Of the 1,084 hydraulic fracturing fluid chemicals identified by the EPA, 629 were inorganic chemicals, mixtures, or polymers, and thus they did not have estimated physicochemical properties reported in the EPI Suite™ database. Knowing the chemical properties of a spilled fluid is essential to predicting how and where it will travel in the environment. Although we can make some generalizations about the physicochemical properties of these chemicals and how spilled chemicals may move in the environment, the distribution of properties could change if we obtained data for all known fracturing fluid chemicals (as well as for those listed as CBI).

There has been limited research on the fate and transport of spilled chemicals on site. We have provided a limited overview discussing the processes that may be important, but the processes are complex. There is great uncertainty in how these chemicals will move in the environment. These processes are complicated by the data gaps in fluid characteristics, especially present in mixtures, and there is limited understanding on how chemicals act in mixture in the environment. Hydraulic fluid mixtures are different than other previously studied mixtures (like petroleums, coal tars, and polychlorinated biphenyls (PCBs). Those mixtures are of chemicals of similar classes, while hydraulic fracturing fluids are chemicals covering a range of different chemical classes.

There is a lack of field data at hydraulic fracturing sites. There is a lack of baseline ground and surface water quality data. This lack of data limits our ability to assess the relative change to water quality from a spill or attribute the presence of a contaminant to a specific source. There is a lack of publicly or readily accessible sampling of soils and groundwater after a fracturing job is complete. The lack of data and uncertainty on what chemicals are used for hydraulic fracturing makes it unclear what chemicals to measure. Further uncertainty lies in the limited analytical techniques for chemicals used in hydraulic fracturing.

There are uncertainties and data gaps in the current information on spills. The EPA spills report included data from January 2006 to April 2012 from nine states, nine service companies, and nine oil and gas production well operators (U.S. EPA, 2015a). This data contained over 36,000 reported spills. From this data set, only 457 were determined to be definitively associated with hydraulic fracturing and occurred on or near the well pad. With these data, it is impossible to know if all these spill reports capture all spills occurring at hydraulic fracturing sites. The available data might not extrapolate to the rest of the nation. Spill reports had limited information on spill causes, containment and mitigation measures, and sources of spills. The actual chemicals spilled, the total mass, and the composition are generally not included. There are little available data on impacts of spills, due to a lack of baseline data and incomplete documentation of follow-up actions and testing.

In general, then, we are limited in our ability to fully assess potential impacts on drinking water resources from chemical spills, based on current available information. To improve our understanding we need: more information on the chemical composition of additives and fracturing fluids and the physicochemical properties of chemicals used; baseline monitoring and field studies of spilled chemicals; ground and surface water drinking water resources located and identified, with quality conditions performed before and after hydraulic fracturing; detailed site-specific environmental conditions; more information on containment and mitigation measures and their effectiveness; and more detail on the characteristics of spills, such as the exact chemicals and the amount spilled (mass, concentration, volume).

5.10.4 Conclusions

This chapter discusses the factors that affect the potential for the chemical mixing stage of the hydraulic fracturing water cycle to impact drinking water resources. Reports have demonstrated that spills and releases of chemicals and fluids have occurred during the chemical mixing stage and have reached soils and surface waters with the potential to reach groundwater. The potential for spilled fluids to reach, and therefore impact, ground or surface water resources depends on the composition of the spilled fluid, spill characteristics, spill response activities, and the fate and transport of the spilled fluid. There is no standard composition for a hydraulic fracturing fluid, which consists of base fluid, proppant, and additives. The EPA identified 1,084 chemicals that have been reported to be used nationwide, and these chemicals cover a wide variety of chemical classes and physicochemical properties, and this number is increasing. These chemicals cover a range of classes and physicochemical properties. The type of fluid and the number, volume, and type of chemicals used vary from site to site. Hydraulic fracturing fluids generally consist of a mixture of chemicals, which affects the potential for a release to reach a drinking water resource and the severity of the potential impact. State and industry spill data collected and reviewed by the EPA and others indicate that small (approximately 30 gal or 100 L) and large spills (greater than 1,000 gal or 4,000 L) can reach surface water resources. While small spills have reached surface water resources (and have the potential to reach groundwater resources), large volume spills are more likely to travel longer distances and thus have a greater potential to reach ground and surface water resources. Large volume spills, particularly of concentrated additives, also have a greater potential to result in more severe impacts on drinking water resources, because they can deliver a large quantity of potentially hazardous chemicals to ground or surface water resources.

Chapter 6. Well Injection

Abstract

The well injection stage of the hydraulic fracturing water cycle involves the injection of hydraulic fracturing fluids through the oil and gas production well and their movement in the production zone. Subsurface pathways created during this stage—including the production well and newly created fractures—can allow hydraulic fracturing fluids or naturally occurring fluids to reach groundwater resources.

This chapter examines two types of pathways by which hydraulic fracturing fluids and liquids and/or gases that exist in the subsurface can move to, and affect the quality of, subsurface drinking water resources. First, fluids can move via pathways adjacent to or through the production well as a result of inadequate design, construction, or degradation of the casing or cement. Second, fluid movement can occur within the subsurface geologic formations via fractures extending out of oil/gas-containing formations, by intersecting abandoned or active offset wells, or via naturally occurring faults and fractures.

The primary factors that can affect the frequency or severity of impacts to drinking water associated with injection for hydraulic fracturing are: (1) the condition of the well's casing and cement and their placement relative to drinking water resources, (2) the vertical separation between the production zone and formations that contain drinking water resources, and (3) the presence/proximity and condition of wells near the hydraulic fracturing operation.

We identified two cases where hydraulic fracturing activities affected the quality of drinking water resources due to well construction issues, including inadequate cement or ruptured casing. Additionally, there are places where oil and gas reservoirs and drinking water resources co-exist in the same formation and hydraulic fracturing operations occur, which results in the introduction of hydraulic fracturing fluids into the drinking water resource. There are other cases involving the migration of stray gas where hydraulic fracturing could be a contributing cause to impacts on drinking water resources.

While there is evidence that these pathways have formed and that groundwater quality has been impacted, there are limited nationally available data on the performance of wells used in hydraulic fracturing operations, pre- and post-hydraulic fracturing groundwater quality, and the extent of the fractures that develop during hydraulic fracturing operations.

These data limits, in combination with the geologic complexity of the subsurface environment and the fact that these processes cannot be directly observed, make determining the frequency of such impacts challenging.

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6. Well Injection

6.1 Introduction

In the well injection stage of the hydraulic fracturing water cycle, hydraulic fracturing fluids (primarily water, mixed with the types of chemicals and proppant described in Chapter 5) are injected into a well under pressure.[1] These fluids flow under pressure through the well, then exit the well and move into the formation, where they create fractures in the rock. This process is also known as a fracture treatment or a type of stimulation.[2] The fractures, which typically extend hundreds of feet away from the well, are designed to remain within the production zone to access as much oil or gas as possible by using an appropriate amount of water and chemicals to complete the operation.[3]

Production wells are sited and designed primarily to optimize production of oil or gas, which requires isolating water-bearing formations from hydrocarbon-bearing formations in order to prevent the water from diluting the hydrocarbons and to protect drinking water resources.[4] However, problems with the well's components or improperly sited, designed, or executed hydraulic fracturing operations (or combinations of these) could adversely impact the quality of drinking water resources. (Note that, due to the subsurface nature of activities in the well injection stage, the drinking water resources that may be directly impacted are groundwater resources; see Chapter 2 for additional information about groundwater.[5])

The well and the geologic environment in which it is located are a closely linked system. Wells are often designed with multiple barriers (i.e., isolation afforded by the well's casing and cement and the presence of subsurface rock formations) to prevent fluid movement between oil/gas zones and drinking water resources. Therefore, this chapter discusses (1) the well (including its construction and operation) and (2) the characteristics of or features in the subsurface geologic formations that could provide or have provided pathways for migration of fluids to drinking water resources. If present, and in combination with the existence of a fluid and a physical force that moves the fluid, these pathways can lead to impacts on the quality of drinking water resources throughout the life of the well, including during and after hydraulic fracturing.[6]

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[1] A fluid is a substance that flows when exposed to an external pressure; fluids include both liquids and gases.

[2] In the oil and gas industry, "stimulation" has two meanings—it refers to (1) injecting fluids to clear the well or pore spaces near the well of drilling mud or other materials that block or inhibit optimal production (i.e., matrix treatment) and (2) injecting fluid to fracture the rock to optimize the production of oil or gas. This chapter focuses on the latter.

[3] The "production zone" (sometimes referred to as the target zone or the targeted rock formation) refers to the portion of a subsurface rock zone that contains oil or gas to be extracted (sometimes using hydraulic fracturing). "Producing formation" refers to the larger geologic unit in which the production zone occurs.

[4] A subsurface formation (or "formation") is a mappable body of rock of distinctive rock type(s) and characteristics (such as permeability and porosity) with a unique stratigraphic position.

[5] Government agencies and other organizations use a variety of terms to describe potable groundwater and groundwater resources. In this chapter, we use the general term "groundwater resources" to refer to drinking water resources that occur underground. However, other terms are used in specific contexts to reflect the language used in cited materials.

[6] The primary physical force that moves fluids within the subsurface is a difference in pressure. Fluids move from areas of higher pressure to areas of lower pressure when a pathway exists. Density-driven buoyancy may also serve as a driving force; see Section 6.3 for more information.

Fluids can move via pathways adjacent to or through the production well that are created in response to the stresses exerted during hydraulic fracturing operations if the well is not able to withstand these stresses (Section 6.2). While wells are designed and constructed to isolate fluids and maximize the production of oil and gas, inadequate construction or degradation of the casing or cement can allow fluid movement that can impact drinking water quality. Potential issues associated with wells may be related to the following:

- Inadequate or degraded casing. This may be influenced by the number of casing strings and the depths to which they are set, compatibility with the geochemistry of intersected formations, the age of the well, whether re-fracturing is performed, and other operational factors.
- Inadequate or degraded cement. This may be influenced by a lack of cement in key subsurface intervals, poor-quality cement, improperly placed cement, or degradation of cement over time.

Fluid movement can also occur via induced fractures and/or other features within subsurface formations (Section 6.3). While the hydraulic fracturing operation may be designed so that the fractures will remain within the production zone, it is possible that, in the execution of the hydraulic fracturing treatment, fractures can extend beyond their designed extent. Four scenarios associated with induced fractures may contribute to fluid migration or communication between zones:

- Flow of injected and/or displaced fluids through pore spaces in adjacent rock formations out of the production zone due to pressure differences and buoyancy effects.
- Fractures extending out of oil/gas formations into drinking water resources or zones that are in communication with drinking water resources or fracturing into zones containing drinking water resources.
- Fractures intersecting artificial structures, including active (producing) or inactive offset wells near the well that is being stimulated (i.e., well communication) or abandoned or active mines.
- Fractures intersecting geologic features that can act as pathways for fluid migration, such as existing permeable faults and fractures.

This chapter describes the conditions that can contribute to or cause the development of the pathways listed above, the evidence for the existence of these pathways, examples of impacts on the quality of drinking water resources associated with these pathways that have been documented in the literature, and the factors that can affect the frequency or severity of those impacts. (See Chapter 10 for a discussion of factors and practices that can reduce the frequency or severity of impacts to drinking water quality.)

The interplay between the well and the subsurface features is complex and not directly observable; therefore, sometimes it is not possible to identify what specific element is contributing to or is the primary cause of an impact on drinking water resources. For example, concerns have been raised regarding stray gas detected in groundwater in natural gas production areas (for additional information about stray gas, see Sections 6.2.2 and 6.3.2.4).[1] Stray gas migration is a technically complex phenomenon, because there are many potential naturally occurring or artificially created routes for migration of gas into aquifers, including along production wells and via naturally existing or induced fractures. It is also challenging to determine the source of the natural gas and whether the mobilization is related to oil or gas production activities.

Furthermore, identifying cases where contamination of drinking water resources occurs due to oil and gas production activities—including hydraulic fracturing operations—requires extensive amounts of site and operational data, collected before and after hydraulic fracturing operations. (See Section 6.4 for additional information on data limitations.) Where such data do exist and provide evidence of contamination, we present it in the following sections. We do not attempt to predict which of these pathways is most likely to occur or to lead to a drinking water impact, or the magnitude of an impact that might occur as a result of migration via any single pathway, unless the information is available and documented based on collected data. However, a qualitative assessment of the factors that can affect the frequency or severity of impacts on drinking water quality associated with the well injection stage is possible; see Section 6.4.

6.2 Fluid Migration Pathways Within and Along the Production Well

In this section, we discuss pathways for fluid movement along or through the production well used in the hydraulic fracturing operation. While these pathways can form during other times within the life of an oil and gas well, the repeated high pressure stresses exerted during hydraulic fracturing operations can make maintaining the mechanical integrity of the well more difficult (Council of Canadian Academies, 2014).[2] Section 6.2.1 presents the purpose of the various well components and typical well construction configurations. Section 6.2.2 describes the pathways for fluid movement that can potentially develop within the production well and wellbore and the conditions that lead to pathway development, either as a result of the original design of the well, degradation over time or use, or hydraulic fracturing operations.

While we discuss casing and cement separately, it is important to note that these are related—inadequacies in one of these components can lead to stresses on the other. For example, flaws in cement may expose the casing to corrosive fluids. Furthermore, casing and cement work together in the subsurface to form a barrier to fluid movement, and it may not be possible to distinguish whether mechanical integrity problems are related to the casing, the cement, or both. For additional information on well design and construction, see Appendix D.

6.2.1 Overview of Well Construction

Production wells are constructed to transport hydrocarbon resources from the reservoirs in which they are found to the surface. They are also used to isolate fluid-bearing zones (containing oil, gas,

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[1] Stray gas refers to the phenomenon of natural gas (primarily methane) migrating into shallow drinking water resources or to the surface.

[2] Mechanical integrity of a well refers to the absence of significant leakage within the injection tubing, casing, or packer (referred to as internal mechanical integrity) or outside of the casing (referred to as external mechanical integrity).

or fresh water) from each other. Multiple barriers (i.e., casing and cement) are often present, and they act together to prevent both horizontal fluid movement (in or out of the well) and vertical fluid movement (along the wellbore from deeper oil- or gas-bearing formations to drinking water resources). Proper design and construction of the casing, cement, and other well components in the context of the location of drinking water resources and maintaining mechanical integrity throughout the life of a well are necessary to prevent migration of hydraulic fracturing fluids and formation fluids into drinking water resources.

A well is a multiple-component system that typically includes casing, cement, and a completion assembly, and it may be drilled vertically, horizontally, or in a deviated orientation (Figure 6-1).[1,2] These components work together to prevent unintended fluid movement into, out of, or along the well. Due to the presence of multiple barriers within the well and the geologic system in which it is placed, the existence of a pathway for fluid movement through a component of this system does not necessarily mean that an impact on a drinking water resource has occurred or will occur.

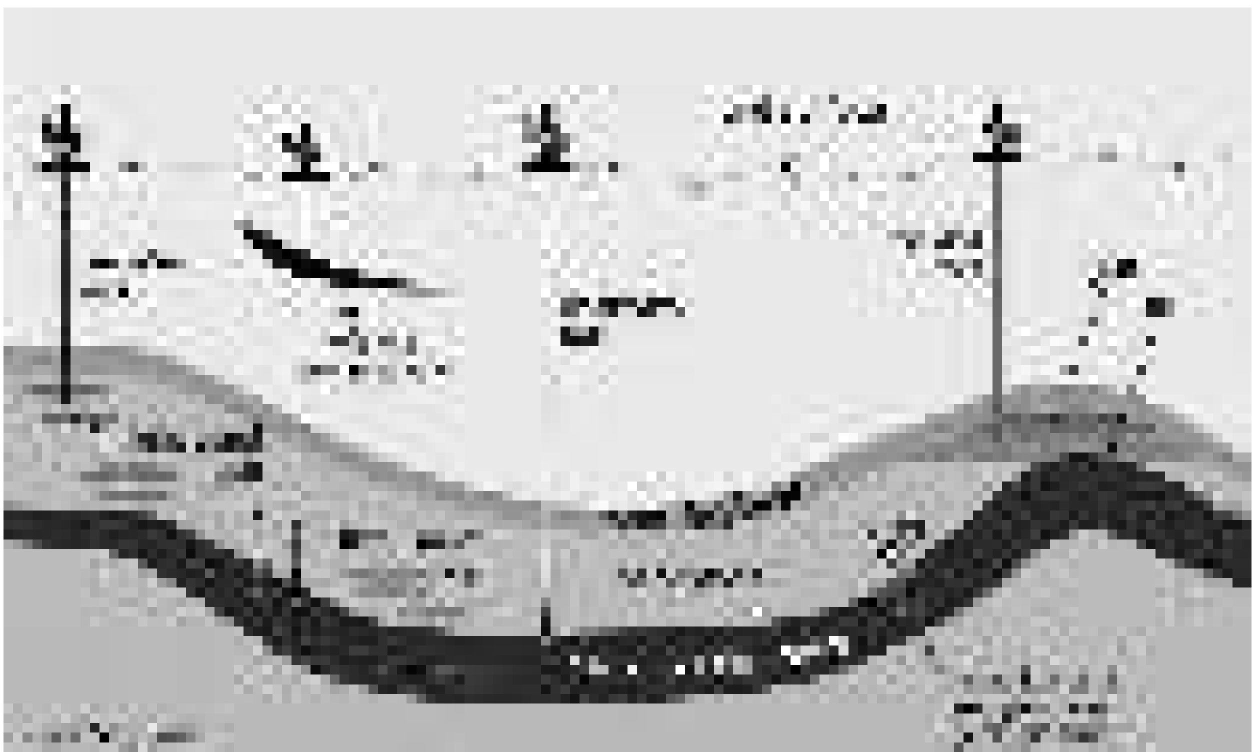

Figure 6-1. Schematic cross-section of general types of oil and gas resources and the orientations of production wells used in hydraulic fracturing.

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[1] Completion is a term used to describe the assembly of equipment at the bottom of the well that is needed to enable production from an oil or gas well. It can also refer to the activities and methods (including hydraulic fracturing) used to prepare a well for production following drilling.

[2] For the purposes of this assessment, a well's orientation refers to its inclination from verticality. Wells drilled straight downward are considered to be vertical, wells drilled directionally to end up parallel to the production zone's bedding plane are considered horizontal, and directionally drilled wells that are neither vertical nor horizontal are referred to as deviated. In industry usage, a well's orientation commonly refers both to its inclination from vertical and the azimuthal (compass) direction of a directionally drilled wellbores.

Casing primarily acts as a barrier to lateral movement of fluids, and cement primarily acts as a barrier to unintended vertical movement of fluids. Together, casing and cement are important in preventing fluid movement into drinking water resources, and are the focus of this section. Figure 6-2 illustrates the configurations and types of casing and cement and other features that may occur in oil and gas production wells. The figure depicts an idealized representation of the components of a production well; it is important to note that there is a wide variety in the design of hydraulically fractured oil and gas wells in the United States (U.S. EPA, 2015n), and the descriptions in the figure or in this chapter do not represent every possible well design.

6.2.1.1 Casing

Casing is steel pipe that is placed into the drilled wellbore to maintain the stability of the wellbore, to transport the hydrocarbons from the subsurface to the surface, and to prevent intrusion of other fluids into the well and wellbore (Hyne, 2012; Renpu, 2011). A long continuous section of casing is referred to as a casing string, which is composed of individual lengths of casing (known as casing joints) that are threaded together using casing collars. In different sections of the well, multiple concentric casing strings of different diameters can be used, depending on the construction of the well.

The presence of multiple layers of casing strings can isolate and protect geologic zones containing drinking water. In addition to conductor casing, which prevents the hole from collapsing during drilling, one to three other types of casing may be also present in a well. The types of casing include (from largest to smallest diameter) surface casing, intermediate casing, and production casing (GWPC, 2014; Hyne, 2012; Renpu, 2011). One or more of any of these types of casing (but not necessarily all of them) may be present in a well. Surface casing often extends from the wellhead down to the base (i.e., the bottom or lowest part) of the drinking water resource to be protected. Wells also may be constructed with production liners, which are anchored or suspended from inside the bottom of the previous casing string. Production liners serve the same purpose as production casing but extend only to the end of the previous casing, rather than all the way to the surface. Wells may also have production tubing, which is used to transport the hydrocarbons to the surface. Tie-back liners may be used to extend a production liner to the surface when downhole pressure or corrosive conditions warrant additional protection of the intermediate or production casing.

Among the wells represented by the Well File Review (described in Text Box 6-1), between one and four casing strings were present (the Well File Review did not evaluate conductor casings). A combination of surface and production casings was most often reported, followed by a combination of surface, intermediate, and production strings. All of the production wells used in hydraulic fracturing operations in the Well File Review had surface casing, while approximately 39% of the wells (an estimated 9,100 wells) had intermediate casing, and 94% (an estimated 21,900 wells) had production casing (U.S. EPA, 2015n).[1, 2]

[1] 9,100 wells (95% confidence interval: 2,900 – 15,400 wells).

[2] 21,900 wells (95% confidence interval: 19,200 – 24,600 wells).

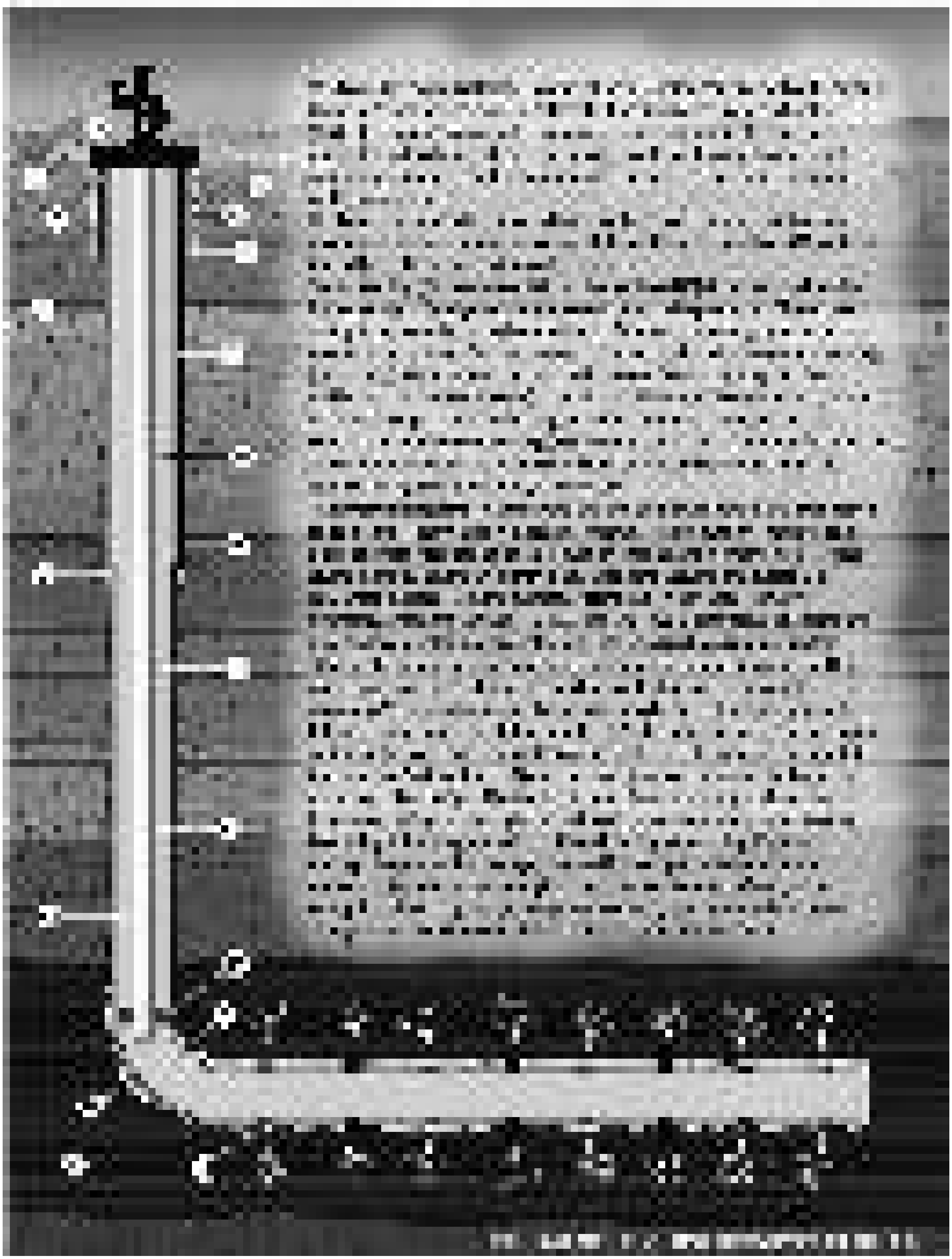


Figure 6-2. Overview of well construction.

Hydraulic fracturing operations impose a variety of stresses on the well components. In order to prevent the formation of pathways to drinking water resources, the casing should be designed with sufficient strength to withstand the stresses it will encounter during the installation, cementing, hydraulic fracturing, production, and post-production phases of the life of the well. These stresses, illustrated in Figure 6-3, include burst pressure (the interior pipe pressure that will cause the casing to burst), collapse pressure (the pressure applied to the outside of the casing that will cause it to collapse), tensile stress (the stress related to stretching exerted by the weight of the casing or tubing being raised or lowered in the hole), compression and bending (the stresses that result from pushing along the axis of the casing or bending the casing), and cyclic stress (the stress caused by frequent or rapid changes in temperature or pressure). While the injection stage represents a relatively brief portion of the life of a hydraulic fracturing well (Section 3.3), injection imposes the highest stresses the well is likely to encounter.

Text Box 6-1. The Well File Review.

The EPA conducted a survey of onshore oil and gas production wells that were hydraulically fractured by nine oil and gas service companies in the continental United States between approximately September 2009 and September 2010. This effort, known as the "Well File Review," produced two reports. The first report, *Review of Well Operator Files for Hydraulically Fractured Oil and Gas Production Wells: Well Design and Construction* (U.S. EPA, 2015n) describes well design and construction characteristics and their relationships to the location of operator-reported drinking water resources and the number and relative location of constructed barriers (i.e., casing and cement) that can block pathways for potential subsurface fluid movement. A second report, *Review of Well Operator Files for Hydraulically Fractured Oil and Gas Production Wells: Hydraulic Fracturing Operations* (U.S. EPA, 2016c) presents information on hydraulic fracturing job characteristics and the reported use of casing pressure tests, annular pressure monitoring, surface treating pressure monitoring, and microseismic monitoring conducted before or during hydraulic fracturing operations; it also explores the roles of well mechanical integrity and induced fracture growth as they relate to the potential for subsurface fluid movement to intersect protected groundwater resources.

The survey was based on a sample of 323 hydraulically fractured oil and gas production wells. Results of the research are presented as rounded estimates of the frequency of occurrence of hydraulically fractured production well design, construction, and operational characteristics with 95% confidence intervals (CIs). The results are statistically representative of an estimated 23,200 onshore oil and gas production wells hydraulically fractured in 2009 and 2010 by nine service companies where an estimated 28,500 hydraulic fracturing jobs were performed.

In addition, the casing must be resistant to corrosion from contact with the formations and any fluids that might be transported through the casing, including hydraulic fracturing fluids, brines, and oil or gas. Casing strength or corrosion resistance can be increased by using fiberglass or high-strength alloys or by increasing the thickness of the casing.

One way to ensure that the strength of the casing is sufficient to withstand the stresses imposed by hydraulic fracturing operations is to pressure test the casing. The casing can be pressurized to the pressure anticipated during hydraulic fracturing operations and shut-in periods; if the well can hold the pressure, it is considered to be leak-free and therefore should be able to withstand the pressures of hydraulic fracturing. However, if the test pressure is less than the hydraulic fracturing pressure, the casing is determined to be leak-free, but its suitability to resist the stresses associated with the planned fracturing operation is less certain.

The Well File Review (U.S. EPA, 2016c) found that pressure tests were performed prior to an estimated 15,600 of 28,500 hydraulic fracturing jobs the EPA studied, including cases where a frac string was pressure tested.[1] In 52% of those pressure tests performed (representing 28% of the hydraulic fracturing jobs studied), the well was tested to a pressure equal to or greater than the maximum pressure that occurred during the hydraulic fracturing job (U.S. EPA, 2016c).[2] Thus, in a significant number of hydraulic fracturing jobs (i.e., 72% of the wells studied), there are no data in the well files to indicate that the casing was tested in a manner that could ensure the adequacy of the casing to withstand the pressures of hydraulic fracturing. While, in some cases, casing may not have been pressure tested because a frac string was to be installed to protect the casing from the increased pressure, only 10% of fracturing jobs were conducted using a frac string.

Figure 6-3. The various stresses to which the casing will be exposed.

In addition to the stresses illustrated, the casing will be subjected to bending and cyclic stresses. Source: U.S. EPA (2012d).

[1] 15,600 jobs (95% confidence interval: 11,800 – 19,300 jobs).

[2] 52% of pressure tests (95% confidence interval: 20 – 82% of tests).

6.2.1.2 Cement

Cement is one of the most important components of a well for providing zonal isolation and reducing impacts on drinking water. Cement in the space between the casing and formation isolates fluid-containing formations from each other, protects the casing from exposure to formation fluids, and provides additional strength to the casing. The strength of the cement and its compatibility with the formation and fluids encountered are important for maintaining mechanical integrity throughout the life of the well.

A variety of methods are available for placing the cement, evaluating the adequacy of the cementing process and the resulting cement job, and repairing any identified deficiencies. Cement is most commonly emplaced by pumping the cement down the inside of the casing to the bottom of the wellbore and then up the space between the outside of the casing and the formation (or the next largest casing string). This method is referred to as the primary cement job and can be performed as a continuous event in a single stage (i.e., "continuous cementing") or in multiple stages (i.e., "staged cementing"). Staged cementing may be used when, for example, the estimated weight and pressure associated with standard cement placement could damage weak zones in the formation (Crook, 2008).

Deficiencies in the cementing process can result from poorly centered casing, poor removal of drilling mud behind the casing, cement shrinkage, premature gelation, excessive fluid loss, improper mixing, or lost cement.[1, 2] Cement deficiencies can be reduced by proper design of the cementing process including use of casing centralizers, proper design of the cement, proper mud removal, and use of cement additives (Kirksey, 2013).[3] If any deficiencies or defects in the primary cement job are identified, remedial cementing may be performed. See Text Box 6-2 for an example of an incident where cementing issues were studied as part of an evaluation of drinking water well impacts.

Text Box 6-2. Dimock, Pennsylvania.

In 2009, shortly after drilling and hydraulic fracturing in the Marcellus Shale commenced in the area, residents near the township of Dimock, Pennsylvania reported that natural gas was appearing or increasing in their water wells (Hammond, 2016; PA DEP, 2009a).

Water wells in the area largely draw from the Catskill Formation and range in depth from less than 50 ft (15 m) to more than 500 ft (150 m) (Molofsky et al., 2013). In this area, the Marcellus Shale is about 7,000 ft (2,000 m) below the surface and its natural gas is extracted through vertical and horizontal wells (Hammond, 2016). Methane exists naturally in the subsurface in this part of Pennsylvania, including in the Catskill Formation and the geologic formations below it (Baldassare et al., 2014; Molofsky et al., 2013; Molofsky et al., 2011).

(Text Box 6-2 is continued on the following page.)

[1] Gelation is the process in the setting of the cement where it begins to solidify and lose its ability to transmit pressure to the formation.

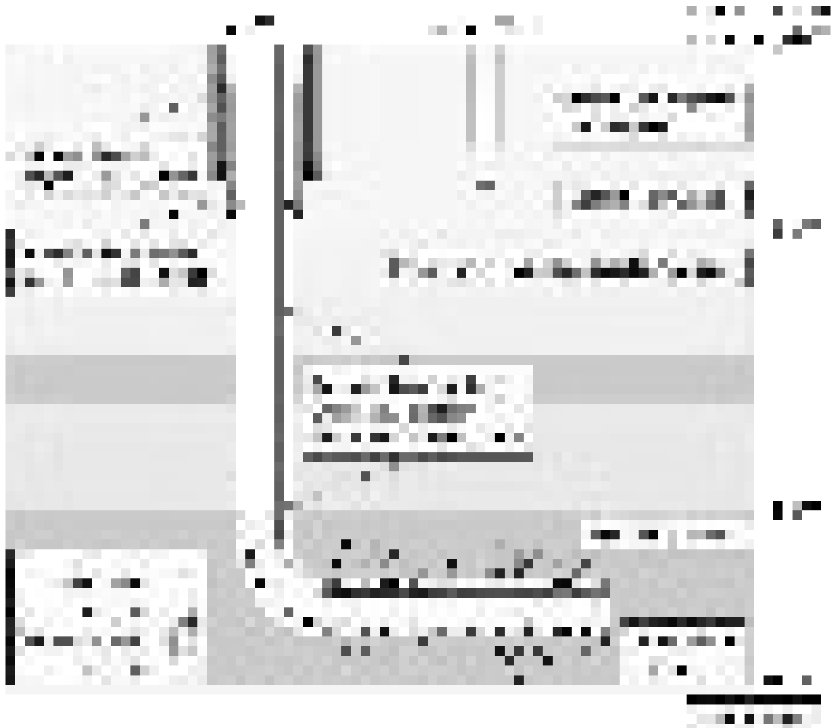
[2] Lost cement refers to a failure of the cement or the spacer fluid used to wash the drilling fluid out of the wellbore to be circulated back to the surface, indicating that the cement has escaped into the formation.

[3] Centralizers are used to keep the casing in the center of the hole and allow an even cement job.

Text Box 6-2 (continued). Dimock, Pennsylvania.

The Pennsylvania Department of Environmental Protection (PA DEP) investigated and made a determination that 18 water wells located within a 9 mi^2 (23 km^2) area had been negatively affected as a result of natural gas extraction activities. For approximately two years, during which there was a partial ban on gas well drilling and hydraulic fracturing in the vicinity, the gas company plugged four gas wells and undertook remedial construction actions at 18 additional gas wells (including remedial cementing at several wells, adding as much as 6,300 ft (1,900 m) of cement behind the production casings) (PA DEP, 2010b, d, 2009a).

The figure below presents a simplified geologic representation of water wells and one type of horizontal gas well completed within the geologic formations in the area. The location of remedial cementing performed in some gas wells is indicated.



Several studies in this and surrounding areas have focused on the geochemistry of the groundwater, in particular on gas composition, and noble and natural gas isotopes in the water. Results are consistent with an accumulation of stray gas originating from greater depth and moving to the Catskill Formation (Jackson et al., 2013c; Molofsky et al., 2013; Molofsky et al., 2011). However, the identity of the geologic formation(s) sourcing the natural gas is not always certain and may be consistent with sourcing from either the Marcellus (as suggested by Jackson et al. (2013c)), or the intervening geologic formations (Molofsky et al., 2013).

The role of hydraulic fracturing in the migration of gas to the Catskill Formation, and the specific pathways by which this migration occurred, is even less certain. Some investigators suspect that the initial gas well construction allowed natural gases from deeper formations to move upward along uncemented wellbores (Hammond, 2016; PA DEP, 2010b, d, 2009a). However, no publicly available information exists to document whether hydraulic fracturing may have aided fluid movement along wellbores to enter drinking water resources from greater depths. Reviews of information, such as hydraulic fracturing job reports showing the intervals hydraulically fractured, injection rates, and pressure monitoring, would support an evaluation of whether hydraulic fracturing might have played a role in the migration of natural gas to drinking water wells in the area.

Among the wells represented in the Well File Review, over 90% of cemented casings were cemented using primary cementing methods. Secondary or remedial cementing was used on an estimated 8% of casings (most often on surface and production casings and less often on intermediate casings).[1] The remedial cementing techniques employed in these wells included cement squeezes, cement baskets, and pumping cement down the annulus (U.S. EPA, 2015n). See Appendix D for more information on remedial cementing techniques.

The cement does not always need to be continuous along the entire length of the well to protect drinking water resources; rather, protection of drinking water resources depends on a good cement seal across the appropriate subsurface zones, including all fresh water- and hydrocarbon-bearing zones. One study of wells in the Gulf of Mexico found that, if at least 50 ft (15 m) of high quality cement was present, pressure differentials as high as 14,000 psi (97 MPa) would not lead to breakdown in isolation between geologic zones (King and King, 2013).

Most wells have cement behind the surface casing, which is a key barrier to contamination of drinking water resources. The surface casings in nearly all of the wells used in hydraulic fracturing operations represented in the Well File Review (93% of the wells, or an estimated 21,500 wells) were fully cemented.[2,3] None of the wells studied in the Well File Review had completely uncemented surface casings.

The length and location of cement behind intermediate and production casings can vary based on the presence and locations of over-pressured formations, formations containing fluids, or geologically weak formations (i.e., those that are prone to structural failure when exposed to changes in subsurface stresses). State regulations and economics also play a role.

In the Well File Review, the intermediate casings of most of the wells studied were fully cemented, although there were relatively wide 95% confidence intervals in the results. Among production casings, about half were partially cemented, about a third were fully cemented, and the remainder were either uncemented or their cementing status was undetermined. Among the approximately 9,100 wells represented in the Well File Review that are estimated to have intermediate casing, the intermediate casing was fully cemented in an estimated approximately 7,300 wells (80%) and partially cemented in an estimated 1,700 wells (19%).[4,5] Production casings were partially cemented in 47% of the wells, or approximately 10,900 wells (U.S. EPA, 2015n).[6]

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[1] 8% of casings (95% confidence interval: 3% – 14% of casings).

[2] The Well File Review defined fully cemented casings as casings that had a continuous cement sheath from the bottom of the casing to at least the next larger and overlying casing (or the ground surface, if surface casing). Partially cemented casings were defined as casings that had some portion of the casing that was cemented from the bottom of the casing to at least the next larger and overlying casing (or ground surface), but were not fully cemented. Casings with no cement anywhere along the casing, from the bottom of the casing to at least the next larger and overlying casing (or ground surface), were defined as uncemented.

[3] 21,500 wells (95% confidence interval: 19,500 – 23,600 wells).

[4] 9,100 wells (95% confidence interval: 2,900 – 15,400 wells).

[5] 7,300 wells (95% confidence interval: 600 – 13,900 wells).

[6] 10,900 wells (95% confidence interval: 6,900 – 14,900 wells).

The Well File Review also estimated the number of wells with a continuous cement sheath along the outside of the well. An estimated 6,800 of the wells represented in the study (29%) had cement from the bottom of the well to the ground surface, and approximately 15,300 wells (66%) had one or more uncemented intervals between the bottom of the well and the surface.[1,2] In the remaining wells, the location of the top of the cement was uncertain, so no determination could be made regarding whether the well had a continuous cement sheath along the outside of the well (U.S. EPA, 2015n).

A variety of logs are available to evaluate the quality of cement behind the well casing. Among wells in the Well File Review, the most common type of cement evaluation log run was a standard acoustic cement bond log (U.S. EPA, 2015n). Standard acoustic cement bond logs are used to evaluate both the extent of the cement placed along the casing and the cement bond between the cement, casing, and wellbore. Cement bond indices calculated from standard acoustic cement bond logs on the wells in the Well File Review showed a median bond index of 0.7 just above the hydraulic fracturing zone; this value decreased to 0.4 over a measured distance of 5,000 ft (2,000 m) above the hydraulic fracturing zone (U.S. EPA, 2015n).[3] While standard acoustic cement bond logs can give an average estimate of bonding, they cannot alone indicate zonal isolation, because they may not be properly run or calibrated (Boyd et al., 2006; Smolen, 2006). One study of 28 wells found that cement bond logs failed to predict communication between formations 11% of the time (Boyd et al., 2006). In addition, they cannot discriminate between full circumferential cement coverage by weaker cement and lack of circumferential coverage by stronger cement (King and King, 2013; Smolen, 2006). A few studies have compared cement bond indices to zonal isolation, with varying results. For example, Brown et al. (1970) showed that among 16 South American wells with varying casing size and cement bond indices, a cemented 5.5 in (14 cm) diameter casing with a bond index of 0.8 along as little as 5 ft (1.5 m) can act as an effective seal. The authors also suggest that an effective seal in wells having calculated bond indices differing from 0.8 are expected to have an inverse relationship between bond index and requisite length of the cemented interval, with longer lengths needed along casing having a lower bond index. Another study recommends that wells undergoing hydraulic fracturing should have a given cement bond over an interval three times the length that would otherwise be considered adequate for zonal isolation (Fitzgerald et al., 1985). Conversely, King and King (2013) concluded field tests from wells studied by Flournoy and Feaster (1963) had effective isolation when the cement bond index ranged from 0.31 to 0.75.

External mechanical integrity tests (MITs), including temperature logs, noise logs, and radioactive tracer logs, are another means to evaluate the zonal isolation performance of well cement. Instead of measuring the apparent quality of the cement, external MITs measure whether there is evidence of fluid movement along the wellbore (and potentially to a drinking water resource). An external MIT conducted before the hydraulic fracturing job can allow detection of channels in the cement that could allow injected fluids to move out of the production zone. An external MIT performed

[1] 6,800 wells (95% confidence interval: 1,600 – 11,900 wells).

[2] 15,300 wells (95% confidence interval: 10,500 – 20,100 wells).

[3] Cement bond logs are used to calculate a bond index, which varies between 0 and 1, with 1 representing the strongest bond and 0 representing the weakest bond.

after hydraulic fracturing operations can detect any fluid movement resulting from cement damage caused by the hydraulic fracturing job. It is important to note that, if a well fails an MIT, this does not mean there is a failure of the well or that drinking water resources are impacted. An MIT failure is a warning that something needs to be addressed, and a loss of mechanical integrity is an event that can result in fluid movement from the well if remediation is not performed. More details on MITs are available in Appendix D.

Monitoring the treatment pressure of the hydraulic fracturing operation can also detect problems occurring during fracturing. Sudden changes in pressure during hydraulic fracturing operations can be indicative of failures in the cement or casing. This type of monitoring is performed in nearly all hydraulic fracturing jobs: the Well File Review (U.S. EPA, 2016c) found that the treatment pressure was monitored in 97% (or 27,700) of all hydraulic fracturing jobs studied.[1]

6.2.1.3 Well Orientation

A well can be drilled and constructed with any of several different orientations: vertical, horizontal, and deviated. The well's orientation can be important, because it affects the difficulty of drilling, constructing, and cementing the well. In particular, as described in Section 6.2.2, constructing and cementing horizontal wells present unique challenges (Sabins, 1990). In a vertical well, the wellbore is vertical throughout its entire length, from the wellhead at the surface to the production zone. Deviated wells are usually drilled vertically in the shallowest part of the well but are then drilled directionally, deviating from the vertical direction at some point such that the bottom of the well is at a significant lateral distance away from the point in the subsurface directly under the wellhead. In a horizontal well, the well is drilled vertically to a point known as the kickoff point, where the well turns toward the horizontal, extending into and parallel with the approximately horizontal targeted producing formation (Figure 6-2).

Among wells evaluated in the Well File Review, about 65% were vertical, 11% were horizontal, and 24% were deviated wells (U.S. EPA, 2015n).[2] This is generally consistent with information available in industry databases—of the approximately 16,000 oil and gas wells used in hydraulic fracturing operations in 2009 (one of the years for which the data for the Well File Review were collected), 39% were vertical, 33% were horizontal, and 28% were either deviated or the orientation was unknown (DrillingInfo, 2014b). See Section 3.3 for additional information on the use of horizontal wells in the United States.

6.2.1.4 Well Completion

Another important aspect of well construction is the way in which the well is completed into the production zone, because the well's completion is part of the system of barriers and must be intact for the well to operate properly. A variety of completion configurations are available. The most common configuration is for casing to extend to the end of the wellbore and be cemented in place (U.S. EPA, 2015n; George et al., 2011; Renpu, 2011). In these cased and cemented completions, the

[1] 27,700 jobs (95% confidence interval: 24,800 – 30,600 jobs).

[2] The Well File Review considered any non-horizontal well in which the well bottom was located more than 500 ft (152 m) laterally from the wellhead as being deviated.

cement provides the primary containment of fluids to the production zone. Before hydraulic fracturing begins, perforations are made through the casing and cement into the production zone. It is through the perforated casing and cement that hydraulic fracturing is conducted. In some cases, a smaller temporary casing, known as a temporary frac string, is inserted inside the production casing to protect the casing from the high pressures imposed during hydraulic fracturing operations.

A different type of a cased completion uses production casing set on formation packers, where the production casing extends through the production zone and the length of the casing extending through the drilled horizontal wellbore is left uncemented, but has a series of formation packers that swell to seal the annulus between the casing and the formation.[1] With these completions, the production zone is fractured in separate stages through ports that open between the formation packers. When formation packers are used, they provide the primary isolation of hydraulic fracturing fluids during hydraulic fracturing.

Another type of completion is an open hole completion. When open hole completions are used, the entire production zone is fractured all at once in a single stage or may be fractured in separate stages using a temporary frac string set on one or more temporary formation packers that are positioned to a different interval for each stage. If a temporary frac string is used in an open hole completion, its packer(s) provide the primary isolation of hydraulic fracturing fluids during hydraulic fracturing and if no temporary frac string is used, then the next higher casing in the well provides the primary isolation of hydraulic fracturing fluids during the treatment.

Among wells represented in the Well File Review, an estimated 6% of wells (1,500 wells) had open hole completions, 6% of wells (1,500 wells) used formation packers, and the rest were cased and cemented (U.S. EPA, 2015n).[2,3]

In some cases, wells may be re-completed after the initial construction, with re-fracturing if production has decreased (Vincent, 2011). Re-completion also may include additional perforations in the well at a different interval to produce from a new formation, lengthening the wellbore, or drilling new laterals from an existing wellbore. In 95% of the re-completions represented in the Well File Review, hydraulic fracturing occurred at shallower depths than the previous job (U.S. EPA, 2016c).[4]

6.2.2 Factors that can Affect Fluid Movement to Drinking Water Resources

The following sections describe the pathways for fluid movement that can develop within the production well and wellbore. We also describe the conditions leading to the development of fluid movement pathways and, where available, evidence that a pathway has allowed fluid movement to

[1] A formation packer is a specialized casing part that has the same inner diameter as the casing but whose outer diameter expands to make contact with the formation and seal the annulus between the uncemented casing and formation, preventing migration of fluids.

[2] 1,500 wells with open hole completions (95% confidence interval: 10 – 4,800 wells).

[3] 1,500 wells using formation packers (95% confidence interval: 1,400 – 1,600 wells).

[4] 95% of jobs (95% confidence interval: 75 – 99% of jobs).

occur within the casing or cement, and—in the case of sustained casing pressure (Section 6.2.2.4)—a combination of factors within the casing and cement. (See Figure 6-4 for an illustration of potential fluid movement pathways related to casing and cement.)

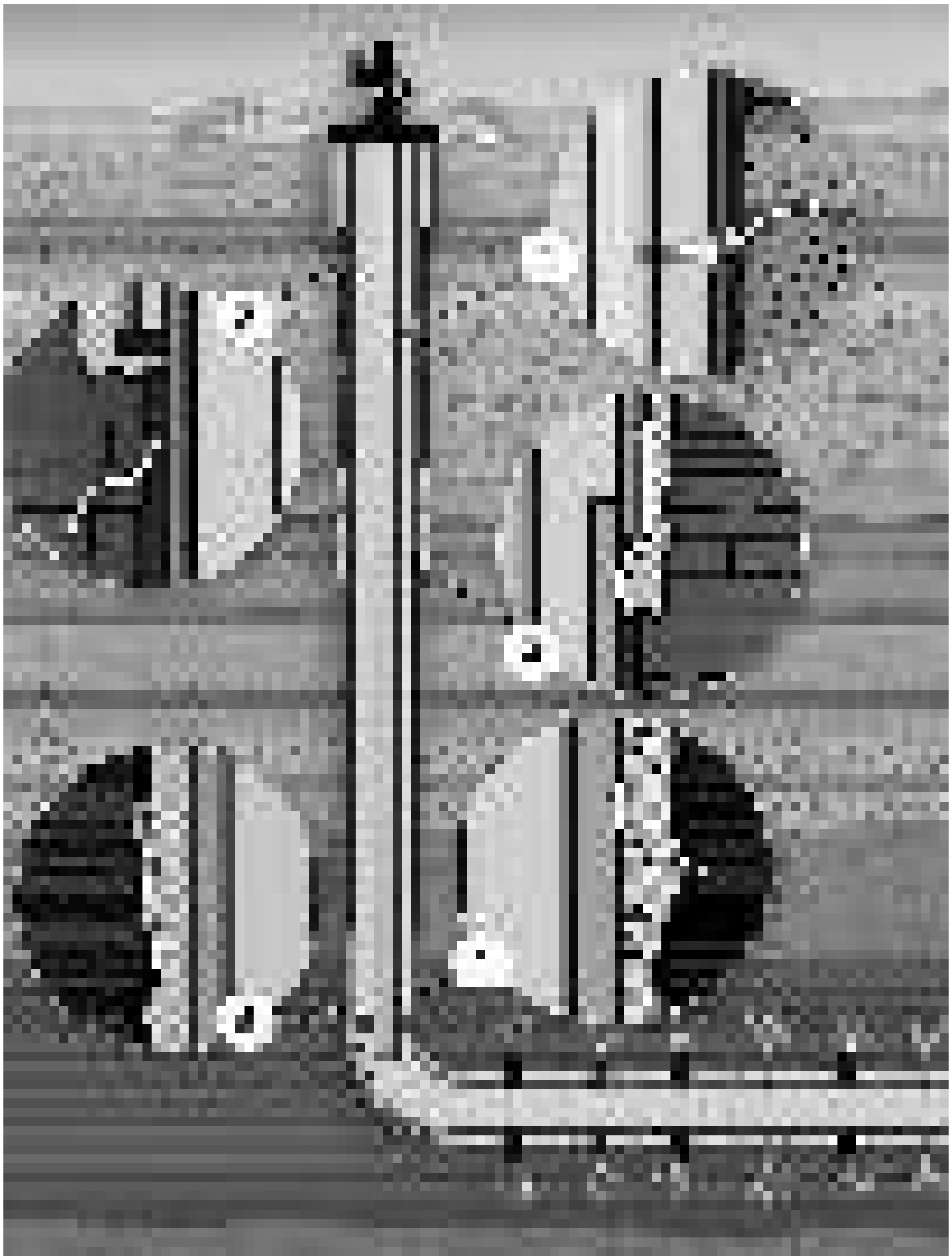

Figure 6-4. Potential pathways for fluid movement in a cemented wellbore.

These pathways (represented by the white arrows) include: (1) casing and tubing leak into a permeable formation, (2) migration along an uncemented annulus, (3) migration along microannuli between the casing and cement, (4) migration through poor cement, and (5) migration along microannuli between the cement and formation. Note: the figure is not to scale and is intended to provide a conceptual illustration of pathways that may develop within the well.

We describe information regarding the rate at which these pathways have been identified in hydraulic fracturing wells when it is available. Where such information does not exist, we present the results of research on oil and gas production wells in general or on injection wells, including those used for the geologic sequestration of carbon dioxide.[1] Publicly accessible information is insufficient to determine whether wells intended for hydraulic fracturing are constructed differently from production wells where no fracturing is conducted. See Chapter 10 for additional discussion of data gaps. It is not generally possible, based on the literature reviewed for this assessment, to determine the precise degree to which hydraulic fracturing created, or moved fluids along, the pathways described or whether all of the wells studied were hydraulically fractured. Nor is it generally possible to estimate the degree to which wells that were hydraulically fractured have a significantly different number of redundant barriers to protect drinking water resources than other production wells. However, given the applicability of well construction technology to address the subsurface conditions encountered in hydraulic fracturing operations and production or injection operations in general, the information presented here is considered relevant to the assessment.

6.2.2.1 Pathways Related to Well Casing

High pressures associated with hydraulic fracturing operations can damage casing and lead to fluid movement that can impact drinking water quality. As noted above, the casing string through which hydraulic fracturing fluids are injected is subject to higher internal pressures during hydraulic fracturing operations than during other phases in the life of a production well. To withstand the stresses created by the high pressure of hydraulic fracturing, the well and its components must have adequate strength and elasticity. If the casing is compromised or is otherwise not strong enough to withstand these stresses (Figure 6-3), a casing failure can result. If undetected or not repaired, casing failures can serve as pathways for hydraulic fracturing fluids to leak out of the casing. Below we present data or information suggesting that pathways along the casing are present or allowing fluid movement. See Chapter 10 for more information on factors that can increase or decrease the frequency or severity of impacts to drinking water quality associated with well construction.

Hydraulic fracturing fluids or fluids present within the well casing could flow into other zones in the subsurface if there is a leak in the casing, and cement is inadequate or not present. As described below, pathways for fluid movement associated with well casing can be related to the original design or construction of the well, degradation of the casing over time, or problems that can arise through extended use as the casing succumbs to stresses.

Casing failure can also occur if the wellbore passes through a structurally weak geologic zone that shears and deforms the well casing. Such shearing is common when drilling through zones containing salt (Renpu, 2011). The changes in the pressure field in the portions of the formation near the wellbore during hydraulic fracturing can also cause mechanically weak formations to shear, potentially damaging the well's casing or cement. Palmer et al. (2005) demonstrated through modeling that hydraulic fracturing within coal that had a low unconfined compressive strength

[1] An injection well is a well into which fluids are being injected (40 Code of Federal Regulations (CFR) 144.3).

could cause shear failure of the coalbeds surrounding the wellbore. Shearing of the coalbed layers can cause the casing to deform and potentially fail.

Corrosion in uncemented zones is the most common cause of casing failure. This can occur if uncemented sections of the casing are exposed to corrosive substances such as brine or hydrogen sulfide (Renpu, 2011). Corrosion commonly occurs at the collars that connect sections of casing or where equipment is attached to the casing. Corrosion at collars can exacerbate problems with loose or poorly designed connections, which are another common cause of casing leaks (Agbalagba et al., 2013; Brufatto et al., 2003). Watson and Bachu (2009) found that 66% of all casing corrosion occurred in uncemented well sections, as shown in Pathway 1 of Figure 6-4.

As noted above, the casing and cement work together to strengthen the well and provide zonal isolation. Uncemented casing does not necessarily lead to fluid migration. However, migration can occur if the casing in an uncemented zone fails during hydraulic fracturing operations.

Other mechanical integrity problems have been found to vary with the well environment, particularly environments with high pressures and temperatures. Wells in high pressure/high temperature environments, wells with thermal cycling, and wells in corrosive environments can have life expectancies of less than 10 years (Agbalagba et al., 2013).

The depth of the surface casing relative to the base of the drinking water resource to be protected is an important factor in protecting the drinking water resource. In a limited risk modeling study of selected injection wells in the Williston Basin, Michie and Koch (1991) found the risk of aquifer contamination from leaks from the inside of the well to the drinking water resource was seven in 1,000,000 injection wells if the surface casing was set deep enough to cover the drinking water resource, and that the risk increased to six in 1,000 wells if the surface casing was not set deeper than the bottom of the drinking water resource. An example where surface casing did not extend below drinking water resources comes from an investigation of 14 selected drinking water wells with alleged water quality problems in the Wind River and Fort Union formations near Pavillion, Wyoming (WYOGCC, 2014b). The state found that the surface casing of oil and gas wells was shallower than the depth of three of the 14 drinking water wells. Some of the oil and gas wells with shallow surface casing had elevated gas pressures in their annuli (WYOGCC, 2014b). The presence of gas in the annuli, combined with surface casing that is set above the lowest drinking water resource, could allow migration of gas into drinking water resources.

Fleckenstein et al. (2015) found that the depth of surface casing and the presence of uncemented gas zones are major factors in determining the likelihood of well failures and contamination. Their study in the Wattenberg field in Colorado classified the wells in the field into seven categories based on the depth of surface casing, the presence of cement, and the presence of intermediate gas zones above the production zone (Table 6-1). The categories were arranged in order of risk, with category 1 wells being at the highest risk of allowing fluid migration and category 7 wells being the least likely to allow migration. The overall barrier failure rate was 2.4% of all wells, and the overall catastrophic failure rate was 0.06% of all wells. A remediation effort was made in order to decrease the likelihood of fluid migration, which included the plugging of 1,103 of the 17,948 wells studied. All the wells shown in the table are vertical wells that were drilled between 1970 and 2013. Similar categories were created for the 973 horizontal wells in the field. No failures were recorded for any of the horizontal wells.

Table 6-1. Failure rates of vertical wells in the Wattenberg field, Colorado.
From Fleckenstein et al. (2015).

Category and description[a, b]	Total wells	Wells with barrier failures[c]	Wells with catastrophic failures[d]
1--Shallow surface casing and exposed (uncemented) over-pressured intermediate gas zones	399	92 (23.06%)	3 (0.75%)
2 – Shallow surface casing and exposed under-pressured intermediate gas zones	7,811	276 (3.53%)	6 (0.08%)
3 – Shallow surface casing but no exposed gas zones	3,407	20 (0.59%)	1 (0.03%)
4 – Shallow surface casing with production casing cemented to bottom of surface casing	1,063	0 (0%)	0 (0%)
5 – Deep surface casing with production casing cement below top of gas	1,374	13 (0.95%)	0 (0%)
6 – Deep surface casing with production casing cement above top of gas	2,069	0 (0%)	0 (0%)
7 – Deep surface casing with production casing cement to bottom of surface casing	705	0 (0%)	0 (0%)
Total	16,828	401 (2.4%)	10 (0.06%)

[a] The study defined shallow surface casing as casing that did not extend below the Fox Hills Aquifer, a deep aquifer that had not been identified and protected by the state prior to 1994.

[b] Uncemented zones could be located along the intermediate or production casings.

[c] Barrier failures were considered to have occurred when there were signs of a failure, but no contamination.

[d] A catastrophic failure was considered to have occurred where there was contamination of drinking water aquifers (i.e., the presence of thermogenic gas in a drinking water well) and evidence of a well defect such as exposed intermediate gas zone or casing leaks.

Sherwood et al. (2016) examined complaint records in the same field. They reviewed 29 Colorado Oil and Gas Commission complaint records associated with 32 incidents at 42 drinking water wells in which thermogenic methane was detected. (See Text Box 6-3 for more information on thermogenic and biogenic methane.) Of the 29 complaints, 10 were determined to be caused by oil and gas wellbore failures, one was suspected of being a wellbore failure but not confirmed, three were settled in court with documents being sealed, and the remaining 15 were unresolved.[1] If all 32 cases are assumed to be associated with an individual oil and gas well, that would result in a failure rate of 0.06% of all oil and gas wells in the basin, the same failure rate as found in the Fleckenstein et al. (2015) study. As in the Fleckenstein study, surface casing that was set too shallow and

[1] This paper defined a wellbore failure as the failure of one or more barriers to fluid movement in the wellbore (e.g., cement, casing, etc.).

uncemented intermediate zones were the main contributing factors to wellbore failure. All 11 of the confirmed or suspected wellbore failures involved vertical wells that were drilled before 1933 and had surface casing shallower than nearby aquifers. Of these wells, seven had been hydraulically fractured. The study noted that the failure rate was fairly constant over time with about two new cases per year since 2000 and that the rate had not changed since high rates of hydraulic fracturing of horizontal wells became prevalent around 2010. This is consistent with the study's finding of no failures in horizontal wells.

During hydraulic fracturing operations in September of 2010 near Killdeer, in Dunn County, North Dakota, the production, surface, and conductor casing of the Franchuk 44-20 SWH well ruptured, causing fluids to spill to the surface (Jacob, 2011). The rupture occurred during the 5th of 23 planned stages of hydraulic fracturing when the pressure spiked to over 8,390 psi (58 MPa). Ruptures were found in two locations along the production casing—one just below the surface and one at about 55 ft (17 m) below ground surface. The surface casing ruptured in three places down to a depth of 188 ft (57 m), and the conductor casing ruptured in one place. Despite a shutdown of the pumps, the pressure was still sufficient to cause fluid to travel through the ruptured casings and to flow to the surface. Ultimately, over 166,000 gal (628,000 L) of fluids and approximately 2,860 tons (2,595 metric tons) of contaminated soil were removed from the site (Jacob, 2011).

The EPA investigated the Killdeer site as part of its Retrospective Case Study in Killdeer, North Dakota: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources (U.S. EPA, 2015i). As part of the study, water quality samples were collected from three domestic wells, nine monitoring wells, two supply wells, one municipal well, and one state well in July 2011, October 2011, and October 2012. Two study wells installed less than 60 ft (20 m) from the production well (NDGW08 and NDGW07) had significant differences in water quality compared to the remaining study wells.[1] These two wells showed differences in ion concentrations (e.g., chloride, calcium, magnesium, sodium, strontium) and tert-butyl alcohol (TBA). The sampling identified brine contamination that was consistent with mixing of local groundwater with brine from Madison Group formations, which the well had penetrated. The TBA was consistent with degradation of tert-butyl hydroperoxide, a component of the hydraulic fracturing fluid used in the Franchuk well. Based on the analysis of potential sources of contamination, the EPA determined that the only potential sources of TBA were gasoline spills, leaky underground storage tanks, and hydraulic fracturing fluids. However, the lack of MTBE and other signature compounds associated with gasoline or fuels strongly suggests that the rupture (blowout) was the only source consistent with findings of high brine and TBA concentrations in the two wells.[2] For additional information about impacts at the Killdeer site, see Section 6.3.2.2.

[1] Based on comparison with historical Killdeer aquifer water quality data, the remaining study wells were in general consistent with historical background data; these wells were then used for the data analysis as background wells. Comparisons of TBA between the study data and historical data could not be made since no historical data for TBA were found for the Killdeer aquifer.

[2] A well blowout is the uncontrolled flow of fluids out of a well.

Inadequate casing or cement can respond poorly when blowout preventers activate.[1] When blowout preventers are activated, they immediately stop the flow in the well, which can create a sudden pressure increase in the well. If the casing or cement are not strong enough to withstand the increased pressure when this occurs, well components can be damaged (The Royal Society and the Royal Academy of Engineering, 2012) and the potential for fluid release and migration in the subsurface increases. Blowouts can also occur during the production phase, and cause spills on the surface that can affect drinking water resources; see Section 7.4.2.2.

While well construction and hydraulic fracturing techniques continue to change, the pressure- and temperature-related stresses associated with hydraulic fracturing remain as factors that can affect the integrity of the well casing. Tian et al. (2015) investigated one such case where temperature effects led to casing damage in China. In the Changning-Weiyuan basin in China, 13 of 33 wells (39.4%) suffered casing damage, with most of the wells experiencing the damage after fracturing. The authors found that injection of the cooler hydraulic fracturing fluid led the casing temperature to drop from the formation temperature of 212°F to 64°F (100°C to 18°C) in some cases. This drop in temperature, in turn, caused pockets of high pressure fluid outside the casing to contract. If the temperature dropped below 136°F (58°C), the effect was sufficient to form a vacuum outside the casing, potentially leading to casing deformation. Areas of the casing with severe doglegs (i.e., bends in the well) and where there was a smaller space between the casing and formation were more prone to this type of damage. While the conditions in this Chinese basin may or may not represent conditions in U.S. basins, they do demonstrate that temperature changes during hydraulic fracturing can place additional stress on the well and highlight their importance as a consideration in casing design. In the case mentioned, increasing the space around the casing, decreasing dogleg angles, properly removing drilling mud, and using high strength, low elasticity cement were found to improve performance.

Sugden et al. (2013) used numerical simulation to examine a similar problem using parameters chosen to represent the Haynesville Shale. They found that injecting a fluid at 70°F (21°C) could cool the wellbore temperature from 320°F to 96°F (160°C to 36°C). The temperature change was 90% complete within the first half hour of hydraulic fracturing operations. They also found that a well with a 20 degree per 100 ft (31 m) dogleg decreased the pressure required to damage the well casing by 850 psi (5.9 MPa). The study also reported that cooling of fluids in voids in the cement can lead to contraction of the fluids. In low permeability shales, fluid cannot flow in fast enough to compensate, and the pressure in the void can drop significantly. Sugden et al. (2013) report that such cement voids can reduce the pressure needed to rupture the casing by 40%.

Emerging isotopic techniques can be used to identify the extent to which stray gas occurring in drinking water resources is linked to casing failure. (See Text Box 6-3 for more information on stray gas.) Darrah et al. (2014) used hydrocarbon and noble gas isotope data to investigate the source of gas in eight identified "contamination clusters" that occurred in the Marcellus and Barnett shales. Seven of these clusters were stripped of atmospheric gases (Argon-36 and Neon-20) and were

[1] A blowout preventer (BOP) is casinghead equipment that prevents the uncontrolled flow of oil, gas, and mud from the well by closing around the drill pipe or sealing the hole (Oil and Gas Mineral Services, 2010). BOPs are typically a temporary component of the well, in place only during drilling and perhaps through hydraulic fracturing operations.

enriched in crustal gases, indicating the gas migrated quickly from depth without equilibrating with intervening formations. The rapid transport was interpreted to mean that the migration did not occur along natural fractures or pathways, which would have allowed equilibration to take place. Based on the isotopic results, the authors also ruled out the possibility that the gas was carried upward (relative to the surface) as the geologic formation in which it formed was uplifted over geologic time. Possible explanations for the rapid migration include transport up the well and through a leaky casing (Pathway 1 in Figure 6-4) or along uncemented or poorly cemented intervals from shallower depths (Pathways 2 through 5 in Figure 6-4). In four Marcellus Shale clusters, gas found in drinking water wells had isotopic signatures and ratios of ethane to methane that were consistent with those in the producing formation. The authors conclude that this suggests that gas migrated along poorly constructed wells from the producing formation, likely with improper, faulty, or failing production casings. In three clusters, the isotopic signatures and ethane to methane ratios were consistent with formations overlying the Marcellus. The authors suggest that this migration occurred from the shallower gas formations along uncemented or improperly cemented wellbores. In another Marcellus cluster in the study, deep gas migration was linked to a subsurface well, likely from a failed well packer.

Text Box 6-3. Stray Gas Migration.

Stray gas refers to the phenomenon of natural gas (primarily methane) migrating into shallow drinking water resources, into water wells or other types of wells, to the surface, or to near-surface features (e.g., basements, streams, or springs). The source of the migrating gas can be natural gas reservoirs (either conventional or unconventional), or from coal mines, landfills, leaking gas wells, leaking gas pipelines, buried organic matter, or natural microbial processes (Li and Carlson, 2014; Baldassare, 2011). Although methane is not a regulated drinking water contaminant, its presence in drinking water resources can initiate chemical and biological reactions that release or mobilize other contaminants. Over time, it can promote more reducing conditions in groundwater, potentially leading to reductive dissolution of iron and manganese and the possible liberation of naturally occurring contaminants, such as arsenic, that are potentially associated with iron and manganese (U.S. EPA, 2014f). In addition, methane can accumulate to explosive levels in confined spaces (like basements or cellars) if it exsolves (degases) from groundwater into those spaces. (See Section 9.5.5 for information about the hazards associated with methane exposure.)

Detectable levels of dissolved natural gas exist in some aquifers, even in the absence of human activity (Gorody, 2012). In northern Pennsylvania and New York, low levels of methane are frequently found in water wells in baseline studies, prior to commercial oil or gas development (Christian et al., 2016; Kappel, 2013; Kappel and Nystrom, 2012); for example, one USGS study detected methane in 80% of sampled wells in Pike County, Pennsylvania (Senior, 2014). The origin of methane in groundwater can be either thermogenic (produced by high temperatures and pressures in deeper formations, such as the gas found in the Marcellus Shale) or biogenic (produced in shallower formations by bacterial activity in anaerobic conditions).

Gas occurrence is linked to local and regional geologic characteristics. In some cases, thermogenic methane occurs naturally in shallow formations because the formation itself was uplifted (relative to the surface) over geologic time. In other cases, it has migrated there via one or more pathways. For example, Brantley et al. (2014) suggest that northern Pennsylvania's glacial history can help explain why stray gas is more common

(Text Box 6-3 is continued on the following page.)

Text Box 6-3 (continued). Stray Gas Migration.

there than in the southern part of the state. Christian et al. (2016), Mcphillips et al. (2014), Molofsky et al. (In Press), and Wilson (2014) all identified correlations between the presence of methane in water wells and certain geologic, hydrographic, and geochemical parameters, such as valley locations and the presence of coal beds.

Stray gas migration can be a technically complex phenomenon to study, in part because there are many potential sources and routes for migration. When a particular site lacks detailed monitoring data, especially baseline measurements, determination of sources and migration routes is complicated and challenging. Examining the concentrations and isotopic compositions of methane and higher molecular weight hydrocarbons such as ethane and propane can aid in determining the source of stray gas (Tilley and Muehlenbachs, 2012; Baldassare, 2011; Rowe and Muehlenbachs, 1999). Isotopic composition and methane-to-ethane ratios can help determine whether the gas is thermogenic or biogenic in origin and whether it is derived from shale or other formations (Gorody, 2012; Muehlenbachs et al., 2012; Barker and Fritz, 1981). Isotopic analysis can also be used to identify the strata where the gas originated and provide evidence for migration mechanisms (Darrah et al., 2014). For example, isotope-based techniques have been used to investigate the potential sources of methane in drinking water wells in Dimock, Pennsylvania (Hammond, 2016), and Jackson et al. (2013c) found evidence of potential Marcellus gas contamination in some Pennsylvania drinking water wells using stable-isotopic ratios, while other wells in the area appeared to be contaminated by shallower sources (not associated with gas production).

However, determining the source of methane does not necessarily establish the migration pathway. Multiple researchers (e.g., Siegel et al., 2015; Jackson et al., 2013c; Molofsky et al., 2013; Révész et al., 2012; Osborn et al., 2011) have described biogenic and/or thermogenic methane in groundwater supplies in Marcellus gas production areas, although the sources and pathways of migration are generally unknown. Well casing and cementing issues may be an important source of stray gas problems (Jackson et al., 2013c); however, other potential subsurface pathways are also discussed in the literature. Zhang and Soeder (2016) suggested that air-drilling practices used to construct the vertical component of gas wells can affect methane migration by creating groundwater surges in the shallow subsurface. The type of well may also play a role; in one study, deviated gas wells in Canada were three to four times more likely than vertical wells to have evidence of gas migration to the surface (Jackson et al., 2013b).

In the absence of data on specific pathways, some researchers have investigated geographic correlations. Jackson et al. (2013c) and Osborn et al. (2011) found that thermogenic methane concentrations in well water increased with proximity to Marcellus Shale production sites. In contrast, Molofsky et al. (2013) found the presence of gas to be more closely correlated with topography and elevation, and (Siegel et al., 2015)found no correlation between methane in groundwater and proximity to production wells. Kresse et al. (2012) investigated methane concentration and isotopic geochemistry in shallow groundwater in the Fayetteville Shale area, and found no evidence that the water had been influenced by shale gas activities. Similarly, Li and Carlson (2014), while not ruling out potential leakage pathways from deeper reservoirs, found no systematic correlation between increasing well drilling density in the Wattenberg Field in Colorado and near-surface stray gas concentrations.

EPA conducted retrospective case studies to investigate stray gas in northeastern Pennsylvania and the Raton Basin of Colorado. As described in the northeastern Pennsylvania case study report, *Retrospective Case Study in Northeastern Pennsylvania: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2014f), 27 of 36 drinking water wells within the study area (75%) contained elevated methane concentrations. For some of the wells, the EPA concluded that the methane (of both thermogenic and biogenic origin) was naturally occurring gas, not attributable to gas exploration activities. In others, it

(Text Box 6-3 is continued on the following page.)

Text Box 6-3 (continued). Stray Gas Migration.

appeared that methane had entered the water wells following well drilling and hydraulic fracturing. In most cases, the methane in the wells likely originated from intermediate formations between the production zone and the surface; however, in some cases, the methane appears to have originated from deeper layers such as those where the Marcellus Shale is found (U.S. EPA, 2014f). The Raton Basin case study examined the Little Creek Field, where potentially explosive quantities of methane entered drinking water wells in 2007. As described in the EPA's *Retrospective Case Study in the Raton Basin, Colorado: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2015k), the methane was found to be primarily thermogenic in origin, modified by biologic oxidation (U.S. EPA, 2015k). Secondary biogeochemical changes related to the migration and reaction of methane within the shallow drinking water aquifer were reflected in the characteristics of the Little Creek Field groundwater (U.S. EPA, 2015k).

The sources of methane in the two studies could be determined with varying degrees of certainty. Narrowly identifying the most likely pathway(s) of migration has been more difficult. In northeastern Pennsylvania, while the sources could not be definitively determined, the Marcellus Shale could not be excluded as a potential source in some wells based on isotopic signatures, methane-to-ethane ratios, and isotope reversal properties (U.S. EPA, 2014f). The Pennsylvania Department of Environmental Protection (PA DEP) cited at least two operators for failure to prevent gas migration at wells within the study area. Evidence cited by the state included isotopic comparison of gas samples from drinking water wells, water bodies, and gas wells; inadequate cement jobs; and sustained casing pressure (although, under Pennsylvania law, oil or gas operators can be cited if they cannot disprove the contamination was caused by their well using pre-drilling samples) (Llewellyn et al., 2015). A separate study (Ingraffea et al., 2014) showed that wells in this area had higher incidences of mechanical integrity problems relative to wells in other parts of Pennsylvania. While the study did not definitively show that stray gas was linked to construction problems, it does imply that there may be more difficulties in constructing wells in this area. In the Little Creek Field in the Raton Basin, the source of methane was identified as the Vermejo coalbeds. While the nature of the migration pathway is unknown, modeling suggests that it could have occurred along natural rock features in the area and/or along a gas production well (U.S. EPA, 2015k). Because the production wells were shut in shortly after the incident began, the wells could not be inspected to determine whether a mechanical integrity failure in the wellbore was a likely cause of the migration.[1]

These two case studies illustrate the considerations involved with understanding stray gas migration and the difficulty in determining sources and migration pathways. To more conclusively determine sources and migration pathways, studies in which data are collected on mechanical integrity and hydrocarbon gas (e.g., methane, ethane) concentrations both before and after hydraulic fracturing operations, in addition to the types of data summarized above, would be needed.

In the Wattenberg Field in Colorado, Li et al. (2016a) investigated the concentration of various ions in water from an uncontaminated aquifer, an aquifer containing thermogenic methane, and produced water from oil and gas wells to understand the transport of aqueous- and gas-phase fluids at the site. The results indicated that the methane that was contaminating water wells was not transported with aqueous phase fluids; the authors suggested that this can provide evidence for migration mechanisms, because certain pathways (e.g., migration from improperly sealed well

[1] Shutting in a well refers to sealing off a well by either closing the valves at the wellhead, a downhole safety valve, or a blowout preventer.

casings) could potentially result in gas-phase but not aqueous-phase migration. See Text Box 6-4 for another example of an investigation into the occurrence of stray gas in drinking water wells.

Text Box 6-4. Parker County, Texas.

Peer-reviewed studies have been conducted within the Barnett Shale area, which includes Parker County, Texas. These include sampling studies of private water well composition, noble gas content, and isotopic signatures of natural gases, as well as analysis of existing water sample data. Disagreement exists about the origin of the increased natural gas in private well water.

One suggested possibility is that production casing annuli could serve as a migration pathway for natural gas from formations located between the Barnett and the Trinity to reach overlying intervals (including the Trinity aquifer) (Darrah et al., 2014). However, using measurements of hydrocarbon and noble gas isotopes, Wen et al. (2016) suggests the source of methane in the Trinity aquifer water wells is directly from the underlying Strawn Formation and not from pathways associated with the gas production wells although the timing of methane entry into the Strawn is not known.

6.2.2.2 Pathways Related to Cement

Fluid movement can result from inadequate well design or construction (e.g., cement loss or other problems that arise in cementing of wells) or degradation of the cement over time (e.g., corrosion or the formation of microannuli), which may, if undetected and not repaired, cause the cement to succumb to the stresses exerted during hydraulic fracturing.[1] The well cement must be able to withstand the subsurface conditions and the stresses encountered during hydraulic fracturing operations. This section presents data and information that can help indicate that pathways within the cement are present or allowing fluid movement.

Uncemented zones can allow fluids or brines to move into drinking water resources. If a fluid-containing zone is left uncemented, the open annulus between the formation and casing can act as a pathway for migration of that fluid. Fluids can enter the wellbore along any uncemented section of the wellbore if a sufficient pressure gradient is present. Once the fluids have entered the wellbore, they can travel up along the entire uncemented length of the wellbore as shown in Pathway 2 of Figure 6-4.

As mentioned in Section 6.2.2.1, Fleckenstein et al. (2015) found uncemented gas zones to be a significant factor in barrier failures in wells in the Wattenberg basin in Colorado. A report on the Pavillion field by AME (2016) identified a similar set of risk factors for fluid migration including: uncemented production casing, shallow surface casing, and the presence of both an intermediate pressurized gas zone and a permeable groundwater zone encountered in the same production wellbore.

Because of their low density and buoyancy, gaseous fluids such as methane will migrate up the wellbore if an uncemented wellbore is exposed to a gas-containing formation. Gas may then be able

[1] Microannuli are very small openings that form between the cement and its surroundings and that may serve as pathways for fluid migration to drinking water resources.

to enter other formations (including drinking water resources) if the wellbore is uncemented and the pressure in the annulus is sufficient to force fluid into the surrounding formation (Watson and Bachu, 2009; Harrison, 1985). The rate at which the gas can move will depend on the difference in pressure between the annulus and the formation (Wojtanowicz, 2008). See Chapter 10 for a discussion of practices, such as well testing, that can decrease the frequency of such gas migration that could impact drinking water quality.

In several cases, poor or failed cement has been linked to stray gas migration (Text Box 6-3). A Canadian study found that uncemented portions of casing were the most significant contributors to gas migration (Watson and Bachu, 2009). The same study also found that 57% of all casing leaks occurred in uncemented segments. In the study by Darrah et al. (2014) (Section 6.2.2.1), using isotopic data, four clusters of gas contamination were linked to poor cementing. In three clusters in the Marcellus and one in the Barnett, gas found in drinking water wells had isotopic signatures consistent with intermediate formations overlying the producing zone. This suggests that gas migrated from the intermediate units along the well annulus, along uncemented portions of the wellbore, or through channels or microannuli.

Cementing of the surface casing is the primary aspect of well construction intended to protect drinking water resources. Most states require the surface casing to be set and cemented from the level of the lowermost drinking water resource to the surface (GWPC, 2014). Most wells—including those used in hydraulic fracturing operations—have such cementing in place. Among the wells represented in the Well File Review, surface casing was found to be fully cemented in 93% of wells. Of these, an estimated 55% of wells (12,600 wells) were cemented to below the operator-reported protected groundwater resource; in an additional 28% of wells (6,400 wells), the operator-reported protected groundwater resources were fully covered by the next cemented casing string.[1,2,3] A portion of the annular space between the casing and the operator-reported protected groundwater resources was uncemented in at least 3% of wells (600 wells) (U.S. EPA, 2015n).[4]

Improper placement of cement can lead to defects in external mechanical integrity. For example, an improper cement job can be the result of loss of cement during placement into a formation with

[1] In the Well File Review, protected groundwater resources were as reported by well operators. For most wells represented in the Well File Review, protected groundwater resources were identified based on state or federal authorization documents. Other data sources used by well operators included aquifer maps, data from offset production wells, open hole log interpretations by operators, operator experience, online databases, and references to a general requirement by the oil and gas agency.

[2] The research that the EPA reviewed used various terms to describe subsurface water resources that are used/potentially used for drinking water. Where another term is relevant to describing the author's research, we use that term; for the purpose of this assessment, all of these terms are considered to fall within the assessment's definition of "drinking water resources." See Chapter 2 for additional information on the definition of a drinking water resource.

[3] 6,400 wells (95% confidence interval: 500 – 12,300 wells).

[4] 600 wells (95% confidence interval: 10 – 1,800 wells). The well files representing an estimated 8% of wells in the Well File Review did not have sufficient data to determine whether the operator-reported protected groundwater resource was uncemented or cemented. In these cases, there was ambiguity either in the depth of the base or the top of the operator-reported protected groundwater resource. An additional 6% of wells represented had surface casing set below the reported protected groundwater resource depth, but because the protected groundwater depth was based on a nearby water well depth, the true base of the protected groundwater resource may be deeper, leaving uncertainty as to whether the surface casing in these wells is set deeper than the base of the protected groundwater resource.

high porosity or fractures, causing a lack of adequate cement across a water- or brine-bearing zone. Additionally, failure to use cement that is compatible with the anticipated subsurface conditions, failure to remove drilling fluids from the wellbore, and improper centralization of the casing in the wellbore can all lead to the formation of channels (i.e., small connected voids) in the cement during the cementing process (McDaniel et al., 2014; Sabins, 1990). If the channels are small and isolated, they may not lead to fluid migration. However, if they are long and connected, extending across multiple formations, or connecting to other existing channels or fractures, they can present a pathway for fluid migration. Figure 6-4 shows a variety of pathways for fluid migration that are possible from failed cement jobs.

One example of how hydraulic fracturing of a well with insufficient and improperly placed cement led to contamination occurred in Bainbridge Township, Ohio. This incident was well studied by the Ohio Department of Natural Resources (ODNR, 2008) and by an expert panel (Bair et al., 2010). The level of detail available for this case is not typically found in studies of such events but was collected because of the severity of the impacts and the resulting legal action. The English #1 well was drilled to a depth of 3,900 ft (1,200 m) below ground surface (bgs) in October 2007 with the producing formation located between 3,600 and 3,900 ft (1,100 and 1,200 m) bgs. Overlying the producing formation were several uneconomic formations containing over-pressured gas (i.e., gas at pressures higher than the hydrostatic pressure exerted by the fluids within the well).[1] The original cement design required the cement to be placed 700 – 800 ft (210 – 240 m) above the producing formation to seal off these areas. During cementing, however, both the spacer fluid and cement were lost in the subsurface, and the cement did not reach the intended height.[2] Despite the lack of sufficient cement, the operator proceeded with hydraulic fracturing.

During the hydraulic fracturing operation in November 2007, about 840 gal (3,200 L) of fluid flowed up the annulus and out of the well. When the fluid began flowing out of the annulus, the operator immediately ceased operations and shut in the well; this caused the pressure in the wellbore to increase. About a month later, there was an explosion in a nearby house where methane had entered from an abandoned and unplugged drinking water well connected to the cellar (Bair et al., 2010). In addition to the explosion, the over-pressured gas entering the aquifer resulted in the contamination of 26 private drinking water wells with methane. The wells, some of which had histories of elevated methane prior to the incident, were taken off-line. By 2010, all of the well owners had been connected to a public water supply (Tomastik and Bair, 2010).

Contamination at the Bainbridge Township site was the result of inadequate cement. The ODNR determined that failure to cement the over-pressured gas formations, proceeding with the hydraulic fracturing operation without adequate cement, and the extended period during which the well was shut in all contributed to the contamination of the aquifer with stray gas (ODNR, 2008). Cement logs found the cement top was at 3,640 ft (1,110 m) bgs, leaving the uneconomic gas-producing formations and a portion of the production zone uncemented. The surface casing was 253 ft (77 m) deep and cemented to the surface. Hydraulic fracturing fluids flowing out of the

[1] Hydrostatic pressure is the pressure exerted by a column of fluid at a given depth. Here, it refers to the pressure exerted by a column of drilling mud or cement on the formation at a particular depth.

[2] Spacer fluid is a fluid pumped before the cement to clean drilling mud out of the wellbore.

annulus provided an indication that hydraulic fracturing had created a path from the producing formation to the well annulus in addition to the uncemented gas zones. Because the well was shut in, the pressure in the annulus could not be relieved, and the gas eventually traveled through natural fractures surrounding the wellbore into local drinking water aquifers (during the time the well was shut in, natural gas seeped into the well annulus and pressure built up from an initial pressure of 90 psi (0.6 MPa) to 360 psi (2.5 MPa)). From the aquifer, the gas moved into drinking water wells and from one of those wells into a cellar, resulting in the explosive accumulation of gas.

The Well File Review found that 3% of all hydraulic fracturing jobs (800 jobs) reported a mechanical integrity failure that allowed fluid to enter an annular space (U.S. EPA, 2016c).[1] The mechanical integrity failures generally resulted in hydraulic fracturing fluid entering the annular space between the casing and formation or between two casings, and were generally noted by increases in annular pressure or fluid bubbling to the surface. Other possible mechanisms for the failures include casing leaks, cement failure, and fractures extending above the height of the cement. (See Section 6.3.2.2 for additional information on fracture overgrowth.) While failures were noted, these do not necessarily indicate there was movement of fluid into a drinking water resource. In most cases, when problems occurred, the hydraulic fracturing operation was stopped and operators addressed the cause of the failure before hydraulic fracturing operations resumed; however, in 0.5% of the hydraulic fracturing jobs (100 jobs) with identified failures, there was no additional barrier between the annular space with fluid and protected drinking water resources.[2] While it could not definitively be determined whether fluid movement into the protected drinking water resource occurred, in these cases, all of the protective barriers intended to prevent such fluid migration failed, leaving the groundwater resource vulnerable to contamination.

While limited literature is available on construction (including cementing) flaws in hydraulically fractured wells, several studies have examined construction flaws in oil and gas wells in general. One study that examined reported drinking water contamination incidents in Texas identified 10 incidents related to drilling and construction activities among 250,000 oil and gas wells (Kell, 2011). The study noted that many of the contamination incidents were associated with wells that were constructed before Texas revised its regulations on cementing in 1969 (it is not clear how old the wells were at the time the contamination occurred). Because this study relied on reported incidents, it is possible that other wells exhibited mechanical integrity issues but did not result in contamination of a drinking water well or were not reported. Therefore, this should be considered a low-end estimate of the number of mechanical integrity issues that could be tied directly to drilling and construction activities. It is important to note that the 10 contamination incidents identified were not associated with wells that were hydraulically fractured (Kell, 2011).

Several investigators have studied violations information from the PA DEP online violation database to evaluate the rates of and possible factors contributing to mechanical integrity problems, including those related to cement. The results of these studies are summarized in Table 6-2.

[1] 800 jobs (95% confidence interval: 10 – 1,700 jobs).

[2] 100 jobs (95% confidence interval: 10 – 300 jobs).

Table 6-2. Results of studies of PA DEP violation data that examined mechanical integrity failure rates.

Study	Violations investigated	Wells studied	Data timeframe	Key findings[a]
Considine et al. (2012)	Violations resulting in environmental damage	3,533	2008–2011	Of 845 environmental damage incidents (which resulted in 1,144 violations), approximately 10% were related to casing or cement problems. The overall violation rate dropped from 52.9% of all wells in 2008 to 20.8% of all wells in 2011.
Davies et al. (2014)	Failure of one of the barriers preventing fluid migration	8,030	2005–2013	Approximately 5% of wells received this type of violation. The incident rate increased to 6.3% when failures noted on forms, but not resulting in violations, were included.
Ingraffea et al. (2014)	Violations and inspection records indicating structural integrity loss	3,391	2000–2012	Wells in unconventional reservoirs experienced a rate of structural integrity loss of 6.2%, while the rate for conventional wells was 1%.
Vidic et al. (2013)	Construction violations related to casing or cement	6,466	2008–2013	Approximately 3.4% of wells received this type of violation.
Olawoyin et al. (2013)	All violations	2,001	2008–2010	Analysis of 2,601 violations from 65 operators based on weighted risks found that potentially risky violations increased 342% over the study period, while total violations increased 110%.
Brantley et al. (2014)	Violations related to well construction issues	7,234	2005 – 2013	Over the period studied, a total of 3.4% of well operators received violations for construction issues. Violations in any given year ranged from 0.6% to 10.8%. Also, 0.24% of wells were cited for methane migration.

[a] While all of these studies used the same database, their results vary because they studied different timeframes and used different definitions of what violations constituted a mechanical integrity problem or failure.

Because a significant portion of Pennsylvania's recent oil and gas activity is in the Marcellus Shale, many of the wells in these studies were most likely used for hydraulic fracturing. For example, Ingraffea et al. (2014) found that approximately 16% of the oil and gas wells drilled in the state between 2000 and 2012 were completed in unconventional reservoirs, and nearly all of these wells were used for hydraulic fracturing. Wells drilled in unconventional reservoirs experienced higher rates of structural integrity loss, as defined by the authors, than conventional wells drilled during the same time period (Ingraffea et al., 2014). The authors did not compare rates of structural integrity loss in conventional wells that were and were not hydraulically fractured; they assumed that unconventional wells were hydraulically fractured and conventional wells were not.

Violation rates resulting in environmental damage among all Pennsylvania wells dropped from 52.9% in 2008 to 20.8% in 2011 (Considine et al., 2012), and the drop may be due to a number of factors. Violations related to failure of cement or other well components represented a minority of all well violations (i.e., among wells that were and were not hydraulically fractured). Of 845 events that caused environmental damage, including but not limited to contamination of drinking water resources, Considine et al. (2012) found that about 10% (85 events) were related to casing and cement problems. The rest of the incidents were related to site restoration and spills; the violations noted are confined to those incidents that caused environmental damage (i.e., the analysis excluded construction flaws that did not have adverse environmental effects). In addition, two wells (0.06%) were found to have contributed to methane migration into drinking water. Ingraffea et al. (2014) identified a significant increase in mechanical integrity problems such as casing leaks, sustained casing pressure, and insufficient cement from 2009 to 2011, rising from 5% to 6% of all newly drilled oil and gas wells, followed by a decrease beginning in 2012 to about 2% of all wells, a reduction of approximately 100 violations among 3,000 wells from 2011 to 2012. The rise in mechanical integrity problems between 2009 and 2011 coincided with an increase in the number of wells in unconventional reservoirs.

While all of the studies shown in the table used the same database, their results vary, not only because of the different timeframes studied, but also because they used different definitions of what violations constituted a mechanical integrity problem or failure. For example, Considine et al. (2012) considered all events resulting in environmental damage—including effects such as erosion—and found a relatively high violation rate. Davies et al. (2014) and Ingraffea et al. (2014) investigated violations related to mechanical integrity, while Vidic et al. (2013) looked only at mechanical integrity violations resulting in fluid migration out of the wellbore; these more specific studies found relatively lower violation rates. Olawoyin et al. (2013) performed a statistical analysis that weighted violations based on risk and found that the most risky violations included those involving pits, erosion, waste disposal, and blowout preventers.

Another source of information on contamination caused by wells is positive determination letters (PDLs) issued by the PA DEP. PDLs are issued in response to a complaint when the state determines that contamination did occur in proximity to oil and gas activities. The PDLs take into account the impact, timing, mechanical integrity, and formation permeability; liability is presumed for wells within a given distance if the oil and gas operator cannot refute that they caused the contamination, based on pre-drilling sampling (Brantley et al., 2014).[1] Brantley et al. (2014) examined these PDLs, and concluded that, between 2008 and 2012, the water supplies of approximately seven properties were impacted; depending on the assumptions used to determine how many unconventional gas wells affected a single property; this equates to a rate of 0.12 to 1.1% of the 6,061 wells begun in that timeframe. While these oil and gas wells were linked to contamination of wells and springs, the

[1] Under Pennsylvania's Oil and Gas Act, operators of oil or gas wells are presumed liable if water supplies within 1,000 ft (305 m) were impacted within 6 months of drilling, unless the claim is rebutted by the operator; this was expanded to 2,500 ft (762 m) and 12 months in 2012.

mechanisms for the impacts (including whether fluids may have been spilled at the surface or if there was a pathway through the well or through the subsurface rock formation to the drinking water resource) were not described by Brantley et al. (2014).

While the studies discussed above present possible explanations for higher violation incidences in unconventional wells that are likely to be hydraulically fractured, it should be noted that other explanations not specific to hydraulic fracturing are also possible. These could include different inspection protocols and different formation types.

Cementing in horizontal wells, which are commonly hydraulically fractured, presents challenges that can contribute to higher rates of mechanical integrity issues. The observation by Ingraffea et al. (2014) that wells drilled in unconventional reservoirs (which are horizontal in Pennsylvania) experience higher rates of structural integrity loss than conventional wells is supported by conclusions of Sabins (1990), who noted that horizontal wells have more cementing problems because they are more difficult to center properly and can be subject to settling of solids on the bottom of the wellbore. Cementing in horizontal wells presents challenges that can contribute to higher rates of mechanical integrity issues.

Thermal and cyclic stresses caused by intermittent operation also can stress cement (King and King, 2013; Ali et al., 2009). Increased pressures and cyclic stresses associated with hydraulic fracturing operations can contribute to cement integrity losses and, if undetected, small mechanical integrity problems can lead to larger ones. Temperature differences between the (typically warmer) subsurface environment and the (typically cooler) injected fluids, followed by contact with the (typically warmer) produced water, can lead to contraction of the well materials (both casing and cement), which introduces additional stresses. Similar temperature changes may occur when multiple fracturing stages are performed. Because the casing and cement have different mechanical properties, they may respond differently to these stress cycles and debond.

Several studies illustrate the effects of cyclic stresses. Dusseault et al. (2000) indicate that wells that have undergone several cycles of thermal or pressure changes will almost always show some debonding between cement and casing. Another laboratory study by De Andrade et al. (2015) found that cycling temperatures between 61°F and 151°F (16°C and 66°C) at 35 bar pressure (2.5 MPa) led to the formation of cracks in cement across both shale and sandstone formations. Cement damage was more significant in sandstone formations and worsened with each thermal cycle. A similar study by Roy et al. (2016) at ambient pressure did not find any cracks larger than 200 microns with temperature fluctuation between -40°F and 158°F (-40°C and 70°C), although numerical modeling of the same scenario predicted that cracks up to 1 to 10 microns would form, which would not have been detected by the methods used. Microannuli formed by this debonding can serve as pathways for gas migration, in particular because the lighter density of gas provides a larger driving force for migration through the microannuli than for heavier liquids.[1] One laboratory study indicated that microannuli on the order of 0.01 in (0.25 mm) could increase effective cement permeability from 1 nD (1×10^{-21} m^2) in good quality cement up to 1 mD (1×10^{-15} m^2) (Bachu and Bennion, 2009). This six-order magnitude increase in permeability shows that even small

[1] Microannuli can also form due to an inadequate cement job, e.g., poor mud removal or improper cement placement rate.

microannuli can significantly increase the potential for flow through the cement. Typically, these microannuli form at the interface between the casing and cement or between the cement and formation. Debonding and formation of microannuli can occur through intermittent operation, pressure tests, and workover operations (Dusseault et al., 2000).[1] While a small area of debonding may not lead to fluid migration, the microannuli in the cement resulting from the debonding can serve as initiation points for fracture propagation if re-pressurized gas enters the microannulus (Dusseault et al., 2000).

A number of modeling studies have indicated that fractures can propagate upwards from existing defects in cement or areas with poorer bonding (Kim et al., 2016; Roy et al., 2016; De Andrade et al., 2015). Feng et al. (2015) showed that fractures in cement tended to propagate upwards along the wellbore instead of radially. Modeling studies have also shown that cements with lower Young's modulus tend to propagate fractures more slowly than stiffer cements (Kim et al., 2016; Feng et al., 2015).[2]

The Council of Canadian Academies (2014) found that the repetitive pressure surges occurring during the hydraulic fracturing process would make maintaining an intact cement seal more of a challenge in these wells. Wang and Dahi Taleghani (2014) performed a modeling study, which concluded that hydraulic fracturing pressures could initiate annular cracks in cement. Another study of well data indicated that cement failure rates are higher in intermediate casings compared to other casings (McDaniel et al., 2014). The failures occurred after drilling and completion of wells, and the authors surmised that the cement failures were most likely due to cyclic pressure stresses caused by drilling. Theoretically, similar cyclic pressure events could also be experienced in the production casing during multiple stages of hydraulic fracturing. Mechanical stresses associated with well operation or workovers and pressure tests also may lead to small cracks in the cement, which may provide migration pathways for fluid.

Corrosion can lead to cement failure. Cement can fail to maintain integrity as a result of degradation of the cement after the cement is set. Cement degradation can result from attack by corrosive brines or chemicals such as sulfates, sulfides, and carbon dioxide that exist in formation fluids (Renpu, 2011). These chemicals can alter the chemical structure of the cement, resulting in increased permeability or reduced strength and leading to loss of cement integrity over time. Additives or specialty cements exist that can decrease cement susceptibility to specific chemicals.

6.2.2.3 Well Age

Hydraulic fracturing within older (legacy) wells has the potential to impact drinking water resources, either due to inadequate design and construction or degradation of the well components over time that afford pathways for the unintended migration of fluids. While new wells can be specifically designed to withstand the stresses associated with hydraulic fracturing operations,

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[1] A workover refers to any maintenance activity performed on a well that involves ceasing operations and removing the wellhead. Depending on the purpose of the workover and the tools used, workovers may induce pressure changes in the well.

[2] Young's modulus, a ratio of stress to strain, is a measure of the rigidity of a material.

older wells, which are sometimes used in hydraulic fracturing operations, may not have been designed to the same specifications, and their reuse for this purpose could be a concern.

Aging and extended use of a well contribute to casing corrosion and degradation, and the potential for fluid migration related to compromised casing tends to be higher in older wells. For example, exposure to corrosive chemicals such as hydrogen sulfide, carbonic acid, and brines can accelerate corrosion (Renpu, 2011). Ajani and Kelkar (2012) studied wells in Oklahoma and found a correlation between well age and mechanical integrity issues. Specifically, in wells spaced between 1,000 and 2,000 ft (300 and 600 m) from a well being fractured, the likelihood of impact on the well (defined in the study as a loss of gas production or increase in water production) rose from approximately 20% to 60% as the well's age increased from 200 days to over 600 days. Age was also found to be a factor in mechanical integrity problems in a study of wells drilled offshore in the Gulf of Mexico (Brufatto et al., 2003).

The Well File Review (U.S. EPA, 2016c, 2015n) provides evidence that fracturing does occur in older wells, including re-entering existing wells to fracture them for the first time or re-fracturing in wells that have been previously fractured. The Well File Review found that the median age of wells being initially fractured was 45 days, with well ages at time of fracturing ranging from 8 days to nearly 51 years. While 64% of the wells studied in the Well File Review were fractured within 6 months of the well spud date, the median age for wells being re-fractured was 6 years.[1,2] An estimated 11% of fracture jobs studied in the Well File Review were re-completions in a different zone than the original fracture job and 8% were re-fractures in the same zone as the original fracture job.[3,4]

The Well File Review also found that well component failures appeared to occur more frequently in older wells that were being re-completed or re-fractured.[5] The failure rate in hydraulic fracturing jobs involving re-completions and re-fractures was 6%, compared to 2% for hydraulic fracturing jobs in wells that had not been previously fractured.[6,7] While the confidence levels overlap, there is an indication that re-fractured and re-completed wells are more likely to suffer a failure of one or more components during hydraulic fracturing operations.

Frac strings, which are specialized pieces of casing inserted inside the production casing, can be used to protect older casing during fracturing. However, the effect of hydraulic fracturing on the cement on the production casing in older wells is unknown. One study on re-fracturing of wells noted that the mechanical integrity of the well was a key factor in determining the success or failure of the fracture treatment (Vincent, 2011). The Well File Review (U.S. EPA, 2016c) found that

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[1] Spudding refers to starting the well drilling process by removing rock, dirt, and other sedimentary material with the drill bit.

[2] 64% of wells (95% confidence interval: 48 – 77% of wells).

[3] 11% of jobs (95% confidence interval: 5 – 23% of jobs).

[4] 8% of jobs (95% confidence interval: 5 – 12% of jobs).

[5] The Well File Review defines a failure as a defect in a well component that allows fluid to flow into an annular space.

[6] 6% failure rate (95% confidence interval: 2 – 19% failure rate).

[7] 2% failure rate (95% confidence interval: 0.5 – 8% failure rate).

failures occurred more frequently in completions using frac strings, with failures occurring 20% of the time, compared to failures occurring 0.9% of the time when a frac string was not used.[1,2]

Note that there are also potential issues related to *where* these older wells are sited. For example, some wells could be in areas with naturally occurring subsurface faults or fractures that could not be detected or fully characterized with the technologies available at the time of construction. It is also possible that, in areas of historic petroleum exploration, old abandoned wells can be present which may have been improperly plugged or have degraded over time.[3] These wells could serve as pathways for fluid migration if they are located within the fracture network of the well; see Section 6.3.2.

6.2.2.4 Sustained Casing Pressure

Sustained casing pressure illustrates how the issues related to casing and cement discussed in the preceding sections can work together and be difficult to differentiate.[4] It is an indicator that pathways within the well related to the well's casing, cement, or both allowed fluid movement to occur. Sustained casing pressure can result from casing leaks, uncemented intervals, microannuli, or some combination of the three, which can be an indication that a well has lost mechanical integrity. Sustained casing pressure can be observed when an annulus (either the annulus between the tubing and production casing or between any two casings) is exposed to a source of nearly continuous elevated pressure. Goodwin and Crook (1992) found that sudden increases in sustained casing pressure occurred in wells that were exposed to high temperatures and pressures. Subsequent logging of these wells showed that the high temperatures and pressures led to shearing of the cement/casing interface and a total loss of the cement bond. Aly et al. (2015) demonstrated methods using a combination of chemical analysis, isotopic analysis, well logs, and drilling records to identify the most likely source of fluids causing sustained casing pressure.

Sustained casing pressure occurs more frequently in older wells and horizontal or deviated wells. One study found that sustained casing pressure becomes a greater concern as a well ages. Sustained casing pressure was found in less than 10% of wells that were less than a year old, but was present in up to 50% of 15-year-old wells (Brufatto et al., 2003). While these wells may not have been hydraulically fractured, the study demonstrates that older wells can exhibit more mechanical integrity problems. Fleckenstein et al. (2015) also found that older wells exhibited more barrier failures, including sustained casing pressure. They reported that 3.53% of the wells in the study with under-pressured intermediate gas zones developed sustained casing pressure, although it is likely the sustained casing pressure was due to poor well design (i.e., under older standards) rather

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[1] 20% failure rate (95% confidence interval: 10 – 36% failure rate).

[2] 0.9% failure rate (95% confidence interval: 0.8 – 1.0% failure rate).

[3] An abandoned well refers to a well that is no longer being used, either because it is not economically producing or it cannot be used because of its poor condition.

[4] Sustained casing pressure is pressure in any well annulus that is measurable at the wellhead and rebuilds after it is bled down, not caused solely by temperature fluctuations or imposed by the operator (Skjerven et al., 2011). If the pressure is relieved by venting natural gas from the annulus to the atmosphere, it will build up again once the annulus is closed (i.e., the pressure is sustained). The return of pressure indicates that there is a small leak in a casing or through uncemented or poorly cemented intervals that exposes the annulus to a pressured source of gas. It is possible to have pressure in more than one of the annuli.

than well age. Watson and Bachu (2009) found that a higher portion of deviated wells had sustained casing pressure compared to vertical wells. Increased pressures and cyclic stresses (Syed and Cutler, 2010) during hydraulic fracturing and difficulty in cementing horizontal wells (Sabins, 1990) also can lead to increased instances of sustained casing pressure (Muehlenbachs et al., 2012; Rowe and Muehlenbachs, 1999).

Sustained casing pressure can be a concern for several reasons. If the pressures are allowed to build up to above the burst pressure of the exterior casing or the collapse pressure of the interior casing, the casing may fail. Increased pressure can also cause gas or liquid to enter lower-pressured formations that are exposed to the annulus either through leaks or uncemented sections. Laboratory experiments by Harrison (1985) demonstrated that over-pressurized gas in the annulus could cause rapid movement of gas into drinking water resources if a permeable pathway exists between the annulus and the groundwater. Over-pressurization of the annulus is commonly relieved by venting the annulus to the atmosphere; however, this does not address the underlying problem in the well and can result in additional releases of methane to the atmosphere.

One example of an area where sustained casing pressure is common is Alberta, Canada, where 14% of the wells drilled since 1971 experienced serious sustained casing flow. This was defined in a study by Jackson and Dussealt (2014) as more than 10,594 ft^3 (300 m^3)/day at pressures higher than 0.48 psi/ft (11 kPa/m) of depth times the depth of the surface casing. Another study in the same area found gas in nearby drinking water wells had a composition consistent with biogenic methane mixing with methane from nearby coalbed methane and deeper natural gas fields (Tilley and Muehlenbachs, 2012).

In a few cases, sustained casing pressure in wells that have been hydraulically fractured may have been linked to drinking water contamination, although it is challenging to definitively determine the actual cause. In one study in northeastern Pennsylvania, methane to ethane ratios and isotopic signatures were used to investigate stray gas migration into domestic drinking water (U.S. EPA, 2014f). Composition of the gas in the water wells was consistent with that of the gas found in nearby gas wells with sustained casing pressures; other possible sources of the gas could not be ruled out. Several gas wells in the study area were cited by the PA DEP for having elevated sustained casing annulus pressures. One such case included four well pads with two wells drilled on each pad in southeastern Bradford County. The wells, drilled between September 2009 and May 2010, were 6,890 to 7,546 ft (2,100 to 2,300 m) deep and had surface casing to 984 ft (300 m). The casing below the surface casing was uncemented. All four wells experienced sustained casing pressure, with pressures ranging from 483 to 909 psi (3.3 to 6.3 MPa). Methane appeared in three nearby domestic drinking water wells in July 2010. Investigation into the cause of the methane contamination identified the drilled gas wells with sustained casing pressure as the most likely cause. The likely path was over-pressured gas from intermediate zones above the Marcellus Shale entering the uncemented well annulus and traveling up the annulus and along bedding planes which intersected the well annulus.[1] The determination was based on multiple lines of evidence, including: no methane present in a pre-drill sample, increases in methane after the wells had been

[1] A bedding plane is the surface that separates two layers of stratified rocks.

drilled, similar isotopic composition of the gas in the domestic wells and the gas in the annular space of the gas wells, and the presence of bedding planes which intersected the uncemented portion of the gas wells leading upwards toward the domestic wells (Llewellyn et al., 2015).

Adequate well design, detection (i.e., through annulus pressure monitoring), and repair of sustained casing pressure reduce the potential for fluid movement. (See Chapter 10 for additional discussion of practices that can reduce the frequency or severity of impacts to drinking water quality.) Watson and Bachu (2009) found that regulations requiring monitoring and repair of sustained casing vent flow or sustained casing pressure had a positive effect on lowering leak rates. The authors also found injection wells initially designed for the higher pressures associated with injection (vs. production) experienced sustained casing pressure less often than those that were retrofitted (Watson and Bachu, 2009). As mentioned above, Fleckenstein et al. (2015) found that placing the surface casing below all potential sources of drinking water and cementing intermediate gas zones significantly reduced sustained casing pressure.

Another study in Mamm Creek, Colorado, obtained similar results. The Mamm Creek field is in an area where lost cement and shallow, gas-containing formations are common. All the wells in the formation were hydraulically fractured (S.S. Papadopulos & Associates, 2008). A number of wells in the area have experienced sustained casing pressure, and methane has been found in several drinking water wells along with seeps into local creeks and ponds. In one well, drilled in January 2004, four pressured gas zones were encountered during drilling and there was a lost cement incident, which resulted in the cement top being more than 4,000 ft (1,000 m) lower than originally intended. Due to high bradenhead pressure (661 psi, or 4.6 MPa), cement remediation efforts were implemented (Crescent, 2011; COGCC, 2004).[1] The operator of this well was later cited by the Colorado Oil and Gas Conservation Commission (COGCC) for causing natural gas and benzene to seep into a nearby creek. The proposed route of contamination was contaminants flowing up the well annulus and then along a fault. The proposed contamination route appeared to be validated because, once remedial cementing was performed on the well, methane and benzene levels in the creek began to drop (Science Based Solutions LLC, 2014). In response to the incident, the state instituted requirements to identify and cement above the top of the highest gas-producing formation in the area and to monitor casing pressures after cementing.

A study in the Woodford Shale in Oklahoma examined how various cement design factors affected sustained casing pressure (Landry et al., 2015). The study focused on wells in the Cana-Woodford basin, a very deep basin at 11,000 to 15,000 ft (3,400 to 4,600 m) below ground surface, where the depth, long laterals, fracture gradients, and low permeability of the formations in the basin make cementing a challenge. One operator had seven test wells in the basin, of which six exhibited sustained casing pressure, usually after hydraulic fracturing operations. In early designs, the operator had not been using centralizers on the horizontal sections of the well, because they increased the frequency of stuck pipe. However, improvements in centralizer design allowed the operator to use centralizers more frequently on later well designs, and the operator tried several different techniques to address the sustained casing pressure problems, with varying results:

[1] Bradenhead pressure is pressure between two casings in an oil and gas well.

- In three of the wells, the operator used three different techniques: a conventional cement job with a water-based drilling mud and single slurry design; oil-based mud with single slurry design; and a foamed cement to cement the vertical portion of the well from the kickoff point up with conventional water-based cement on the lateral. All three of these wells experienced sustained casing pressure after hydraulic fracturing operations.
- In a fourth well, in 2013, the operator used centralizers, with three centralizers per every two casing joints along the lateral and one centralizer per joint in the vertical section. The design also involved an enhanced spacer fluid to remove drilling mud and a self-healing cement in the upper portion of the well. While some channeling was detected in this well, the channels were not connected and did not lead to sustained casing pressure.

The operator constructed an additional 21 wells using the same technique as was performed in the fourth well, and 20 did not show any sustained casing pressure after fracturing. This study shows the importance of cement design factors, such as casing centralization and mud removal, in preventing sustained casing pressure.

Not every well that shows positive pressure in the annulus poses a potential problem. Sustained pressure is only a problem when it exceeds the ability of the wellbore to contain it or when it indicates leaks in the cement or casing (TIPRO, 2012). A variety of management options are available for managing such pressure including venting, remedial cementing, and use of kill fluids in the annulus (TIPRO, 2012).[1] While venting may be a common method to address sustained casing pressure, it does not address the underlying mechanical integrity failure and is only a temporary solution. Furthermore, venting releases fluids at the wellhead which, if gaseous, can contribute to increased atmospheric emissions, or if liquid, potential spills on the surface.

6.3 Fluid Migration Associated with Induced Fractures within Subsurface Formations

This section discusses potential pathways for fluid movement associated with induced fractures and subsurface formations (outside of the well system described in Section 6.2). It examines the potential for fluid migration into drinking water resources by evaluating the development of migration pathways within subsurface formations, the flow of injected and formation fluids, and important factors that affect these processes.

Fluid movement requires both a physical pathway (e.g., via the interconnected pores within a permeable rock matrix or via a fracture in the rock) and a driving force.[2] In subsurface formations, fluid movement is driven by the existence of a hydraulic gradient, which depends on elevation and pressure and is influenced by fluid density, composition, and temperature (Pinder and Celia, 2006).

[1] A kill fluid is a weighted fluid with a density that is sufficient to overcome the formation pressure and prevent fluids from flowing up the wellbore.

[2] Permeability (i.e., intrinsic or absolute permeability) of formations describes the ability of water to move through the formation matrix, and it depends on the rock's grain size and the connectedness of the void spaces between the grains. Where multiple phases of fluids exist in the pore space, the flow of fluids also depends on relative permeabilities.

In the context of hydraulic fracturing, two key factors govern fluid migration during and after the hydraulic fracturing event:

- Pressure differentials in the reservoir, which are influenced both by initial subsurface conditions and by the pressures created by injection and production regimes. Specific factors that may influence pressure differentials include structural or topographic features, over-pressure in the shale reservoir, or a temporary increase in pressure as a result of fluid injection during hydraulic fracturing (Birdsell et al., 2015a).
- Buoyancy, which is driven by density differences among and between gases and liquids. Fluid migration can occur when these density differences exist in the presence of a pathway (Pinder and Gray, 2008).

During hydraulic fracturing, pressurized fluids leaving the well create fractures within the production zone and then enter the formation through these newly created (induced) fractures. Unintended fluid migration can result from this fracturing process. Migration pathways to drinking water resources could develop as a result of changes in the subsurface flow or pressure regime associated with hydraulic fracturing; via fractures that extend beyond the intended formation or that intersect existing natural faults or fractures; and via fractures that intersect offset wells or other artificial structures (Jackson et al., 2013d). These subsurface pathways may facilitate the migration of fluids by themselves or in conjunction with the well-based pathways described in Section 6.2. Fluids potentially available for migration include both fluids injected into the well (including leakoff) and fluids already present in the formation (including brine or natural gas).[1]

The potential for subsurface fluid migration into drinking water resources can be evaluated during two different time periods (Kim and Moridis, 2015):

1. *Following the initiation of fractures in the reservoir, prior to any oil or gas production.* The injected fluid, pressurizing the formation, flows through the fractures and the fractures grow into the reservoir. Fluid leaks off into the formation, allowing the fractures to close except where they are held open by the proppant (Adachi et al., 2007). Fractures will generally continue to propagate until the fluid lost to leakoff is equal to the fluid injection rate (King and Durham, 2015).
2. *During the production period, after fracturing is completed and pressure in the fractures is reduced.* At this time, fluids (including oil/gas and produced water) flow from the reservoir into the well. As fluids are withdrawn from the formation, pore pressure decreases; as a result, the effective stress applied to fractures increases and (in the absence of proppant) fractures will close (Aybar et al., 2015).

Note that these two time periods vary in duration. As described in Chapter 3, the first period of fracture creation and propagation (i.e., the hydraulic fracturing itself) is a relatively short-term process, typically lasting 2 to 10 days, depending on the number of stages in the fracture treatment

[1] Leakoff is the fraction of the injected fluid that infiltrates into the formation and is not recovered (i.e., it "leaks off" and does not return through the well to the surface) during production (Economides et al., 2007). Fluids that leak off and are not recovered are sometimes referred to as "lost" fluids.

design. On the other hand, operation of the well for production covers a substantially longer period (depending on many factors such as the amount of hydrocarbons in place and economic considerations), and can be as long as 40 or 60 years in onshore tight gas reservoirs (Ross and King, 2007).

The following discussion of potential subsurface fluid migration into drinking water resources focuses primarily on the physical movement of fluids and the factors affecting this movement. Section 6.3.1 describes the basic principles of subsurface fracture creation, geometry, and propagation, to provide context for the discussion of potential fluid migration pathways in Section 6.3.2. Geochemical and biogeochemical reactions among hydraulic fracturing fluids, formation fluids, subsurface microbes, and rock formations are another important component of subsurface fluid migration and transport. See Chapter 7 for a discussion of the processes that affect pore fluid biogeochemistry and influence the chemical and microbial composition of produced water.

6.3.1 Overview of Subsurface Fracture Growth

Fracture initiation and growth is a highly complex process due to the heterogeneous nature of the subsurface environment. As shown in Figure 6-5, fracture formation is controlled by the three in situ principal compressive stresses: the vertical stress, the maximum horizontal stress, and the minimum horizontal stress. During hydraulic fracturing, pressurized fluid injection creates high pore pressures around the well. Fractures form when this pressure exceeds the local least principal stress and the tensile strength of the rock (Zoback, 2010; Fjaer et al., 2008).

Fractures propagate (increase in length) in the direction of the maximum principal stress; they are tensile fractures that open in the direction of least resistance and then propagate in the plane of the greatest and intermediate stresses (Nolen-Hoeksema, 2013). Deep in the subsurface, the maximum principal stress is generally in the vertical direction, because the overburden (the weight of overlying rock) is the largest single stress. Therefore, in deep formations, fracture orientation is expected to be vertical. This is the scenario illustrated in Figure 6-5. At shallower depths, where the rock is subjected to less pressure from the overburden, more fracture propagation is expected to be in the horizontal direction. Using tiltmeter data from over 10,000 fractures in various North American shale reservoirs, Fisher and Warpinski (2012) found that induced fractures deeper than about 4,000 ft (1,000 m) are primarily vertical (see below for more information on tiltmeters). Between approximately 4,000 and 2,000 ft (1,000 and 600 m), they observed that fracture complexity increases, and fractures shallower than about 2,000 ft (600 m) are primarily (though not entirely) horizontal.[1] However, local geologic conditions can cause fracture orientations to deviate from these general trends (Ryan et al., 2015). Horizontal fracturing can also occur in deeper

[1] Fracture complexity is the ratio of horizontal-to-vertical fracture volume distribution, as defined by Fisher and Warpinski (2012). Fracture complexity is higher in fractures with a larger horizontal component. For the reasons explained above, this is more likely to occur at shallower depths. However, even in shallow zones, fractures are unlikely to be completely horizontal. As noted by Fisher and Warpinski, “All of the fractures do not necessarily turn horizontal; they might have significant vertical and horizontal components with more of a T-shaped geometry.” In the Fisher and Warpinski data set, the maximum horizontal component of the fractures is approximately 70%.

settings in some less-common reservoir environments where the principal stresses have been altered by salt intrusions or similar types of geologic activity (Jones and Britt, 2009).

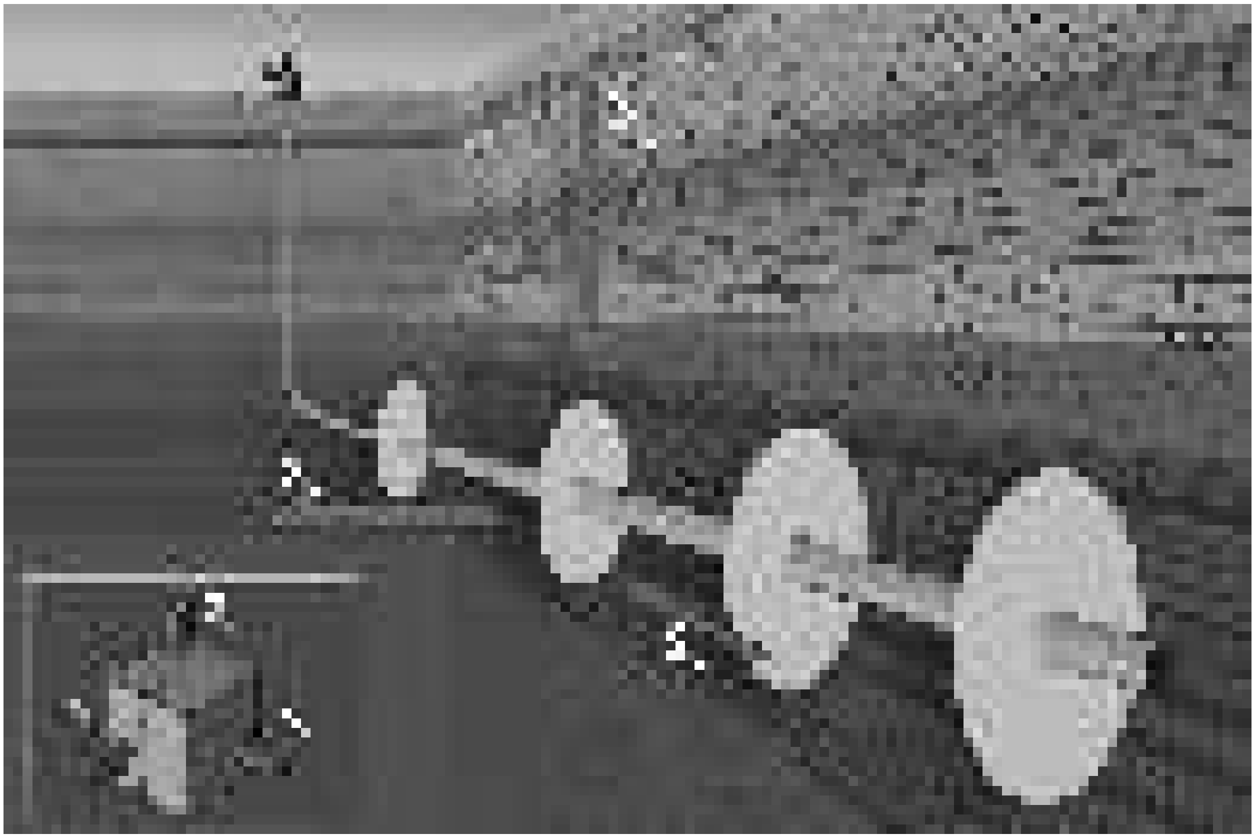


Figure 6-5. Hydraulic fracture planes (represented as ovals), with respect to the principal subsurface compressive stresses: S_V (the vertical stress), S_H (the maximum horizontal stress), and S_h (the minimum horizontal stress).

In addition to the principal subsurface stresses, a variety of factors and processes affect the complex process of fracture creation, propagation, geometry, and containment.[1] Computational modeling techniques have been developed to simulate fracture creation and propagation and to provide a better understanding of this complex process (Kim and Moridis, 2013).[2] Modeling hydraulic fracturing in shale or tight gas reservoirs requires integrating the physics of both flow and geomechanics to account for fluid flow, fracture propagation, and dynamic changes in pore volume and permeability. Some important flow and geomechanical parameters included in these

[1] Fracture geometry refers to characteristics of the fracture such as height and aperture (width).

[2] There are different kinds of mathematical models. Analytical models have a closed-form solution and therefore are relatively simple to solve. In contrast, computational models (also called numerical models) require more extensive computational resources and are used to study the behavior of complex systems.

types of advanced models are: permeability, porosity, Young's modulus, Poisson's ratio, and tensile strength, as well as heterogeneities associated with these parameters.[1]

Based on modeling and laboratory experiments (e.g., by Khanna and Kotousov, 2016; Li et al., 2016c; Li et al., 2016b; de Pater, 2015; Kim and Moridis, 2015; Lee et al., 2015; Narasimhan et al., 2015; Smith and Montgomery, 2015; Wang and Rahman, 2015; Kim and Moridis, 2013), below are some of the factors that have been noted in the literature as influencing fracture growth:

- Geologic properties of the production zone such as rock type and composition, permeability, thickness, and the presence of pre-existing natural fractures;
- The presence, composition, and properties of the liquids and gases trapped in pore spaces;
- Geomechanical properties, including tensile strength, Young's modulus, and the pressure at which the rock will fracture;
- Characteristics of the interface (boundary) between adjacent rock layers; and
- Operational characteristics, including injection rate and pressure, the properties of the hydraulic fracturing fluids, and fracture spacing.

Some modeling investigations have indicated that the vertical propagation of fractures (due to tensile failure) may be limited by shear failure, which increases the permeability of the formation and allows more fluid to leak off into the rock. These findings demonstrate that elevated pore pressure can cause shear failure, thus further affecting matrix permeability, flow regimes, and leakoff (Daneshy, 2009).

It is important to note that, while computational modeling is a useful tool to understand complex systems, modeling has limitations and associated uncertainties. All models rely on assumptions and simplifications, and there is, as stated by Ryan et al. (2015), "currently no single numerical approach that simultaneously includes the most important thermo-hydromechanical and chemical processes which occur during the migration of gas and fluids along faults and leaky wellbores." Uncertainties in selecting values for input parameters and potentially inadequate field data for model verification limit the reliability of model predictions.

In addition to their use in research applications, analytical and numerical modeling approaches are used to design hydraulic fracturing treatments and predict the extent of fractured areas (Adachi et al., 2007). Specifically, modeling techniques are used to assess the treatment's sensitivity to critical parameters such as injection rate, treatment volumes, fluid viscosity, and leakoff. Existing models range from simpler (typically two-dimensional) theoretical models to computationally more complicated three-dimensional models.

Monitoring of hydraulic fracturing operations can also provide insights into fracture development. Monitoring techniques involve both operational monitoring methods and "external" methods not

[1] As described in Section 6.2.2.2, Young's modulus, a ratio of stress to strain, is a measure of the rigidity of a material. Poisson's ratio is a ratio of transverse-to-axial (or latitudinal-to-longitudinal) strain, and it characterizes how a material is deformed under pressure. See Zoback (2010) for more information on the geomechanical properties of reservoir rocks.

directly related to the production operation. Operational monitoring refers to the monitoring of pressure and flow rate, along with related parameters such as fluid density and additive concentrations, using surface equipment and/or downhole sensors (Eberhard, 2011). This monitoring is conducted to ensure the operation is proceeding as planned and to determine if operational parameters need to be adjusted. Interpretation of pressure data can be used to better understand fracture behavior (Kim and Wang, 2014). For example, pressure data from previous hydraulic fracturing operations can indicate whether a geologic barrier to fracture growth exists and whether the barrier has been penetrated, or whether fractures have intersected with natural fractures or faults (API, 2015). Anomalies in operational monitoring data can also indicate whether an unexpected event has occurred, such as communication with another well (Section 6.3.2.3).

As described in Chapter 4, the volume of fluid injected is typically monitored and tracked to provide information on the volume and extent of fractures created (Flewelling et al., 2013). However, numerical investigations have found that reservoir gas flows into the fractures immediately after they open from hydraulic fracturing, and injection pressurizes both gas and water within the fracture to induce further fracture propagation (Kim and Moridis, 2015). Therefore, the fracture volume can be larger than the injected fluid volume. As a result, simple estimation of fracture volume based on the amount of injected fluid may underestimate fracture growth, and additional information (e.g., from geophysical monitoring techniques) is needed to accurately predict the extent of induced fractures.

External monitoring technologies can also be used to collect data on fracture characteristics and extent during hydraulic fracturing and/or production. These monitoring methods can be divided into near-wellbore and far-field techniques. Near-wellbore techniques include the use of tracers, temperature logs, video logs, and caliper logs that measure conditions in and immediately around the wellbore (Holditch, 2007). However, near-wellbore techniques and logs only provide information for, at most, a distance of two to three wellbore diameters from the well and are, therefore, not suited for tracking fractures for their entire length (Holditch, 2007).

Far-field methods, such as microseismic monitoring or tiltmeters, are used if the intent is to estimate fracture growth and height across the entire fractured reservoir area. Microseismic monitoring involves placing geophones in a position to detect the very small amounts of seismic energy generated during subsurface fracturing (Warpinski, 2009).[1] Monitoring these microseismic events gives an idea of the location and size of the fracture network, as well as the orientation and complexity of fracturing (Fisher and Warpinski, 2012). Using the results of microseismic monitoring in conjunction with other information, such as time-lapse, multicomponent seismic data (collected with surface surveys), can provide additional information for understanding fracture complexity and the interaction between natural and induced fractures (D'Amico and Davis, 2015). The Well File Review (U.S. EPA, 2016c) found that microseismic monitoring was conducted at 0.5% (100) of the hydraulic fracturing jobs studied.[2] Tiltmeters, which measure extremely small deformations in the earth, can be used to determine the direction and volume of the fractures and,

[1] Typical microseismic events associated with hydraulic fracturing have a magnitude on the order of -2.5 (negative two and half) (Warpinski, 2009).

[2] 100 jobs (95% confidence interval: 40 – 300 jobs).

within certain distances from the well, to estimate their dimensions (Lecampion et al., 2005). Other monitoring techniques, such as seismic surveys, can also be used to gather information about the subsurface environment. For example, Viñal and Davis (2015) demonstrated the use of time-lapse multi-component seismic surveys to monitor changes in the overburden due to hydraulic fracturing. Chapter 10 provides additional discussion of factors and practices, such as site monitoring, that can reduce the frequency or severity of impacts to drinking water quality.

6.3.2 Migration of Fluids through Pathways Related to Fractures/Formations

As described above, subsurface migration of fluids requires a pathway, induced or natural, with enough permeability to allow fluids to flow, as well as a hydraulic gradient physically driving the movement. The following subsections describe and evaluate potential pathways for the migration of hydraulic fracturing fluids, hydrocarbons, or other fluids from producing formations to drinking water resources. They also present cases where the existence of these pathways has been documented. The potential subsurface migration pathways are categorized as follows: (1) migration out of the production zone through pore space in the rock, (2) migration due to fracture overgrowth out of the production zone, (3) migration via fractures intersecting offset wells or other artificial structures, and (4) migration via fractures intersecting other geologic features, such as permeable faults or pre-existing natural fractures. Although these four potential pathways are discussed separately here, they may act in combination with each other or in combination with pathways along the well (as discussed in Section 6.2) to affect drinking water resources.

The possibility of fluid migration between a hydrocarbon-bearing formation and a drinking water resource can be related to the vertical distance between these formations (Reagan et al., 2015; Jackson et al., 2013d). In general, as the separation distance between the production zone and a drinking water aquifer decreases, the likelihood of upward migration of hydraulic fracturing to drinking water aquifers increases (Birdsell et al., 2015a). The separation distance between hydraulically fractured producing zones and drinking water resources (and these formations' depth from the surface) varies substantially among shale gas plays, coalbed methane plays, and other areas where hydraulic fracturing takes place in the United States (Figure 6-6 and Table 6-3). Many hydraulic fracturing operations target deep shale zones such as the Marcellus or Haynesville/ Bossier, where the vertical distance between the top of the shale formation and the base of drinking water resources may be 1 mi (1.6 km) or greater. This is reflected in the Well File Review, in which approximately half of the wells were estimated to have 5,000 ft (2,000 m) or more of measured distance along the wellbore between the point of the shallowest hydraulic fracturing and the operator-reported base of the protected groundwater resource (U.S. EPA, 2015n).[1] Similarly, in a review of FracFocus data from over 40,000 wells across the United States, Jackson et al. (2015) found that the median depth of wells used for hydraulic fracturing was 8,180 ft (2,490 m) and the mean depth was 8,290 ft (2,530 m).

[1] In the Well File Review, measured depth represents length along the wellbore, which may be a straight vertical distance below ground or may follow a more complicated path, if the wellbore is not straight and vertical. True vertical separation distances were not reported in the Well File Review. Measured distance along a well is equal to the true vertical distance only in straight, vertical wells. Otherwise, the true vertical distance is less than the measured distance.

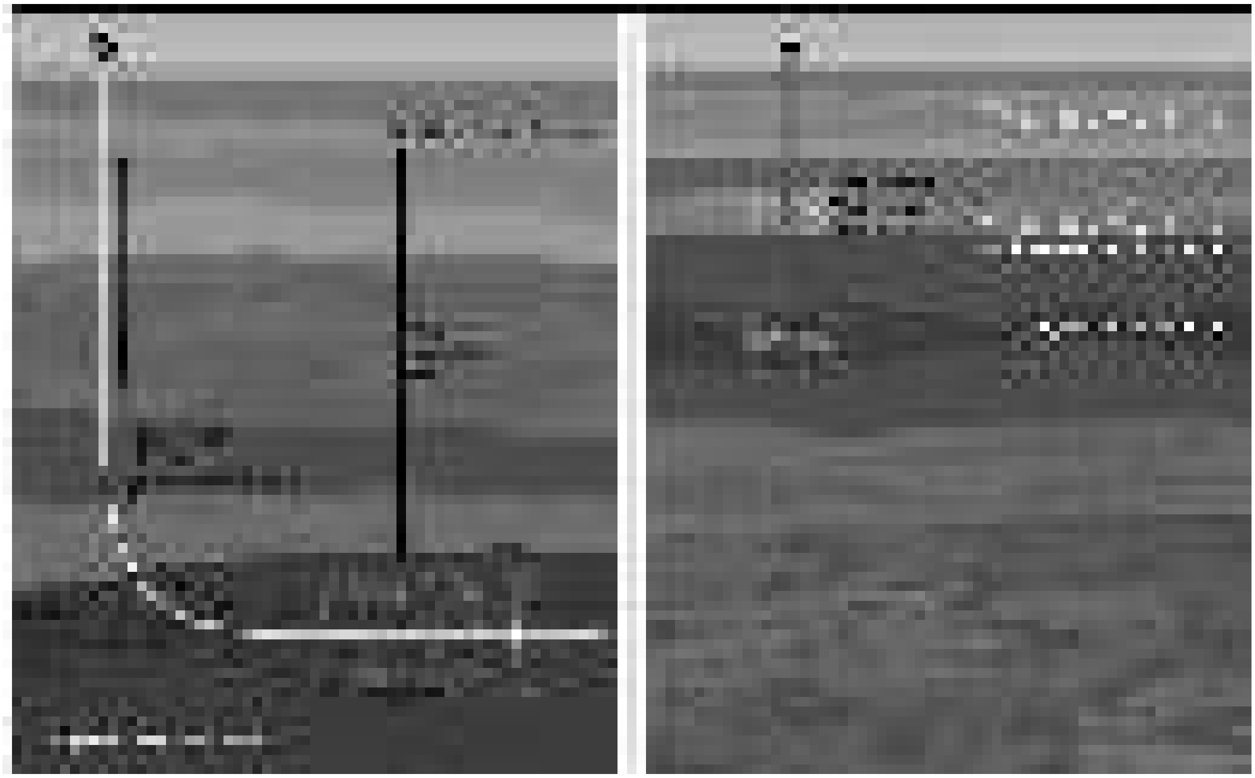


Figure 6-6. Vertical distances in the subsurface separating drinking water resources and hydraulic fracturing depths.

However, as shown in Table 6-3, some hydraulic fracturing operations occur at shallower depths or in closer proximity to drinking water resources. For example, both the Antrim and the New Albany plays are relatively shallow, with distances of 100 to 1,900 ft (31 to 580 m) between the producing formation and the base of drinking water resources. In the Jackson et al. (2015) review of FracFocus data, 16% of wells reviewed were within 1 mi (1.6 km) of the surface and 3% were within 2,000 ft (600 m) of the surface.[1] The distribution of the more shallow hydraulically fractured wells varied nationally but was concentrated in Texas, California, Arkansas, and Wyoming. For example, in California and Arkansas, 88% and 85% of hydraulically fractured wells, respectively, were within about 5,000 ft (2,000 m) of the surface. Overall, the Well File Review found a higher proportion of relatively shallow wells—the data in the Well File Review indicated that 20% of wells used for hydraulic fracturing (an estimated 4,600 wells) had less than 2,000 ft (600 m) between the shallowest point of the fractures and the base of protected groundwater resources (U.S. EPA, 2015n).[2] This is likely because the Well File Review results are more representative of hydraulic fracturing operations across the country; Jackson et al. (2015) acknowledge that their analysis

[1] Jackson et al. (2015) use true vertical depth data from FracFocus; this represents the depth of the well but not necessarily the depth of the fractures. The depth of the fractures may be shallower than the true vertical depth of the well, though Jackson et al. (2015) note that most states do not require operators to submit information on the true vertical depth to the top of the fractures.

[2] 4,600 wells (95% confidence interval: 900 – 8,300 wells). The Well File Review defines this separation distance as the measured depth of the point of shallowest hydraulic fracturing in the well, minus the depth of the operator-reported protected groundwater resource.

underestimates the occurrence of relatively shallow hydraulic fracturing for states in which FracFocus reporting is not required.

Table 6-3. Comparing the approximate depth and thickness of selected U.S. shale gas plays and coalbed methane basins.

Shale data are reported in GWPC and ALL Consulting (2009) and NETL (2013); coalbed methane data are reported in ALL Consulting (2004) and U.S. EPA (2004a). See Chapter 3 for information on the locations of these basins, plays, and formations.

Basin/play/ formation[a]	Approx. depth (ft [m] below surface)	Approx. net thickness (ft [m])	Distance between top of production zone and base of treatable water (ft [m])[b]
Shale plays			
Antrim	600 to 2,200 [200 to 670]	70 to 120 [20 to 37]	300 to 1,900 [90 to 580]
Barnett	6,500 to 8,500 [2,000 to 2,600]	100 to 600 [30 to 200]	5,300 to 7,300 [1,600 to 2,200]
Eagle Ford	4,000 to 12,000 [1,000 to 3,700]	250 [76]	2,800 to 10,800 [850 to 3,290]
Fayetteville	1,000 to 7,000 [300 to 2,000]	20 to 200 [6 to 60]	500 to 6,500 [200 to 2,000]
Haynesville-Bossier	10,500 to 13,500 [3,200 to 4,120]	200 to 300 [60 to 90]	10,100 to 13,100 [3,080 to 3,990]
Marcellus	4,000 to 8,500 [1,000 to 2,600]	50 to 200 [20 to 60]	2,125 to 7,650 [648 to 2,330]
New Albany	500 to 2,000 [200 to 600]	50 to 100 [20 to 30]	100 to 1,600 [30 to 490]
Woodford	6,000 to 11,000 [2,000 to 3,400]	120 to 220 [37 to 67]	5,600 to 10,600 [1,700 to 3,230]
Coalbed methane basins			
Black Warrior (Upper Pottsville)	0 to 3,500 [0 to 1,100]	< 1 to > 70 [< 1 to > 20]	As little as zero[c]
Powder River (Fort Union)	450 to >6,500 [140 to >2,000]	75 [23]	As little as zero[c]
Raton (Vermejo and Raton)	< 500 to > 4,100 [< 200 to > 1,300]	10 to >140 [3 to >43]	As little as zero[c]
San Juan (Fruitland)	550 to 4,000 [170 to 1,000]	20 to 80 [6 to 20]	As little as zero[c]

[a] For coalbed methane, values are given for the specific coal units noted in parentheses.

[b] The base of treatable water is defined at the state level; the information in the table is based on depth data from state oil and gas agencies and state geological survey data.

[c] Formation fluids in producing formations meet the salinity threshold that is used in some definitions of a drinking water resource in at least some areas of the basin. See the discussion after Text Box 6-5 for more information about this definition.

In coalbed methane plays, which are typically shallower than shale gas plays, vertical separation distances can be even smaller. In the Raton Basin of southern Colorado and northern New Mexico, approximately 10% of coalbed methane wells have less than 675 ft (206 m) of separation between the gas wells' perforated intervals and the depth of local water wells. In certain areas of the basin, this distance is less than 100 ft (31 m) (Watts, 2006). In California, nearly half of the hydraulic fracturing has occurred at depths less than about 900 ft (300 m) (CCST, 2015b), with hundreds of wells in the San Joaquin Valley between 150 ft (46 m) and 2,000 ft (600 m) deep (Jackson et al., 2015).

Some hydraulic fracturing operations are conducted within formations containing drinking water resources (Table 6-3). One example of hydraulic fracturing taking place within a geologic formation that is also used as a drinking water source is in the Wind River Basin in Wyoming (Digiulio and Jackson, 2016; WYOGCC, 2014b; Wright et al., 2012). Vertical gas wells in this area target the lower Wind River Formation and the underlying Fort Union Formation, which consist of interbedded layers of sandstones, siltstones, and mudstones. The Wind River Formation also serves as the principal source of domestic, municipal, and agricultural water in this rural area. There are no laterally continuous confining layers of shale in the basin to prevent upward movement of fluids. While flow in the basin generally tends to be downward, local areas of upward flow have been documented (Digiulio and Jackson, 2016). Assessing the relative depths of drinking water resources and hydraulic fracturing operations near Pavillion, Wyoming, Digiulio and Jackson (2016) found that approximately 50% of fracture jobs were within 1,969 ft (600 m) of the deepest domestic drinking water well in the area, and that 10% were within 820 ft (250 m) (Digiulio and Jackson, 2016). Among the wells evaluated by DiGiulio and Jackson, the shallowest fracturing occurred at 1,057 ft (322 m) below ground surface, which is comparable to depths targeted for drinking water withdrawal in the formation. See Text Box 6-5 for more information on Pavillion, Wyoming.

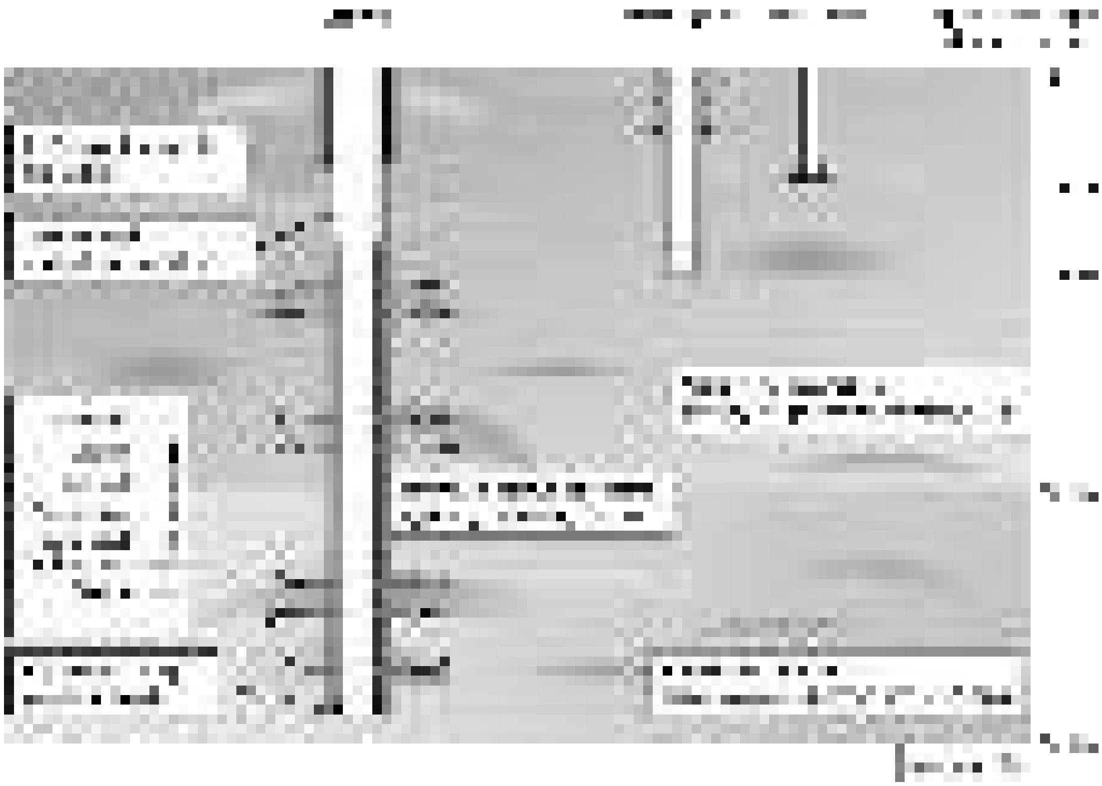
Text Box 6-5. Pavillion, Wyoming.

The Pavillion gas field is located east of the town of Pavillion, Wyoming. In addition to gas production, the field is also home to rural residences that rely on approximately 40 private wells to supply drinking water. The oldest known domestic water well in the field dates to 1934 (AME, 2016). Gas production in the field began in 1960 and, by the 2000s, it had grown to producing from at least 180 wells. Most of these gas wells were drilled since 1990, and approximately 140 to 145 were not plugged as of mid-2016 (AME, 2016; Digiulio and Jackson, 2016).

In the Pavillion gas field the same geologic formation that is used to produce hydrocarbons supplies the area's drinking water (Digiulio and Jackson, 2016). Water wells draw from the Wind River Formation, and gas is extracted from both the Wind River Formation and the underlying Fort Union Formation. The Wind River Formation contains variably permeable strata with lenses of relatively higher permeability rock enriched with natural gas. Water quality is typically freshest nearer the surface, and there is no rock formation acting as a natural barrier to separate the drinking water from hydrocarbons (Digiulio and Jackson, 2016). There is approximately 200 ft (60 m) vertical distance separating the deepest domestic well in the field from the shallowest hydraulic fracturing, although there is approximately 2.5 mi (4 km) lateral distance between them (AME, 2016; Digiulio and Jackson, 2016).

(Text Box 6-5 is continued on the following page.)

Text Box 6-5 (continued). Pavillion, Wyoming.



Following complaints by area residents about changes to their water quality in the mid-2000s, state and federal agencies began a series of investigations, centering on various aspects of the site and supporting differing conclusions about the source and mechanism of the water quality changes (AME, 2016).

Twenty-five pits that were used to dispose of drill cuttings, drilling mud, and spent drilling fluids near some of the water wells were also investigated as a potential source of the groundwater contamination. Based on these evaluations, soil and/or groundwater remediation was performed at approximately six of the pits, no further action was recommended at approximately twelve pits, and the remaining pits are receiving further investigation (AME, 2016).

Samples collected from two monitoring wells at depths between those of the drinking water and active intervals in gas production wells show elevated pH, unexpectedly high potassium values, and several organic constituents, including natural gas, alcohols, phenols, glycols, and benzene, toluene, ethylbenzene, and xylenes (BTEX) (Digiulio and Jackson, 2016). The potential source of chemicals in these two monitoring wells include formation water, contaminants remaining after well construction (AME, 2016) and hydraulic fracturing and other oil and gas activities (Digiulio and Jackson, 2016).

Water samples collected from domestic wells contain dissolved methane and some contain high sodium and sulfate concentrations. Organic chemicals have also been detected in some domestic wells (AME, 2016; Digiulio and Jackson, 2016). These same investigators suspect that pit proximity explains the origin of organic chemicals. In addition, natural gases from intermediate depths not hydraulically fractured are likely moving along some gas wellbores, potentially into zones used for drinking water (AME, 2016).

(Text Box 6-5 is continued on the following page.)

Text Box 6-5 (continued). Pavillion, Wyoming.

Of about 40 production wells at which pressure was measured on the bradenhead annulus between the production and surface casings, about 25% exhibited sustained casing pressure consistent with an ongoing source of gas and/or liquid. Gas samples collected from bradenhead annuli, production tubing and casing, and water wells indicate that the samples have similar gas compositions. This suggests a common origin, which is consistent with long-term migration from a deeper source (AME, 2016; WYOGCC, 2014b).

Production wells may be the source of gas migration, and groundwater immediately around some of the disposal pits has been affected (AME, 2016). However, the investigative reports conclude that identifying the precise source(s) of the water quality issues is challenging due to the lack of comprehensive pre-drilling water quality and other baseline monitoring, the unique hydrogeologic setting, and the difficulty of identifying specific geologic or well pathways.

In other cases, hydraulic fracturing takes place in formations that are not currently being used as sources of drinking water, but that meet the salinity threshold that is used in some definitions of drinking water resources.[1] This occurs in low-salinity coal-bearing formations in the Raton Basin of Colorado (U.S. EPA, 2015k), the San Juan Basin of Colorado and New Mexico (U.S. EPA, 2004a), the Powder River Basin of Montana and Wyoming (as described in Chapter 7), and in several other coalbed methane plays. Hydraulic fracturing in these regions occurs in formations characterized by total dissolved solids (TDS) values substantially lower than the 10,000 mg/L TDS value used in the federal definition of an underground source of drinking water.[2] Across various basins, coalbed methane operations have been reported to occur in formations with 300 to 3,000 mg/L TDS and at depths as shallow as 350 ft (110 m) (U.S. EPA, 2004a). In one field in Alberta, Canada, there is evidence that fracturing in the same formation as a drinking water resource (in combination with mechanical integrity problems; see Section 6.2.2.4) led to gas migration into water wells (Tilley and Muehlenbachs, 2012).

California is another area where hydraulic fracturing occurs in shallow zones with low-salinity groundwater. A study by the California Council on Science and Technology (CCST, 2015b) found that 3% of the hydraulic fracturing in the state occurred within 2,000 ft (600 m) of the surface. In California's San Joaquin Valley, hydraulic fracturing appears to have been conducted in formations with a TDS of less than 1,500 mg/L (CCST, 2014). Another study in California examined the TDS values of water samples taken during oil and gas activities and found that 15% to 19% of the oil and

[1] For the purposes of this discussion, the federal definition of an underground source of drinking water is used. Pursuant to 40 CFR 144.3, an underground source of drinking water is "an aquifer or its portion which supplies any public water system; or which contains a sufficient quantity of groundwater to supply a public water system; and currently supplies drinking water for human consumption; or contains fewer than 10,000 mg/L TDS; and which is not an exempted aquifer." This definition is used by the EPA's Underground Injection Control Program, which regulates injection wells (but not hydrocarbon production wells).

[2] This salinity threshold is used as a point of comparison only. While the definition of an underground source of drinking water is not exactly the same as the definition of a drinking water resource (and many states have their own definitions of protected drinking water zones), the former provides a useful frame of reference when considering the ability of an aquifer to potentially serve as a source of drinking water.

gas activities in Kern County, California, occurred within zones containing water with less than 3,000 mg/L TDS (Kang and Jackson, 2016).[1]

The overall frequency at which hydraulic fracturing occurs in formations that meet the definition of drinking water resources across the United States is uncertain. Some information, however, that provides insights on the occurrence and geographic distribution of this practice is available. According to the Well File Review, an estimated 0.4% (90) of the 23,200 wells represented in that study had perforations used for hydraulic fracturing that were placed shallower than the base of the protected groundwater resources reported by well operators (U.S. EPA, 2015n).[2] Additional information is available from a database of produced water composition data maintained by the U.S. Geological Survey (USGS). The USGS produced water database contains results from analyses of samples of produced water, including (among other data) samples collected from more than 8,500 oil and gas production wells in unconventional formations (coalbed methane, shale gas, tight gas, and tight oil) within the contiguous United States.[3] Just over 5,000 of these samples, which were obtained from wells located in 37 states, reported TDS concentrations. Because the database does not track whether samples were from wells that were hydraulically fractured, the EPA selected samples from wells that were more likely to have been hydraulically fractured by restricting samples to those collected in 1950 or later and to those that were collected from wells producing from tight gas, tight oil, shale gas, or coalbed methane formations.[4] This yielded 1,650 samples from wells located in Alabama, Colorado, North Dakota, Utah, and Wyoming, with TDS concentrations ranging from approximately 90 mg/L to 300,000 mg/L.[5] Of the 1,650 samples, approximately 1,200 (from wells in Alabama, Colorado, Utah, and Wyoming) reported TDS concentrations at or below 10,000 mg/L, indicating that hydraulic fracturing there may have occurred within formations that meet the salinity threshold that is used in some definitions of a drinking water resource. This analysis, in conjunction with the result from the Well File Review, suggests that the overall frequency of this occurrence is relatively low, but is concentrated in particular areas of the country.

6.3.2.1 Flow of Fluids Out of the Production Zone

One potential pathway for fluid migration out of the production formation into drinking water resources is advective or dispersive flow of injected or displaced fluids through the formation matrix. In this scenario, fluids (such as those "lost" to leakoff, which are not recovered during

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[1] Kern County accounts for 85 percent of the hydraulic fracturing that occurs in California (CCST, 2015b).

[2] 90 wells (95% confidence interval: 10 – 300 wells).

[3] The EPA used the USGS Produced Water Geochemical Database Version 2.1 (USGS database v 2.1) for this analysis (http://energy.cr.usgs.gov/prov/prodwat/). The database is comprised of produced water samples compiled by the USGS from 25 individual databases, publications, or reports.

[4] See Chapter 3, Text Box 3-1, which describes how commercial hydraulic fracturing began in the late 1940s.

[5] For this analysis, the EPA assumed that produced water samples collected in 1950 or later from shale gas, tight oil, and tight gas wells were from wells that had been hydraulically fractured. To estimate which coal bed methane wells had been hydraulically fractured, the EPA matched API numbers from coal bed methane wells in the USGS database v 2.1 to the same API numbers in the commercial database DrillingInfo, in which hydraulically fractured wells had been identified by the EPA using the assumptions described in Section 3.4. Wells with seemingly inaccurate (i.e., less than 12 digit) API numbers were also excluded. Only coalbed methane wells from the USGS database v 2.1 that matched API numbers in the DrillingInfo database were retained for this analysis.

production) would flow through the pore spaces of rock formations, moving from the production zone into other formations. In deep, low-permeability shale and tight gas settings and where induced fractures are contained within the production zone, flow through the production formation has generally been considered an unlikely pathway for migration into drinking water resources (Jackson et al., 2013d).

Leakoff into shale gas formations can be as high as 90% or more of the injected volume (Table 7-2). The actual amount of leakoff depends on multiple factors, including the amount of injected fluid, the concentration of different components in the fracture fluid, the hydraulic properties of the reservoir (e.g., permeability), the composition of the formation matrix, the capillary pressure near the fracture faces, and the period of time the well is shut in following hydraulic fracturing before the start of production (Kim et al., 2014; Byrnes, 2011).[1,2] Researchers generally agree that the subsequent flow of this "lost" leakoff fluid is controlled or limited by processes such as imbibition by capillary forces and adsorption onto clay minerals (Dutta et al., 2014; Dehghanpour et al., 2013; Dehghanpour et al., 2012; Roychaudhuri et al., 2011) and osmotic forces (Zhou, 2016; Wang and Rahman, 2015; Engelder et al., 2014).[3,4] It has been suggested that these processes can sequester the fluids in the producing formations permanently or for geologic time scales (Engelder et al., 2014; Engelder, 2012; Byrnes, 2011). Birdsell et al. (2015b) made quantitative estimates of the amount of fluid that could be imbibed in shale formations. Their results indicate that between 15% and 95% of injected fluid volumes may be imbibed in shale gas systems, while amounts are lower in shale oil systems (3% to 27% of injected volumes). In modeling investigations, O'Malley et al. (2015) found that it is likely that most hydraulic fracturing fluid that does not flow back is stored in rock pore spaces (i.e., having displaced the gas that was present there) and not fractures. The amount that can be stored in fractures is highly dependent on the effective interconnected pore lengths.

If the injected fluid is not sequestered in the immediate vicinity of the fracture network, migration into drinking water resources would likely require a substantial upward hydraulic gradient (e.g., due to the pressures introduced during injection for hydraulic fracturing), particularly for brine that is denser than the groundwater in the overlying formations (Flewelling and Sharma, 2014). In the presence of natural gas, buoyancy of the less dense gas could potentially provide an upward flux (Vengosh et al., 2014). However, Flewelling and Sharma (2014) indicated that pressure

[1] Relative permeability is a dimensionless property allowing for the comparison of the different abilities of fluids to flow in multiphase settings. If a single fluid is present, its relative permeability is equal to 1, but the presence of multiple fluids generally inhibits flow and decreases the relative permeability (Schlumberger, 2014).

[2] Shutting in the well after fracturing allows fluids to move farther into the formation, resulting in a higher gas relative permeability near the fracture surface and improved gas production (Bertoncello et al., 2014).

[3] Imbibition is the displacement of a nonwetting fluid (i.e., gas) by a wetting fluid (typically water). The terms wetting or nonwetting refer to the preferential attraction of a fluid to the surface. In typical reservoirs, water preferentially wets the surface, and gas is nonwetting. Capillary forces arise from the differential attraction between immiscible fluids and solid surfaces; these are the forces responsible for capillary rise in small-diameter tubes and porous materials. These definitions are adapted from Dake (1978).

[4] The contrast in water activity between brine and fresh water generates very substantial osmotic pressure differences that will drive fluids into the shale matrix. The osmosis process requires a semi-permeable membrane and a concentration gradient to allow the solvent to pass through it. The clay in the shale formation can provide a function similar to a membrane (Zhou, 2016).

perturbations due to hydraulic fracturing operations are localized to the immediate vicinity of the fractures, due to the very low permeabilities of shale formations; this means that hydraulic fracturing operations are unlikely to generate sufficient pressure to drive fluids into shallow drinking water zones. Some natural conditions could also create an upward hydraulic gradient in the absence of any effects from hydraulic fracturing. However, these natural mechanisms have been found to cause very low flow rates over very long distances, yielding extremely small vertical fluxes in sedimentary basins. These translate to some estimated travel times of 100,000 to 100,000,000 years across a 328 ft (100 m) thick layer with about 0.01 nD (1×10^{-23} m^2) permeability (Flewelling and Sharma, 2014). In an area of the Permian Basin with over-pressured source rocks, Engle et al. (2016) concluded that chemical, isotopic, and pressure data suggest that there is little potential for vertical fluid migration to shallow zones in the absence of pathways such as improperly abandoned wells (Section 6.3.2.3).

To account for the combined effect of capillary imbibition, well operation, and buoyancy in upward fluid migration, Birdsell et al. (2015a) conducted a numerical analysis over five phases of activity at a hypothetical Marcellus-like hydraulic fracturing site: a pre-drilling steady state, the injection of fluids, a shut-in period, production, and the continued migration of hydraulic fracturing fluids after the well is plugged and abandoned. They quantified how much hydraulic fracturing fluid flows back up the well after fracturing, how much reaches overlying aquifers, and how much is permanently sequestered by capillary imbibition (which is treated as a sink term). Their results affirmed that, without a pathway such as a permeable fault or leaky wellbore, it is very unlikely that hydraulic fracturing fluid from a deep shale could reach an overlying aquifer. However, the study did indicate that upward migration on the order of 328 ft (100 m) could occur through relatively low-permeability overburden, even if no discrete, permeable pathway exists.

6.3.2.2 Fracture Overgrowth out of the Production Zone

Fractures extending out of the intended production zone into another formation, or into an unintended zone within the same formation, could provide a potential fluid migration pathway into drinking water resources (Jackson et al., 2013d). This migration could occur either through the fractures themselves or in connection with other permeable subsurface features or formations (Figure 6-7). Such “out-of-zone fracturing” is undesirable from a production standpoint and may occur as a result of inadequate reservoir characterization or fracture treatment design (Eisner et al., 2006). Some researchers have noted that fractures growing out of the targeted production zone could potentially contact other formations, such as higher conductivity sandstones or conventional hydrocarbon reservoirs, which may create an additional pathway for migration into a drinking water resource (Reagan et al., 2015). In addition, fractures growing out of the production zone could potentially intercept natural, preexisting fractures (discussed in Section 6.3.2.4) or active or abandoned wells near the well where hydraulic fracturing is performed (discussed in Section 6.3.2.3).

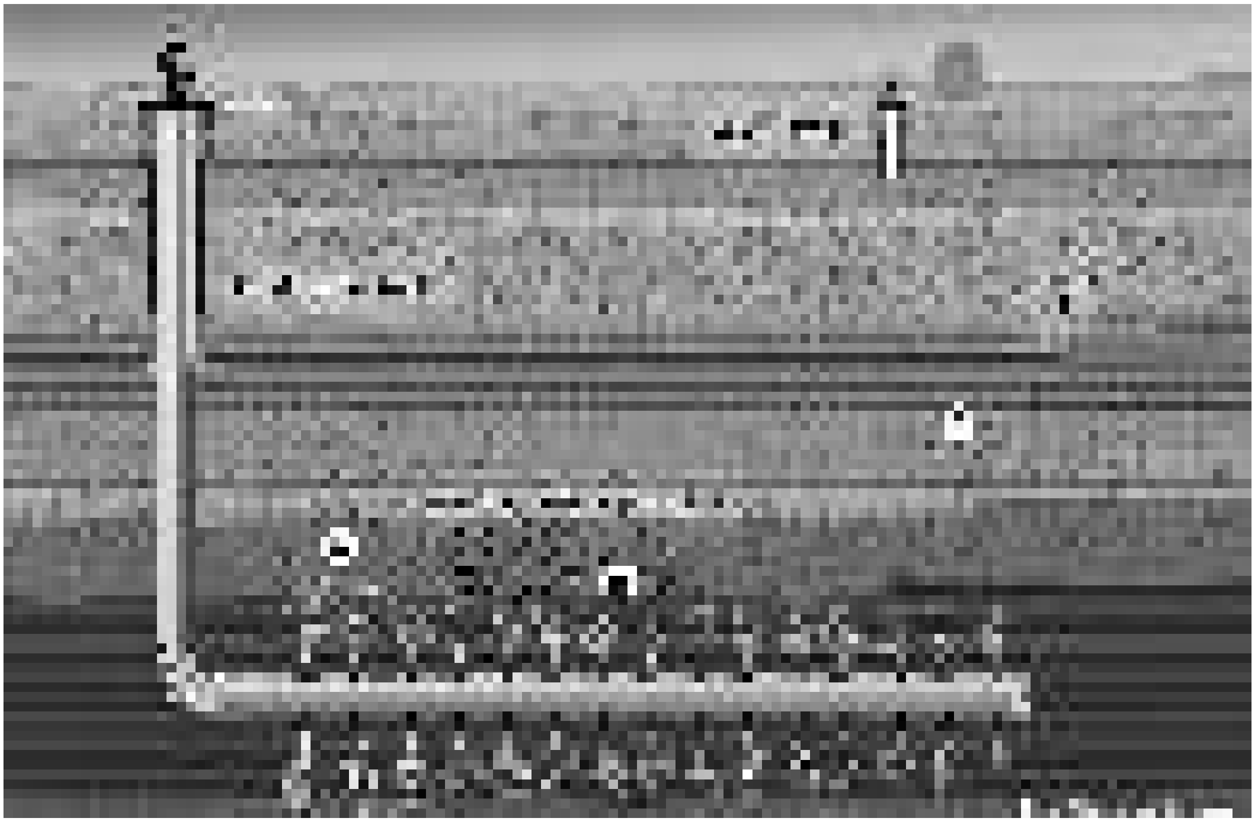

Figure 6-7. Conceptualized depiction of potential pathways for fluid movement out of the production zone: (a) induced fracture overgrowth into over- or underlying formations; (b) induced fractures intersecting natural fractures; and (c) induced fractures intersecting a permeable fault.

The fracture's geometry (Section 6.3.1) affects its potential to extend beyond the intended zone and serve as a pathway to drinking water resources. Vertical heights of fractures created during hydraulic fracturing operations have been measured in several U.S. shale plays, including the Barnett, Woodford, Marcellus, and Eagle Ford, using microseismic monitoring and tiltmeters (Fisher and Warpinski, 2012). These data indicate typical fracture heights extending from tens to hundreds of feet.[1] Davies et al. (2012) analyzed this data set and found that the maximum fracture height was 1,929 ft (588 m) and that 1% of the fractures had a height greater than 1,148 ft (350 m). This may raise some questions about fractures being contained within the producing formation, as some Marcellus fractures were found to extend vertically for at least 1,500 ft (460 m), while the maximum thickness of the formation is generally 350 ft (110 m) or less (MCOR, 2012). However, the majority of fractures within the Marcellus were found to have heights less than 328 ft (100 m), suggesting limited possibilities for fracture overgrowth exceeding the separation between shale reservoirs and shallow aquifers (Davies et al., 2012). This is consistent with modeling results found by Kim and Moridis (2015) and others, as described below. Where the producing formation is not

[1] As described in Section 6.3.1, microseismic data represent the small amounts of seismic energy generated during subsurface fracturing. The Fisher and Warpinski dataset includes the top and bottom depths of mapped fracture treatments in the four shale plays mentioned, giving the maximum propagation length.

continuous horizontally, the lateral extent of fractures may also become important. For example, in the Fisher and Warpinski (2012) data set, fractures were found to extend to horizontal lengths greater than 1,000 ft (300 m).

Results of National Energy Technology Laboratory (NETL) research in Greene County, Pennsylvania, are generally consistent with those reported in the Fisher and Warpinski (2012) data set. Microseismic monitoring was used at six horizontal Marcellus Shale wells to identify the maximum upward extent of brittle deformation (i.e., rock breakage) caused by hydraulic fracturing (Hammack et al., 2014). At three of the six wells, fractures extending between 1,000 and 1,900 ft (300 and 580 m) above the Marcellus Shale were identified. Overall, approximately 40% of the microseismic events occurred above the Tully Limestone, the formation overlying the Marcellus Shale. The microseismic data suggest that fracture propagation occurs above the Tully Limestone, which is sometimes referred to as an upper barrier to hydraulic fracture growth (Hammack et al., 2014). However, all microseismic events were at least 5,000 ft (2,000 m) below drinking water aquifers, as the Marcellus Shale is one of the deepest shale plays (Table 6-3), and no impacts to drinking water resources or another local gas-producing interval were identified. See Text Box 6-6 for more information on the Greene County site.

Text Box 6-6. Monitoring at the Greene County, Pennsylvania, Hydraulic Fracturing Test Site.

Monitoring performed at the Marcellus Shale test site in Greene County, Pennsylvania, evaluated fracture height growth and zonal isolation during and after hydraulic fracturing operations (Hammack et al., 2014). The site has six horizontally drilled wells and two vertical wells that were completed into the Marcellus Shale. Pre-fracturing studies of the site included a 3D seismic survey to identify faults, pressure measurements, and baseline sampling for isotopes; drilling logs were also run. Hydraulic fracturing occurred April 24 to May 6, 2012, and June 4 to 11, 2012. Monitoring at the site included the following:

- **Microseismic monitoring** was conducted during four of the six hydraulic fracturing jobs on the site, using geophones placed in the two vertical Marcellus Shale wells. These data were used to monitor fracture height growth above the six horizontal Marcellus Shale wells during hydraulic fracturing.
- **Pressure and production data** were collected from a set of existing vertical gas wells completed in Upper Devonian/Lower Mississippian zones 3,800 to 6,100 ft (1,200 to 1,900 m) above the Marcellus. Data were collected during and after the hydraulic fracturing jobs and used to identify any communication between the fractured areas and the Upper Devonian/Lower Mississippian rocks.
- **Chemical and isotopic analyses** were conducted on gas and water produced from the Upper Devonian/Lower Mississippian wells. Samples were analyzed for stable isotope signatures of hydrogen, carbon, and strontium and for the presence of perfluorocarbon tracers used in 10 stages of one of the hydraulic fracturing jobs to identify possible gas or fluid migration to overlying zones (Sharma et al., 2014a; Sharma et al., 2014b).

As of September 2014, no evidence was found of gas or brine migration from the Marcellus Shale (Hammack et al., 2014), although longer-term monitoring is necessary to confirm that no impacts to overlying zones have occurred (Zhang et al., 2014a).

Similarly, in Dunn County, North Dakota, there is evidence suggestive of out-of-zone fracturing in the Bakken Shale (U.S. EPA, 2015i). At the Killdeer site (Section 6.2.2.1), hydraulic fracturing fluids and produced water were released during a rupture of the casing at the Franchuk 44-20 SWH well. Water quality characteristics at two monitoring wells located immediately downgradient of the Franchuk well reflected a mixing of local Killdeer Aquifer water with deep formation brine. Ion and isotope ratios used for brine fingerprinting suggest that Madison Group formations (which directly overlie the Bakken in the Williston Basin) were the source of the brine observed in the Killdeer Aquifer, and the authors concluded that this provides evidence for out-of-zone fracturing (U.S. EPA, 2015i). Industry experience also indicates that out-of-zone fracturing could be fairly common in the Bakken and that produced water from many Bakken wells has Madison Group chemical signatures (Arkadakskiy and Rostron, 2013; Arkadakskiy and Rostron, 2012; Peterman et al., 2012).

Fracture growth from a deep formation to a near-surface aquifer is generally considered to be limited by layered geological environments and other physical constraints (Fisher and Warpinski, 2012; Daneshy, 2009). For example, differences in in-situ stresses in layers above and below the production zone can restrict fracture height growth in sedimentary basins (Fisher and Warpinski, 2012). High-permeability layers near hydrocarbon-producing zones can reduce fracture growth by acting as a "thief zone" into which fluids can migrate, or by inducing a large compressive stress that acts on the fracture (de Pater and Dong, 2009, as cited in Fisher and Warpinski, 2012). Although thief zones may prevent fractures from reaching shallower formations or growing to extreme vertical lengths, they do allow fluids to migrate out of the production zone into receiving formations, which could (depending on site-specific conditions) potentially contain drinking water resources. A volumetric argument has also been used to discuss limits of vertical fracture growth; that is, the volumes of fluid needed to sustain fracture growth beyond a certain height would be unrealistic (Fisher and Warpinski, 2012). However, as described in Section 6.3.1, fracture volume can be greater than the volume of injected fluid due to the effects of pressurized water combined with the effects of gas during injection (Kim and Moridis, 2015). Nevertheless, some numerical investigations suggest that, unless unrealistically high pressures and injection rates are applied to an extremely weak and homogeneous formation that extends up to the near surface, hydraulic fracturing generally induces stable and finite fracture growth in a Marcellus-type environment and fractures are unlikely to extend into drinking water resources (Kim and Moridis, 2015).

Modeling studies have identified other factors that can affect the containment of fractures within the producing formation. As discussed above, additional numerical analysis of fracture propagation during hydraulic fracturing has demonstrated that contrasts in the geomechanical properties of rock formations can affect fracture height containment (Gu and Siebrits, 2008) and that geological layers present within shale gas reservoirs can limit vertical fracture propagation (Kim and Moridis, 2015). In another modeling study, Myshakin et al. (2015) applied a multi-layered geologic model to study whether fracture growth can extend upward through overlying strata and reach drinking water resources in a Marcellus Shale-type environment. Most fractures were predicted either to extend upward to the overlying layer (about 46%) or to remain in the Marcellus Shale (about 34%). About 20% of the fractures were predicted to extend further upward into or above the overlying limestone. These model results are consistent with microseismic events observed above the Tully Limestone in Greene County, Pennsylvania (Hammack et al., 2014), where the fracture heights ranged from 0 to 700 ft (0 to 200 m) and most of the fractures terminated less than 100 ft (31 m) above the top of the Marcellus.

If fractures were to propagate from the production zone to drinking water resources, other factors would need to be in place for fluid migration to occur. Using a numerical simulation, Reagan et al. (2015) investigated potential short-term migration of gas and water between a shale or tight gas formation and a shallower groundwater unit, assuming that a permeable pathway already exists between the two formations. Note that, for the purposes of this study, the pathway was assumed to be pre-existing, and Reagan et al. (2015) did not model the hydraulic fracturing process itself.

The subsurface system evaluated in the Reagan et al. (2015) modeling investigation included a horizontal well used for hydraulic fracturing and gas production, a connecting pathway between the producing formation and the aquifer, and a shallow vertical water well in the aquifer (Figure 6-7). The parameters and scenarios used in the study are shown in Table 6-4; two vertical separation distances between the producing formation and the aquifer were investigated, along with a range of production zone permeabilities and other variables used to describe four production scenarios. The horizontal well was assigned a constant bottomhole pressure of half the initial pressure of the target reservoir, not accounting for any over-pressurization from hydraulic fracturing. (As noted in Section 6.3.2.1, over-pressurization during hydraulic fracturing can create an additional driving force for upward migration.) In the simulation, migration was assessed immediately after hydraulic fracturing and for up to a 2-year simulation period representing the production stage.

Results of this modeling investigation indicate a generally downward water flow within the connecting fracture (i.e., flow from the aquifer through the connecting fracture into the hydraulically induced fractures in the production zone) with some upward migration of gas (Reagan et al., 2015). In certain simulated cases, gas breakthrough (the appearance of gas at the base of the drinking water aquifer) was also observed. The key parameter affecting migration of gas into the aquifer was the production regime, particularly whether gas production (which drives migration toward the production well) was occurring in the reservoir. Simulations that included a producing gas well showed only a few instances of breakthrough, while simulations without gas production (i.e., that assumed the well was shut-in) tended to result in breakthrough. When gas breakthrough did occur, the breakthrough times ranged from minutes to 20 days. However, in all cases, the gas escape was limited in duration and scope, because the amount of gas available for immediate migration toward the shallow aquifer was limited to that initially stored in the hydraulically induced fractures after the stimulation process and prior to production. These simulations indicate that the target reservoir may not be able to replenish the gas that was available for migration prior to production.

Based on the results of the Reagan et al. (2015) modeling study, gas production from the reservoir appears likely to mitigate gas migration, both by reducing the amount of available gas and depressurizing the induced fractures (which counters the buoyancy of any gas that may escape from the production zone into the connecting fracture). Production at the gas well also creates pressure gradients that drive a downward flow of water from the aquifer via the fracture into the producing formation, increasing the amount of water produced at the gas well. Furthermore, the effective permeability of the connecting feature is reduced during water (downward) and gas (upward) counter-flow within the fracture, further retarding the upward movement of gas or allowing gas to dissolve into the downward flow. However, Reagan et al. (2015) did find an increased potential for gas release from the producing formation in cases where there is no gas production following hydraulic fracturing. The potential for gas migration during shut-in periods following hydraulic fracturing and prior to production may be more significant, especially when out-of-zone fractures are formed. Without the effects of production, gas can rise via buoyancy, with any downward-flowing water from the aquifer displacing the upward-flowing gas.

Reagan et al. (2015) also found that the permeability of a connecting fault or fracture may be an important factor affecting the potential upward migration of gas (although not as significant as the production regime). For the cases where gas escaped from the production zone, the maximum volume of migrating gas depended upon the permeability of the connecting feature: the higher the permeability, the larger the volume. The modeling results also showed that lower permeabilities delay the downward flow of water from the aquifer, allowing the trace amount of gas that entered into the fracture early in the modeled period to reach the aquifer, which was otherwise predicted to dissolve in the water flowing downward in the feature. Similarly, the permeabilities of the target reservoir, fracture volume, and the separation distance were found to affect gas migration, because they affected the initial amount of gas stored in the hydraulically induced fractures. In contrast, the permeability of the drinking water aquifer was not found to be a significant factor in the assessment.

Table 6-4. Modeling parameters and scenarios investigated by Reagan et al. (2015).
This table illustrates the range of parameters included in the Reagan et al. (2015) modeling study. See Figure 6-7, Figure 6-8, and Figure 6-9 for conceptualized illustrations of these scenarios.

Model parameter or variable	Values investigated in model scenarios
All scenarios	
Lateral distance from connecting feature to water well	328 ft (100 m)
Vertical separation distance between producing formation and drinking water aquifer	656 ft (200 m); 2,625 ft (800 m)
Producing formation permeability range	1 nD (1 x 10^{-21} m^2); 100 nD (1 x 10^{-19} m^2); 1 µD (1 x 10^{-18} m^2)
Drinking water aquifer permeability	0.1 D (1 x 10^{-13} m^2); 1 D (1 x 10^{-12} m^2)
Initial conditions	Hydrostatic
Production well bottom hole pressure	Half of the initial pressure of the producing formation (not accounting for over-pressurization from hydraulic fracturing)
Production regime	Production at both the water well and the gas well; Production at only the water well; Production at only the gas well; No production

Model parameter or variable	Values investigated in model scenarios
Fracture pathway scenarios	
Connecting feature permeability	1 D (1 x 10^{-12} m^2); 10 D (1 x 10^{-11} m^2); 1,000 D (1 x 10^{-9} m^2)
Offset well pathway scenarios	
Lateral distance from production well to offset well	33 ft (10 m)
Cement permeability of offset well	1 µD (1 x 10^{-18} m^2); 1 mD (1 x 10^{-15} m^2); 1 D (1 x 10^{-12} m^2); 1,000 D (1 x 10^{-9} m^2)

6.3.2.3 Migration via Fractures Intersecting with Offset Wells and Other Artificial Structures

Another potential pathway for fluid migration is one in which hydraulic fracturing fluids or displaced subsurface fluids move through newly created fractures into an offset well or its fracture network, resulting in a process called well communication (Jackson et al., 2013d). The offset well can be an abandoned (i.e., plugged), inactive, or actively producing well. In addition, if the offset well has also been used for hydraulic fracturing, the fracture networks of the two wells might intersect. The situation where hydraulic fractures propagate to (and inject fluid into and/or cause pressure increases in) other existing wells or hydraulic fractures is referred to as a "frac hit" and is known to occur in areas with a high density of wells (Jackson et al., 2013a).

Frac hits can be more common in unconventional production settings compared to conventional production settings, due to the closer/denser well spacing (King and Valencia, 2016). Figure 6-8 provides a schematic to illustrate fractures that intercept an offset well, and Figure 6-9 depicts (in a simplified illustration) how the fracture networks of two such wells might intersect. This can be a particular concern in shallower formations, where the local least principal stress is vertical (resulting in more horizontal fracture propagation), and in situations where there are drinking water wells in the same formation as wells used for hydraulic fracturing.

Instances of well communication have been known to occur and are described in well records and the oil and gas literature. For example, an analysis of operator data collected by the New Mexico Oil Conservation Division (NM OCD) in 2013−2014 identified 120 instances of well communication in the San Juan Basin between 2007 and 2013 (Vaidyanathan, 2014). In some cases, well communication incidents have led to documented production and/or environmental problems. A study in the Barnett Shale noted two cases of well communication, one with a well 1,100 ft (340 m) away and the other with a well 2,500 ft (760 m) away from the initiating well; ultimately, one of the offset wells had to be re-fractured because the well communication halted production (Craig et al., 2012). In some cases, the fluids that intersect the offset well flow up the wellbore and spill onto the surface. In its report *Review of State and Industry Spill Data: Characterization of Hydraulic Fracturing-Related Spills*, the EPA (2015m) recorded 10 incidents in which fluid spills were attributed to well communication events (see Text Box 5-10 for more information on this effort).[1] The Well File Review (U.S. EPA, 2016c) reports that 1% of the wells (an estimated 280 wells) represented in the study reported a frac hit, where the hydraulic fracturing operation documented in the Well File Review led to communication with a nearby well.[2] (It was not possible to determine whether fluids reached protected groundwater resources during these frac hits based on information in the well files.) While the subsurface effects of frac hits have not been extensively studied, these cases demonstrate the possibility of fluid migration via communication with other wells and/or their fracture networks. More generally, well communication events can indicate fracture behavior that was not intended by the treatment design.

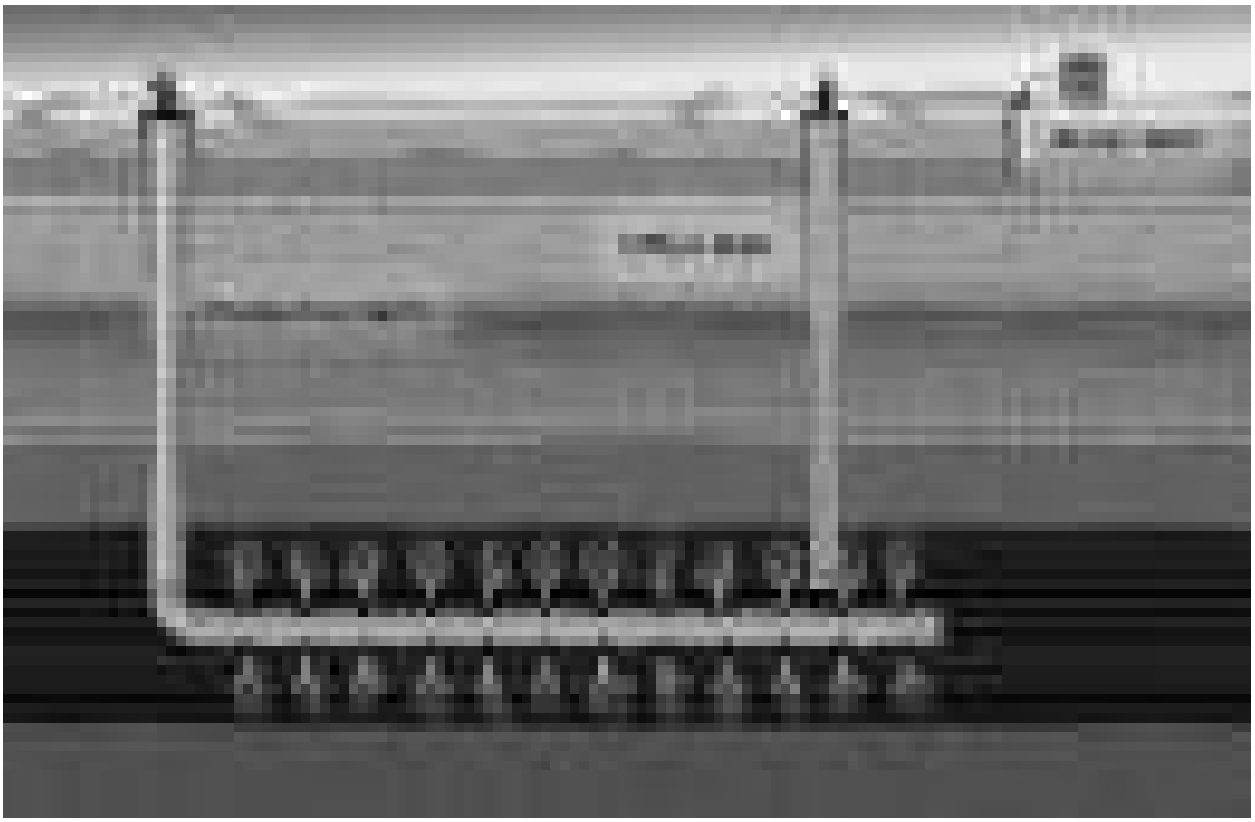

Figure 6-8. Induced fractures intersecting an offset well (in a production zone, as shown, or in overlying formations into which fracture growth may have occurred).

This image shows a conceptualized depiction of potential pathways for fluid movement out of the production zone (not to scale).

[1] These spills are represented by line numbers 163, 236, 265, 271, 286, 287, 375, 376, 377, and 380 in Appendix B of U.S. EPA (2015m).

[2] 280 wells (95% confidence interval: 240 – 320 wells).

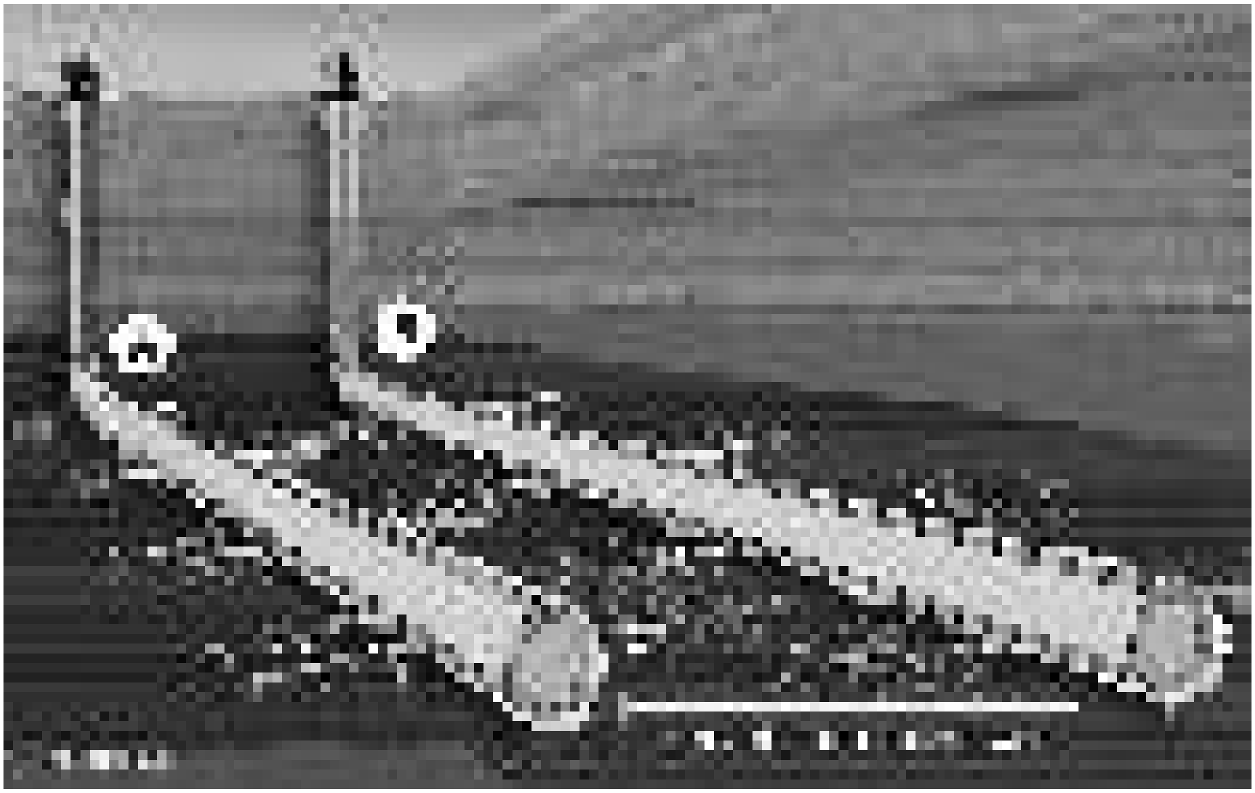

Figure 6-9. Well communication (a frac hit).
This image shows a conceptualized depiction of the fractures of a newly fractured well (Well A) intersecting the existing fracture network created during a previous hydraulic fracturing operation in an offset well (Well B). Evidence of this interaction may be observed in the offset well as a pressure change, lost production, and/or introduction of new fluids. Depending on the condition of the offset well, this can result in fluid being spilled onto the surface, rupturing of cement and/or casing and hydraulic fracturing fluid leaking into subsurface formations, and/or fluid flowing out through existing flaws in the casing and/or cement. (Figure is not to scale.)

A well communication event is usually observed at the offset well as a pressure spike, due to the elevated pressure from the originating well, or as an unexpected drop in the production rate (Lawal et al., 2014; Jackson et al., 2013a). Ajani and Kelkar (2012) performed an analysis of frac hits in the Woodford Shale in Oklahoma, studying 179 wells over a 5-year period. The authors used fracturing records from the newly completed wells and compared them to production records from surrounding wells. The authors assumed that sudden changes in production of gas or water coinciding with fracturing at a nearby well were caused by communication between the two wells, and increased water production at the surrounding wells was assumed to be caused by hydraulic fracturing fluid flowing into these offset wells. The results of the Oklahoma study showed that 24 wells had decreased gas production or increased water production within 60 days of the initial gas production at the nearby fractured well. A total of 38 wells experienced decreased gas or increased water production up to a distance of 7,920 ft (2,410 m), which the study authors defined as the distance between the midpoints of the laterals; 10 wells saw increased water production from as far away as 8,422 ft (2,567 m). In addition, one well showed a slight increase in gas production rather than a decrease.[1]

Other studies of well communication events have relied on similar information. In the NM OCD operator data set, the typical means of detecting a well communication event was through pressure changes at the offset well, production lost at the offset well, and/or fluids found in the offset well. In some instances, well operators determined that a well was producing fluid from two different formations, while in one instance, the operator identified a potential well communication event due to an increase in production from the offset well (Vaidyanathan, 2014). In another study, Jackson et al. (2013a) found that the decrease in production due to well communication events was much greater in lower permeability reservoirs. The authors note an example where two wells 1,000 ft (300 m) apart communicated, reducing production in the offset well by 64%. These results indicate that the subsurface interactions of well networks or complex hydraulics driven by each well at a densely populated (with respect to wells) area are important factors to consider for the design of hydraulic fracturing treatments and other aspects of oil and gas production.

The key factor affecting the likelihood of a well communication event and the impact of a frac hit is the location of the offset well relative to the well where hydraulic fracturing was conducted (Ajani and Kelkar, 2012). In the Ajani and Kelkar (2012) analysis, the likelihood of a communication event was less than 10% in wells more than 4,000 ft (1,000 m) apart, but rose to nearly 50% in wells less than 1,000 ft (300 m) apart. Well communication was also much more likely with wells drilled from the same pad. The affected wells were found to be in the direction of maximum horizontal stress in the field, which correlates with the expected direction of fracture propagation. Modeling work by Myshakin et al. (2015) is generally consistent with these results, indicating that the risk of fluid movement through pre-existing wellbores or open faults is negligible unless hydraulic fractures are located very close to these features.[2]

Statistical modeling by Montague and Pinder (2015) investigated the probability that a hypothetical new well used for hydraulic fracturing within the area of New York underlain by the Marcellus Shale would intersect an existing wellbore. The results indicated that this probability would be from 0 to 3.45%. The model incorporated the depth of the hypothetical new well, the vertical growth of induced fractures, and the depth and locations of existing nearby wells. The model also assumed that the existing wells are vertical and fracture growth is not impacted by nearby wells or existing fractures. However, the authors concluded that the inclusion of horizontal wells within the data set could increase the chance of intersection with induced fractures.

Well communication may be more likely to occur where there is less resistance to fracture growth. Such conditions may be related to existing production operations (e.g., where previous hydrocarbon extraction has reduced the pore pressure, changed stress fields, or affected existing fracture networks) or the existence of high-permeability rock units (Jackson et al., 2013a). As Ajani and Kelkar (2012) found in the Woodford Shale, one of the deepest major shale plays (Table 6-3), induced fractures tend to enter portions of the reservoir that have already been fractured as

[1] The numbers of wells cited in the study reflect separate analyses, and the numbers cited are not additive.

[2] In the Myshakin et al. (2015) paper, the authors do not quantify or explain what is meant by "very close."

opposed to entering previously unfractured rocks, ultimately causing interference in offset wells. Mukherjee et al. (2000) described this tendency for asymmetric fracture growth toward depleted areas in low-permeability gas reservoirs due to pore pressure depletion from production at offset wells. The authors note that pore pressure gradients in depleted zones would affect the subsurface stresses. Therefore, depending on the location of the new well with respect to depleted zone(s) and the orientation of the existing induced fractures, the newly created fracture can be asymmetric, with only one wing of the fracture extending into the depleted area and developing significant length and conductivity (Mukherjee et al., 2000). The extent to which the depleted area affects fracturing depends on factors such as cumulative production, pore volume, hydrocarbon saturation, effective permeability, and the original reservoir or pore pressure (Mukherjee et al., 2000). Similarly, high-permeability rock types acting as thief zones may also cause preferential fracturing due to a higher leakoff rate into these layers (Jackson et al., 2013a).

In addition to location, the potential for impact on a drinking water resource also depends on the condition of the offset well. (See Section 6.2 for information on the mechanical integrity of well components.) In their analysis, Ajani and Kelkar (2012) found a correlation between well communication and well age: older wells were more likely to be affected. If the cement in the annulus between the casing and the formation is intact and the well components can withstand the stress exerted by the pressure of the fluid, nothing more than an increase in pressure and extra production of fluids would occur during a well communication event. However, if the offset well is not able to withstand the pressure of the hydraulic fracturing fluid, well components could fail (Figure 6-4), allowing fluid to migrate out of the well.

The highest pressures most hydraulic fracturing wells will face during their life spans occur during the process of fracturing (Section 3.3). In some cases, temporary equipment is installed in wells during fracturing to protect the well against the increased pressure. Therefore, many producing wells may not be designed to withstand pressures typical of hydraulic fracturing (Enform, 2013) and can experience problems when fracturing occurs in nearby wells. Depending on the location of the weakest point in the offset well, this could result in fluid being spilled onto the surface; rupturing of cement and/or casing and hydraulic fracturing fluid leaking into subsurface formations; and/or fluid flowing out through existing flaws in the casing and/or cement. (See Chapters 5 and 7 for additional information on how such spills can affect drinking water resources.) For example, a documented well communication event near Innisfail, Alberta, Canada (Text Box 6-7) occurred when several well components failed, because they were not rated to handle the increased pressure caused by the well communication (ERCB, 2012). In addition, if the fractures were to intersect an uncemented portion of the wellbore, the fluids could potentially migrate into formations that are uncemented along the wellbore.

In older wells near a hydraulic fracturing operation, plugs and cement can degrade over time; in some cases, abandoned wells may never have been plugged properly. Before the 1950s, most well plugging efforts were focused on preventing water from the surface from entering oil fields. As a result, many wells from that period were abandoned with little or no cement (NPC, 2011b). This can be a significant issue in areas with legacy (i.e., historic) oil and gas exploration and when wells are re-entered and hydraulically fractured (or re-fractured) to increase production in a reservoir. In one study, 18 of 29 plugged and abandoned wells in Quebec were found to show signs of leakage (Council of Canadian Academies, 2014). Similarly, a PA DEP report cited three cases where migration of natural gas had been caused by well communication events with old, abandoned wells, including one case where private drinking water wells were affected (PA DEP, 2009c). In Tioga County, Pennsylvania, following hydraulic fracturing of a shale gas well, an abandoned well nearby produced a 30 ft (9 m) geyser of brine and gas for more than a week (Dilmore et al., 2015).

Text Box 6-7. Well Communication at a Horizontal Well near Innisfail, Alberta, Canada.

In most cases, well communication during fracturing results in a pressure surge accompanied by a drop in gas production and additional flow of produced water or hydraulic fracturing fluid at an offset well. However, if the offset well is not capable of withstanding the high pressures of fracturing, more significant damage can occur.

In January 2012, fracturing at a horizontal well near Innisfail in Alberta, Canada, caused a surface spill of fracturing and formation fluids at a nearby operating vertical oil well. According to the investigation report by the Alberta Energy Resources Conservation Board (ERCB, 2012), pressure began rising at the vertical well less than two hours after fracturing ended at the horizontal well.

Several components of the vertical well facility—including surface piping, discharge hoses, fuel gas lines, and the pressure relief valve associated with compression at the well—were not rated to handle the increased pressure and failed. Ultimately, the spill released, in addition to gas, an estimated 19,816 gal (75,012 L^3) of hydraulic fracturing fluid, brine, and oil covering an area of approximately 656 ft by 738 ft (200 m by 225 m).

The ERCB determined that the lateral of the horizontal well passed within 423 ft (129 m) of the vertical well at a depth of approximately 6,070 ft (1,850 m) below the surface in the same formation. The operating company had estimated a fracture half-length of 262 to 295 ft (80 to 90 m) based on a general fracture model for the field.[1] While there were no regulatory requirements for spacing hydraulic fracturing operations in place at the time, the 423 ft (129 m) distance was out of compliance with the company's internal policy to space fractures from adjacent wells at least 1.5 times the predicted half-length. The company also did not notify the operators of the vertical well of the hydraulic fracturing operations. The incident prompted the ERCB to issue *Bulletin 2012-02—Hydraulic Fracturing: Interwellbore Communication between Energy Wells*, which outlines expectations for avoiding well communication events and preventing adverse effects on offset wells.

Various studies estimate the number of abandoned wells in the United States to be significant. The Interstate Oil and Gas Compact Commission (IOGCC, 2008) estimates that over one million wells were drilled in the United States prior to the enactment of state oil and gas regulations, and the status and location of many of these wells are unknown. A recent estimate of wells completed before the adoption of statewide well abandonment criteria in 1957 in Pennsylvania placed the range at 305,000 to 390,000 wells in the state, with more than 176,000 of those wells likely abandoned pre-1957 (Dilmore et al., 2015). As of 2000, PA DEP's well plugging program reported that it had documented 44,700 wells that had been plugged and 8,000 that were in need of plugging, and approximately 184,000 additional wells with an unknown location and status (PA

[1] The fracture half-length is the radial distance from a wellbore to the outer tip of a fracture propagated from that well (Schlumberger, 2014).

DEP, 2000). A similar evaluation from New York State found that the number of unplugged wells was growing in the state despite an active well plugging program (Bishop, 2013). In the Midwest, Sminchak et al. (2014) examined two areas of historical oil and gas exploration as part of an investigation of potential carbon dioxide sequestration sites. They found that a 4.3 mi by 4.3 mi (6.9 km by 6.9 km) square area in Michigan contained 22 abandoned oil and gas wells and a 9.3 mi by 9.3 mi (15.0 km by 15.0 km) square area in Ohio contained 359 abandoned oil and gas wells.

Various state programs exist to plug identified orphaned wells, but they face the challenge of identifying and addressing a large number of wells.[1] In some cases, remote sensing technologies can be used to identify wells for which no records exist. For example, an NETL study in Pennsylvania found that helicopter-based high-resolution magnetic surveys can be used to accurately locate wells with steel casing; wells with no steel casing exhibit weak or no magnetic anomaly and are not detected by such surveys (Veloski et al., 2015). Chapter 10 includes a discussion of factors and practices, including those related to active and abandoned wells near hydraulic fracturing operations, that can reduce the frequency of impacts to drinking water quality.

The Reagan et al. (2015) numerical modeling study included an assessment of migration via an offset well as part of its investigation of potential fluid migration from a producing formation into a shallower groundwater unit (Section 6.3.2.2). For the offset well pathway, it was assumed that the hydraulically induced fractures intercepted an older offset well with deteriorated components. (This assessment can also be applicable to cases where potential migration may occur via the production well-related pathways discussed in Section 6.2) The highest permeability value tested for the connecting feature represented a case with an open wellbore. A key assumption for this investigation was that the offset well was already directly connected to a permeable feature in the reservoir or within the overburden.

Similar to the cases for permeable faults or fractures discussed in Section 6.3.2.2, the study investigated the effect of multiple well- and formation-related variables on potential fluid migration (Table 6-4). Based on the simulation results, an offset well pathway can have a greater potential for gas release from the production zone into a shallower groundwater unit than the fracture pathway discussed in Section 6.3.2.2 (Reagan et al., 2015). This difference is primarily due to the total pore volume of the connecting pathway within the offset well; if the offset well pathway has a significantly lower pore volume compared to the fracture pathway, this would reduce possible gas storage in the connecting feature and increase the speed of buoyancy-dependent migration. However, as with the fracture scenario, the gas available for migration in this case is still limited to the gas that is initially stored in the hydraulically induced fractures. Accordingly, any incidents of gas breakthrough in the model results were limited in both duration and magnitude.

In their modeling study, Reagan et al. (2015) found that production at the gas well (the well used for hydraulic fracturing) also affects the potential upward migration of gas and its arrival times at the drinking water formation due to its effect on the driving forces (e.g., pressure gradient). Similar to the fracture cases described in Section 6.3.2.2, production in the target reservoir appears to mitigate upward gas migration, both by reducing the amount of gas that might otherwise be

[1] An orphaned well is an inactive oil or gas well with no known (or financially solvent) owner.

available for upward migration and by creating a pressure gradient toward the production well. Only scenarios without the mitigating feature of gas production result in upward migration into the aquifer. This assessment also found a generally downward water flow within the connecting well pathway, which is more pronounced when the production well is operating and there is de-pressurization within the fractures. The producing formation and aquifer permeabilities appear not to be significant factors for upward gas migration via this pathway. Instead, Reagan et al. (2015) found the permeability of the connecting well to be the key factor affecting the migration of gas to the aquifer and the water well. Very low permeabilities (less than 1 mD, or 1×10^{-15} m^2) for the connecting well lead to no migration of gas into the aquifer regardless of the vertical separation distance, whereas larger permeabilities presented a greater potential for gas breakthrough.

Brownlow et al. (2016) also modeled communication with an abandoned well. The modeling exercise was based on operator data from the Eagle Ford Shale. Two types of cases were modeled: cases with an open (unplugged) abandoned well (which the authors note are known to occur in Texas) and cases with an abandoned well that was converted into a water well after the lower portion of the well had been filled with drilling mud (a practice allowed in Texas until 1967). The modeling results indicated that fluid could potentially migrate up both types of abandoned wells, with relatively greater flow rates in open abandoned wells and in abandoned wells closer to the well used for hydraulic fracturing. Similar to the Reagan et al. (2015) study, the production regime was also a key factor; when production and flowback were included in the simulation, they were found to inhibit upward migration. Modeled flow rates through the mud-filled well were comparable to those found by Reagan et al. (2015) with higher flows predicted through the open well.

A similar study was conducted by Nowamooz et al. (2015), who modeled a hypothetical well in the Utica Shale in Quebec. They assumed a 7.9 in (200 mm) wellbore with an approximately 2 in (51 mm) annulus space filled with intact cement. The researchers varied the permeability of the cement from 1 μD (1×10^{-19} m^2) to 1 mD (1×10^{-15} m^2). The results indicated that, at the highest permeability of 1 mD, a flow of methane of 1.02×10^{-2} ft^3/day (2.9×10^{-4} m^3/d) was possible. This was two orders of magnitude higher than the flow when the cement permeability was 1 μD (1×10^{-19} m^2). The wellbore permeabilities used by Nowamooz et al. (2015) appear to be consistent with actual permeabilities observed in the field, which can vary widely. For example, a study of 31 abandoned oil and gas wells in Pennsylvania found effective permeability values along the wellbores in the range of 10^{-6} to 10^2 mD (1×10^{-21} to 1×10^{-13} m^2) (Kang et al., 2015).

In the same way that fractures can propagate to intersect offset wells, they can also potentially intersect other artificial subsurface structures including mine shafts or solution mining sites. No known incidents of this type of migration have been documented. However, the Bureau of Land Management (BLM) has identified over 48,000 abandoned mines in the United States and is adding new mines to its inventory every year (BLM, 2015). In addition, the Well File Review identified an estimated 800 cases where wells used for hydraulic fracturing were drilled through mining voids, and an additional 90 cases of drilling through gas storage zones or wastewater disposal zones (U.S.

EPA, 2015n).[1,2] The analysis suggests emplacing cement within such zones can be challenging, which, in turn, could lead to a loss of zonal isolation (as described in Section 6.2) and create a pathway for fluid migration.

6.3.2.4 Migration via Fractures Intersecting Geologic Features

Potential fluid migration via natural, permeable fault or fracture zones in conjunction with hydraulic fracturing has been recognized as a potential contamination hazard for several decades (Harrison, 1983). Natural fracture systems have a strong influence on the success of a fracture treatment, and the topic has been studied extensively from the perspective of optimizing treatment design (e.g., Dahi Taleghani and Olson, 2011; Weng et al., 2011; Vulgamore et al., 2007). While porous flow in unfractured shale or tight sand formations is assumed to be negligible due to very low formation permeabilities (as discussed in Section 6.3.2.1), the presence of small natural fractures known as "microfractures" within tight sand or shale formations is widely recognized, and these fractures affect fluid flow and production strategies. Naturally occurring permeable faults and larger-scale fractures within or between formations can potentially allow for more significant flow pathways out of the production zone (Jackson et al., 2013d). Figure 6-7 illustrates the concept of induced fractures intersecting with permeable faults or fractures extending out of the target reservoir.[3]

The specific effects of natural fractures on fluid migration, and the mechanisms by which these effects occur, are not completely understood. While naturally occurring microfractures can impact the growth of induced fractures (e.g., by affecting the tensile strength of a shale layer), studies based on modeling and monitoring data generally do not indicate that they contribute to fracture growth in a way that could affect the frequency or severity of impacts. Microfractures could affect fluid flow patterns near the induced fractures by increasing the effective contact area. Conversely, these microfractures could act as capillary traps for the hydraulic fracturing fluid during treatment (contributing to fluid leakoff) and potentially hinder hydrocarbon flow due to lower gas relative permeabilities (Dahi Taleghani et al., 2013). Ryan et al. (2015) suggested that some natural fracture processes/patterns (such as the presence of two subvertical fracture sets) can contribute to upward gas migration, while others (such as small fracture sets with low connectivity that are confined to individual geologic layers) can preclude it.

In some areas, larger-scale geologic features may affect potential fluid flow pathways. As discussed in Text Box 6-3, baseline measurements taken before shale gas development show evidence of thermogenic methane in some shallow aquifers, suggesting that, in some cases, natural subsurface pathways exist and might allow for naturally occurring migration of gas over geologic time (Robertson et al., 2012). There is also evidence demonstrating that gas undergoes mixing in

[1] 800 wells (95% confidence interval: 10 – 1,900 wells).

[2] 90 wells (95% confidence interval: 50 – 100 wells).

[3] Faults and fractures can exhibit a range of permeabilities. For example, permeable (also referred to as "transmissive" or "conductive") faults or fault segments have enough permeability to allow fluids to flow along or across them, while others are relatively impermeable and can serve as barriers to flow. These differences in permeability are associated with geologic conditions such as rock type, depth, and stress regime. Generally, when researchers refer to the potential for migration via natural geologic features, it is assumed that these features are sufficiently permeable to serve as a pathway.

subsurface pathways (Baldassare et al., 2014; Molofsky et al., 2013; Fountain and Jacobi, 2000). Warner et al. (2012) compared recent sampling results to data published in the 1980s and found geochemical evidence for migration of fluids through natural pathways between deep underlying formations and shallow aquifers—pathways that the authors suggest could lead to contamination from hydraulic fracturing activities. In northeastern Pennsylvania, there is evidence that brine from deep saline formations has migrated into shallow aquifers over geologic time, preferentially following certain geologic structures (Llewellyn, 2014). However, this depends on local geologic characteristics and does not appear to happen in all locations; for example, in the Monongahela River Basin in West Virginia, shallow groundwater samples did not show evidence of mixing with deep brines (Boothroyd et al., 2016). As described in Chapter 7, karst features (created by the dissolution of soluble rock) can also serve as a potential pathway of fluid movement on a faster time scale.

Monitoring data show that the presence of natural faults and fractures can affect both the height and width of induced hydraulic fractures. When faults are present, relatively larger microseismic responses are seen and larger fracture growth can occur, as described below. Rutledge and Phillips (2003) suggested that, for a hydraulic fracturing operation in East Texas, pressurizing existing fractures (rather than creating new hydraulic fractures) was the primary process that controlled enhanced permeability and fracture network conductivity at the site. Salehi and Ciezobka (2013) used microseismic data to investigate the effects of natural fractures in the Marcellus Shale and concluded that fracture treatments are more efficient in areas with clusters or “swarms” of small natural fractures, while areas without these fracture swarms require more thorough stimulation. These microseismic data show that swarms of natural fractures within a shale formation can result in a fracture network with a larger width-to-height ratio (i.e., a shorter and wider network) than would be expected in a zone with a low degree of natural fracturing.

A few studies have used monitoring data to specifically investigate the effect of natural faults and fractures on the vertical extent of induced fractures. A statistical analysis of microseismic data by Shapiro et al. (2011) found that fault rupture (movement along a fault) from hydraulic fracturing is limited by the extent of the stimulated rock volume and is unlikely to extend beyond the fracture network. However, as demonstrated by microseismic data presented by Vulgamore et al. (2007), in some settings, the fracture network—and, in this case, the possibility of fault rupture—could extend laterally for thousands of feet. In the Fisher and Warpinski (2012) data set (Section 6.3.2.2), the greatest fracture heights occurred when the hydraulic fractures intersected pre-existing faults. Similarly, Hammack et al. (2014) reported that fracture growth seen above the Marcellus Shale is consistent with the inferred extent of pre-existing faults at the Greene County, Pennsylvania, research site (Section 6.3.2.2 and Text Box 6-6). The authors suggested that clusters of microseismic events may have occurred where preexisting small faults or natural fractures were present above the Marcellus Shale. Viñal (2015) used time-lapse multi-component seismic monitoring to monitor the overburden of the Montney Shale during a hydraulic fracturing operation in Alberta, Canada. The researchers found increases in the anisotropy in the overburden, which they interpreted as fractures being propagated along natural faults out of the shale and into the overburden. At a site in Ohio, Skoumal et al. (2015) found that hydraulic fracturing induced a rupture along a pre-existing fault approximately 0.6 mi (1 km) from the hydraulic fracturing operation. Using a new monitoring method known as tomographic fracturing imaging, Lacazette and Geiser (2013) also found vertical hydraulic fracturing fluid movement from a production well into a natural fracture network for distances of up to 0.6 mi (1.0 km). However, Davies et al. (2013) questioned whether this technique actually measures hydraulic fracturing fluid movement.

Modeling studies have also investigated whether hydraulic fracturing operations are likely to reactivate faults and create a potential fluid migration pathway into shallow aquifers. Results from one study suggest that, under specific circumstances, interaction with a permeable fault could result in fluid migration to the surface but only on relatively long (ca. 1,000 year) time scales (Gassiat et al., 2013). These findings have been disputed in the literature due to certain suggested limitations of the study, including the model setup, assumptions, and calibration; unrealistic fault representation; lack of constraints on fluid overpressure; and exclusion of the capillary imbibition effect (Birdsell et al., 2015b; Flewelling and Sharma, 2015). In response to these critiques, the authors stated that their work was a parametric study in which the model geometry, parameter, and boundary conditions were defined based on data collected from multiple shale gas basins, and the objective of the study was not to calibrate results to a specific site (Lefebvre et al., 2015). Other researchers reject the notion that open, permeable faults coexist with hydrocarbon accumulation (Flewelling et al., 2013). However, it is unclear whether the existence of faults in low permeability reservoirs affects the accumulation of hydrocarbons because, under natural conditions, the flow of gas may be limited due to capillary tension.

Like the other pathways discussed in this section, other conditions in addition to the physical presence of a permeable fault or fracture would need to exist for fluid migration to a drinking water resource to occur. The modeling study conducted by Reagan et al. (2015) and discussed in Section 6.3.2.2 indicates that, if such a permeable feature exists, the transport of gas and fluid flow would strongly depend upon the production regime and, to a lesser degree, the features' permeability and the separation between the reservoir and the aquifer. In addition, the pressure distribution within the reservoir (e.g., over-pressurized vs. hydrostatic conditions) will affect the fluid flow through fractures/faults. As a result, the presence of multiple geologic and well-related factors can increase the potential for fluid migration into drinking water resources. For example, in the Mamm Creek area of Colorado (Section 6.2.2.4), mechanical integrity and drilling-related problems likely acted in concert with natural fracture systems to result in a gas seep into surface water and shallow groundwater (Crescent, 2011). A similar situation occurred in southeastern Bradford County, Pennsylvania (discussed in Section 6.2), where natural fractures intersected an uncemented casing annulus and allowed gas to flow from the annulus into nearby domestic wells and a stream (Llewellyn et al., 2015).

Other modeling studies investigating the potential of fluid migration related to existing faults and fractures have given mixed results. Pfunt et al. (2016) performed modeling based on conditions in the North German Basin, i.e., deep geological settings where undisturbed cap rocks are present between the fractured formation and shallow aquifers. Their modeling indicated that the hydraulic fracturing fluid did not reach the near-surface area either during hydraulic fracturing operations or in the long-term in the presence of highly permeable pathways (fault zones, fractures). Like previous modeling studies, the authors found that the injection pressure and permeability of the connecting fault are two important factors that control upward fluid migration.

Rutqvist et al. (2013) found that, while somewhat larger microseismic events are possible in the presence of faults, repeated events and a seismic slip would amount to a total rupture length of 164 ft (50 m) or less along a fault, not far enough to allow fluid migration between a deep gas reservoir (approximately 6,562 ft or 2,000 m deep) and a shallow aquifer. A follow-up study using more sophisticated three-dimensional modeling techniques also found that deep hydraulic fracturing is unlikely to create a direct flow path into a shallow aquifer, even when hydraulic fracturing fluid is injected directly into a fault (Rutqvist et al., 2015). Similarly, a modeling study that investigated potential fluid migration from hydraulic fracturing in Germany found potential vertical fluid migration up to 164 ft (50 m) in a scenario with high fault zone permeability, although the authors note this is likely an overestimate because their goal was to "assess an upper margin of the risk" associated with fluid transport (Lange et al., 2013). More generally, results from Rutqvist et al. (2013) indicate that fracturing along an initially impermeable fault (as is expected in a shale gas formation) would result in numerous small microseismic events that act to prevent larger events from occurring (and, therefore, prevent the creation of more extensive potential pathways).

Schwartz (2015) modeled methane flow through a hypothetical permeable fault at a well in Germany. Methane flow was modeled through a permeable leakage zone that was 0.1 ft by 13 ft (0.03 m by 4 m) with an assumed permeability in the range of approximately 100 D to of 10,000 D (1×10^{-10} m^2 to 1×10^{-8} m^2). The model indicated that methane could reach a drinking water aquifer approximately 2,953 ft (900 m) above the gas zone in about a half a day and reach a maximum flow after two days. According to the model results, methane entering the aquifer led to an increase in pH, the release of negatively charged constituents such as chromium, and the adsorption of positively charged ions such as arsenic. Decreasing the permeability of the leakage zone by a factor of 100 increased the travel time by a factor of four. In another study, Myshakin et al. (2015) modeled brine migration through a natural and induced fracture network. Their results indicated that the main pathway for vertical migration of hydraulic fracturing fluid to overlying layers is through the induced fractures, and not the natural fractures. The location of hydraulic fractures relative to each other affects the extent of brine migration into overburden layers; compared to single fractures separated by large distances, closely spaced fractures were associated with higher pressures in—and, consequently, more brine migration into—overlying layers.

6.4 Synthesis

In the injection stage of the hydraulic fracturing water cycle, operators inject hydraulic fracturing fluids into a well under pressure that is high enough to fracture the production zone. These fluids flow through the well and then out into the surrounding formation, where they create fractures in the rock, allowing hydrocarbons to flow through the fractures, to the well, and then up the production string.

The production well and the surrounding geologic features function as a system that is often designed with multiple elements that can isolate hydrocarbon-bearing zones and water-bearing zones, including groundwater resources, from each other. This physical isolation optimizes oil and gas production and can protect drinking water resources via isolation within the well (by the casing and cement) and/or through the presence of multiple layers of subsurface rock between the target formations where hydraulic fracturing occurs and drinking water aquifers.

6.4.1 Summary of Findings

In this chapter, we consider impacts to drinking water resources to occur if hydraulic fracturing fluids or other subsurface fluids affected by hydraulic fracturing enter and adversely impact the quality of groundwater resources. Potential pathways for fluid movement to drinking water resources may be linked to one or more components of the well and/or features of the subsurface geologic system. If present, these potential pathways can, in combination with the high pressures under which fluids are injected and pressure changes within the subsurface due to hydraulic fracturing, result in the subsurface movement of fluids to drinking water resources.

The potential for these pathways to exist or form has been investigated through modeling studies that simulate subsurface responses to hydraulic fracturing, and demonstrated via case studies and other monitoring efforts. In addition, the development of some of these pathways—and fluid movement along them—has been documented. It is important to note that, if multiple barriers afforded by the well design and the presence of subsurface rock formations are present, the development of a pathway within this system does not necessarily result in an impact on a drinking water resource.

6.4.1.1 Fluid Movement via the Well

A production well undergoing hydraulic fracturing is subject to higher stresses during the relatively brief hydraulic fracturing phase than during any other period of activity in the life of the well. If the well cannot withstand the stresses experienced during hydraulic fracturing operations, pathways associated with the casing and cement can form that can result in the unintended movement of fluids into the surrounding environment (Section 6.2).

Multiple barriers within the well, including casing, cement, and a completion assembly can, if present, isolate hydrocarbon-bearing formations from drinking water resources located at a different depth. However, inadequate construction, defects in or degradation of the casing or cement, and/or the absence of redundancies such as multiple layers of casing and proper emplacement of cement can allow fluid movement into drinking water resources. Various studies of wells in the Marcellus Shale showed failure rates between 3 and 10%, depending on the type of failure studied (contamination of drinking water resources may or may not have occurred at these wells). The EPA's Well File Review found that 3% of all hydraulic fracturing jobs involved a downhole mechanical integrity failure, which generally resulted in hydraulic fracturing fluid entering the annular space between the casing and formation or between two casing strings.

Ensuring proper well design and mechanical integrity—particularly proper cement placement and quality—are important actions for preventing unintended fluid migration along the wellbore. While not all of the mechanical integrity failures described above resulted in fluid movement to—or contamination of—a drinking water resource, aspects of well design that lead to increased failure rates have the potential to increase the frequency or severity of impacts to drinking water quality associated with hydraulic fracturing operations.

6.4.1.2 Fluid Movement within Subsurface Geologic Formations

Potential subsurface pathways for fluid migration to drinking water resources include flow of fluids out of the production zone into formations above or below it, fractures extending out of the production zone or into other induced fracture networks, intersections of fractures with abandoned or active wells, and hydraulically induced fractures intersecting with faults or natural fractures (Section 6.3).

Vertical separation between the zone where hydraulic fracturing operations occur and drinking water resources reduces the potential for fluid migration to impact the quality of drinking water resources. However, not all hydraulic fracturing operations are characterized by large vertical distances between the production zone and drinking water resources. In coalbed methane plays, which are typically shallower than shale gas plays, these separation distances can be smaller than in other types of formations. Also, in certain areas, hydraulic fracturing is known to take place in formations containing water that meets the salinity threshold that is used in some definitions of a drinking water resource.

Lateral separation between wells undergoing hydraulic fracturing and other wells (including active and abandoned wells) also reduces the potential for fluid migration to impact drinking water resources. While some operators design fracturing treatments to communicate with the fractures of another well and optimize oil and gas production, unintended communication between two wells or their fracture systems can lead to spills in an offset well, which is an indicator of hydraulic fracturing treatments extending beyond their planned design. These well communication incidents, or “frac hits,” have been reported in New Mexico, Oklahoma, and a few other locations. Surface spills from well communication incidents have also been documented. Based on the available information, frac hits most commonly occur on multi-well pads and when wells are spaced less than 1,100 ft (340 m) apart, but they have been observed at wells up to 8,422 ft (2,567 m) away from a well undergoing hydraulic fracturing.

6.4.1.3 Impacts to Drinking Water Resources

We identified some example cases in the literature where the pathways associated with hydraulic fracturing resulted in an impact on the quality of drinking water resources.

One of these cases took place in Bainbridge Township, Ohio, in 2007. Failure to cement over-pressured formations through which a production well passed—and proceeding with the hydraulic fracturing operation without adequate cement and an extended period during which the well was shut in—led to a buildup of natural gas within the well annulus and high pressures within the well. This ultimately resulted in movement of gas from the production zone into local drinking water aquifers (Section 6.2.2.2). Twenty-six domestic drinking water wells were taken off-line and the houses were connected to a public water system after the incident due to elevated methane levels.

Casings at a production well near Killdeer, North Dakota, ruptured in 2010 following a pressure spike during hydraulic fracturing, allowing fluids to escape to the surface. Brine and tert-butyl alcohol were detected in two nearby monitoring wells. Following an analysis of potential sources, the only source consistent with the conditions observed in the two impacted water wells was the well that ruptured during hydraulic fracturing. There is also evidence that out-of-zone fracturing occurred at the well (Sections 6.2.2.1 and 6.3.2.2).

There are other cases where contamination of or changes to the quality of drinking water resources near hydraulic fracturing operations were identified. Hydraulic fracturing remains a potential contributing cause in these cases. For example:

- Migration of stray gas into drinking water resources involves many potential routes, including poorly constructed casing and naturally existing or induced fractures in subsurface formations. Multiple pathways for fluid movement may have worked in concert in northeastern Pennsylvania (possibly due to cement issues or sustained casing pressure), the Raton Basin in Colorado (where fluid migration may have occurred along natural rock features or faulty well seals), and the Wattenberg field in Colorado (where the surface casing depth and the presence of uncemented gas zones are major factors in determining the likelihood of mechanical integrity failures and contamination). While the sources of methane identified in drinking water wells in each study area could be determined with varying degrees of certainty, attempts to definitively identify the pathways of migration have generally been inconclusive (Text Box 6-3).
- At the East Mamm Creek drilling area in Colorado, inadequate placement of cement allowed the migration of methane through natural faults and fractures in the area. This case illustrates how construction issues, sustained casing pressure, and the presence of natural faults and fractures, in conjunction with elevated pressures associated with hydraulic fracturing, can work together to create a pathway for fluids to migrate toward drinking water resources (Sections 6.2.2.2 and 6.3.2.4).

Additionally, there are places in the subsurface where oil and gas resources and drinking water resources co-exist in the same formation. Evidence we examined indicates that some hydraulic fracturing for oil and gas occurs within formations where the groundwater has a salinity of less than 10,000 mg/L TDS. By definition, this results in the introduction of hydraulic fracturing fluids into formations that meet both the Safe Drinking Water Act's salinity-based definition of an underground source of drinking water and the broader definition of a drinking water resource developed for this assessment. According to the data we examined, these formations are generally in the western United States, e.g., near Pavillion, Wyoming. Hydraulic fracturing in a drinking water resource may be of concern in the short-term (where people are currently using these zones as a drinking water supply) or the long-term (if drought or other conditions necessitate the future use of these zones for drinking water).

There are other cases in which production wells associated with hydraulic fracturing are alleged to have caused contamination of drinking water resources. Data limitations in most of those cases (including the unavailability of information in litigation settlements resulting in sealed documents) make it difficult to assess whether or not hydraulic fracturing was a cause of the contamination.

6.4.2 Factors Affecting Frequency or Severity of Impacts

The multiple barriers within the hydraulic fracturing well and the presence of subsurface low-permeability geologic formations between the production zone and drinking water resources isolate fluids from drinking water resources. Because of this, any factors that affect the integrity of the system comprised of the well and the surrounding geology have the potential to affect the frequency or severity of impacts on drinking water quality. The primary factors that can affect the frequency or severity of impacts are: (1) the construction and condition of the well that is being hydraulically fractured, (2) the amount of vertical separation between the production zone and formations that contain drinking water resources, and (3) the location, depth, and condition of nearby wells or natural faults or fractures.

The presence and condition of the well's casing and cement are key factors that affect the frequency or severity of impacts to drinking water resources. Even in wells where there is substantial vertical separation (e.g., thousands of feet), defects in the well can, in theory, allow fluid movement over significant vertical distance. For example, fully cemented surface casing that extends through the base of drinking water resources is a key protective component of the well. Risk evaluation studies of a limited number of injection wells show that, if the surface casing is not set deeper than the bottom of the drinking water resource, the risk of aquifer contamination increases a thousand-fold. A review of wells that were hydraulically fractured in the Wattenberg field in Colorado showed that wells with fewer casing and cementing barriers across gas-bearing zones exhibited higher rates of failures. Most, but not all, wells used in hydraulic fracturing operations have fully cemented surface casing.

The absence of or defects in casing or cement can be the result of inadequate design or construction, including fewer layers of protective casing or when cement is incomplete (i.e., not present across all oil-gas- or water-bearing formations), of inadequate quality, or improperly emplaced. Wells that were constructed pursuant to older, less stringent requirements have a greater likelihood of exhibiting mechanical integrity problems associated with inadequate design and/or construction.

Deviated and horizontal wells may exhibit more casing and cement problems compared to vertical wells. Some (but not all) studies have shown that sustained casing pressure—a buildup of pressure within the well annulus that can indicate the presence of leaks—occurs more frequently in deviated and horizontal wells compared to vertical wells. Cement integrity problems can arise as a result of challenges in centering the casing and placing the cement in these wells. Absent efforts to ensure the emplacement of sufficient cement that is of adequate integrity, the increased use of these wells in hydraulic fracturing operations has the potential to increase the frequency at which associated cementing problems occur. This, in turn, has the potential to increase the frequency of impacts to the quality of drinking water resources.

Even in optimally designed wells, degradation of the casing and cement as they age or due to the cumulative effects of formation or operational stresses exerted on the well over time (e.g., cyclic stresses in multi-stage fractures) can impact the mechanical integrity of the well and affect the frequency of impacts to drinking water quality. Older wells exhibit more mechanical integrity problems compared to newer wells when hydraulically fractured or re-fractured. If mechanical integrity issues exist but are not detected and subsequently addressed, hydraulic fracturing fluids or other fluids can move into drinking water resources and the concentrations of contaminants in those drinking water resources—and therefore the severity of the impact—can increase.

In areas where there is little or no vertical separation between the production zone and drinking water resources, there is a greater potential to increase the frequency or severity of impacts to drinking water quality. For example, when the vertical separation is relatively small and other subsurface pathways (e.g., artificial penetrations) are present, the potential for these pathways to provide a more direct link between the production zone and a drinking water resource is greater than if there is a large separation. As described above, there are places where hydraulic fracturing operations occur in formations meeting the salinity threshold that is used in some definitions of a drinking water resource. The practice of injecting hydraulic fracturing fluids into a formation that also contains a drinking water resource can affect the quality of that water, because it is likely some of that fluid remains in the formation following hydraulic fracturing. The properties (e.g., chemical composition, toxicity, etc.) of hydraulic fracturing fluids or naturally occurring fluids that migrate to drinking water resources can affect the severity of the impact on the quality of those resources (see Chapter 9 for more information on the chemicals used in hydraulic fracturing fluids).

Where the separation between the production zone and drinking water resources is small, and where natural or induced fractures that transect the layers between these formations are present, there is a potential for increased frequency of impacts to drinking water quality via induced or natural fractures or faults. (Impacts via well-related pathways can also be a concern in these situations, as described above.)

Research shows that fractures created during hydraulic fracturing can extend out of the production zone, and that the vertical component of fracture growth is generally greater in deeper formations than shallow formations. Out-of-zone fracturing could be a concern in deeper formations if there is little vertical separation between the production zone and a deep drinking water resource and fractures propagate to unintended vertical heights. If out-of-zone fracturing is not detected (e.g., via monitoring) and subsequently addressed, the impacts to the quality of drinking water resources associated with fluid movement via these induced fractures have the potential to become more severe.

Regardless of the extent of the vertical separation between the production zone and drinking water resources, the presence of active or abandoned wells near hydraulic fracturing operations can increase the potential for hydraulic fracturing fluids to move to drinking water resources. For example, a deficiency in the construction of a nearby well (or degradation of the well's components), can provide a pathway for movement of hydraulic fracturing fluids, methane, or brines that might affect drinking water quality. If the fractures intersect an uncemented portion of a nearby wellbore, the fluids can potentially migrate along that wellbore into any formations where the well is not cemented.

The frequency of impacts to the quality of drinking water resources may increase where wells are densely spaced (particularly in shallow hydraulic fracturing operations where more fracture propagation is expected to be in the horizontal direction). The frequency of impacts may also be higher in mature oil and gas fields that pre-date the use of construction/plugging methods that can withstand the stresses associated with hydraulic fracturing operations. In these mature fields, wells tend to be older so degradation is a concern, and the location or condition of abandoned wells may not be documented. Based on the information presented in this chapter, the increased use of hydraulic fracturing in horizontal wells and in multiple wells on a single pad can increase the likelihood that these pathways could develop. This, in turn, could increase the frequency at which impacts on drinking water quality occur.

See Chapter 10 for a discussion of factors and practices that can reduce the frequency or severity of impacts to drinking water quality.

6.4.3 Uncertainties

Generally, less is known about the occurrence of (or potential for) impacts of injection-related pathways in the subsurface than for other components of the hydraulic fracturing water cycle, which tend to be easier to observe and measure. Furthermore, while there is a large amount of information available on production wells in general, there is little information that is both specific to hydraulic fracturing operations and readily accessible across the states to form a national picture.

6.4.3.1 Limited Availability of Information Specific to Hydraulic Fracturing Operations

There is extensive information available on the design goals for hydraulically fractured oil and gas wells (i.e., to address the stresses imposed by high-pressure, high-volume injection), including from industry-developed best practices documents. Additionally, many studies have documented how production wells have historically been constructed, how they perform, and the rates at which they experience problems that can lead to pathways for fluid movement. However, because of possible differences in well construction and operational practices, it is unknown how historical well performance studies apply to wells used in hydraulic fracturing operations.

Because wells that have been hydraulically fractured must withstand many of the same downhole stresses as other production wells, we consider studies of the pathways for impacts to drinking water quality in production wells to be relevant to identifying the potential pathways relevant to hydraulic fracturing operations. However, without specific data on the as-built construction of wells used in hydraulic fracturing operations, we cannot definitively state whether these wells are consistently constructed to withstand the stresses they may encounter.

There is also, in general, very limited information available on the monitoring and performance of wells used in hydraulic fracturing operations. Published information is sparse regarding mechanical integrity tests (MITs) performed during and after hydraulic fracturing, the frequency at which mechanical integrity issues arise in wells used for hydraulic fracturing, and the degree and speed with which identified issues are addressed. There is also little information available regarding MIT results for the original hydraulic fracturing event in wells built for that purpose, for wells that are later re-fractured, or for existing, older wells not initially constructed for hydraulic fracturing but repurposed for that use.

These limitations on hydraulic fracturing-specific information make it difficult to provide definitive estimates of the rate at which wells used in hydraulic fracturing operations experience the types of mechanical integrity problems that can contribute to the movement of hydraulic fracturing fluids or other fluids to drinking water resources.

There is also a limited number of peer-reviewed published studies based on groundwater sampling that provide evidence to assess whether formation brines, hydraulic fracturing fluids, or gas move in unintended ways through the subsurface during and after hydraulic fracturing. Subsurface monitoring data (i.e., data that characterize the presence, migration, or transformation of fluids within subsurface formations related to hydraulic fracturing operations) are scarce relative to the tens of thousands of oil and gas wells that are estimated to be hydraulically fractured across the country each year (see Chapter 3 for more information on the occurrence of hydraulic fracturing in the United States).

Information on fluid movement within the subsurface and the extent of fractures that develop during hydraulic fracturing operations is also limited. For example, limited information is available in the published literature on how flow regimes or other subsurface processes change at sites where hydraulic fracturing is conducted. Instead, much of the available research, and therefore the literature, addresses how hydraulic fracturing and other production technologies perform to optimize hydrocarbon production. In addition, much of the published data on fracture propagation are for shale formations, and no large-scale data sets on fracture growth in other unconventional formations exist or are publicly available.

These limitations on hydraulic fracturing-specific information make it difficult to provide definitive estimates of the rate at which wells used in hydraulic fracturing operations experience the types of mechanical integrity problems that can contribute to unintended fluid movement.

6.4.3.2 Limited Systematic, Accessible Data on Well Performance or Subsurface Movement

While the oil and gas industry generates a large amount of information on well performance as part of operations, most of this is proprietary, or otherwise not readily available to the public in a compiled or summary manner. Therefore, no national or readily accessible way exists to evaluate the design and performance of individual wells or wells in a region, particularly in the context of local geology or the presence of other wells and/or hydraulic fracturing operations. Many states have large amounts of operator-submitted data, but information about construction practices or the performance of individual wells is typically not in a searchable or aggregated form that would enable assessments of well performance under varying settings, conditions, or timeframes. Although it is collected in some cases, there is no collection, reporting, or publishing of baseline (pre-drilling and/or pre-fracturing) and post-fracturing monitoring data on a national basis that could indicate the presence or absence of hydraulic fracturing-related fluids in shallow zones and whether or not migration of those fluids has occurred. (See Chapter 10 for additional discussion of data limitations.) Ideally, data from groundwater monitoring are needed to complement theories and modeling on potential pathways and fluid migration.

While some of the types of impacts described above can occur quickly (i.e., on the scale of days or weeks, as with mechanical integrity problems or well communication events), other impacts (e.g., in slow-moving, deep groundwater) may be detectable only on much longer timescales. Without comprehensive collection and review of information about how hydraulic fracturing operations perform, fluid movement could occur without early detection, which could, in turn, increase the severity of any resultant impacts to drinking water quality. For example, testing the mechanical integrity of wells, monitoring the extent of the fractures that form, and conducting pre- and post-hydraulic fracturing water quality monitoring can detect fluid movement (or the potential for fluid movement) and provide opportunities to mitigate or minimize the severity of impacts associated with unforeseen events.

The limited amount of available information also hinders our ability to evaluate how frequently drinking water impacts are occurring, the probability that these impacts occur, or to what extent they are tied to specific well construction, operation, and maintenance practices. This also significantly limits our ability to evaluate the aggregate potential for hydraulic fracturing operations to affect drinking water resources or to identify the potential cause of drinking water contamination in areas where hydraulic fracturing occurs. The absence of this information greatly limits the ability to make quantitative statements about the frequency or severity of these impacts.

6.4.4 Conclusions

The production well and the surrounding geologic features function as a system that provides multiple barriers that can isolate hydrocarbon-bearing zones and water-bearing zones, including drinking water resources. Because of this, factors affecting the integrity of any of these barriers have the potential to adversely affect the quality of drinking water resources.

We have identified a number of pathways by which hydraulic fracturing fluids can reach and affect the quality of drinking water resources. These pathways include migration via inadequate casing and/or cement in the hydraulic fracturing well, fluid movement in the subsurface via fractures extending out of the target zone, or vertical fluid movement via other natural or artificial structures.

The primary factors affecting the frequency or severity of impacts to drinking water quality associated with hydraulic fracturing operations include the condition of the casing and cement of the production well and their placement relative to drinking water resources, the extent of the vertical separation between the production zone and drinking water resources, and the presence and condition of offset wells or natural faults or fractures near the hydraulic fracturing operation.

There is evidence that, in some cases highlighted in the literature, these pathways have formed and the quality of drinking water resources has been impacted. We do not know the frequency of such impacts associated with the injection stage of the hydraulic fracturing water cycle, however. This is related to the following: the subsurface environment is geologically complex, the relevant production processes cannot be directly observed, and publicly available data that can support an evaluation of the impacts of hydraulic fracturing on the quality of drinking water resources is, in general, very limited.

Chapter 7. Produced Water Handling

Abstract

Produced water is a byproduct of hydrocarbon production and flows to the surface through the production well, along with oil and gas. Operators must store and dispose of (or in some cases treat) large amounts of non-potable produced water, either on site or off site, and spills or releases of produced water have the potential to impact drinking water resources. Unlike produced water from conventional oil and gas production, produced water generated following hydraulic fracturing initially contains returned hydraulic fracturing fluids. Much of the hydraulic fracturing fluid remains below ground; the median amount of fluid returned to the surface is 30% or less. Up to several million gallons of water can be produced from each well, with production generally decreasing with time.

Produced water contains several classes of constituents: salts, metals, radioactive materials, dissolved organic compounds, and hydraulic fracturing chemicals and their transformation products (the result of reactions of these chemicals in the subsurface). The concentrations of these constituents change with time, as the initially returning hydraulic fracturing fluid blends with formation water. Typically, this means that the produced water becomes more saline with time. Produced water composition and volume vary from well to well, both among different formations and within formations. A large number of organic compounds have been identified in produced water, many of which are naturally occurring petroleum hydrocarbons; some are known hydraulic fracturing chemicals. Only a few transformation products have been identified, and they include chlorinated organics.

Spills and releases of produced water with a variety of causes have been documented at different steps in the production process. The causes include human error, equipment or container failure (for instance, pipeline, tank or storage pit leaks), accidents, and storms. Unauthorized discharges may account for some releases as well. An estimated half of the spills are less than 1,000 gal (3,800 L). A small number of much larger spills has been documented, including a spill of 2.9 million gal (11 million L). Both short- and long-term impacts to soil, groundwater, and surface from spills have occurred. For many spills, however, the impacts are unknown. The potential of spills of produced water to affect drinking water resources depends upon the release volume, duration, and composition, as well as watershed and water body characteristics.

Data are lacking to characterize the severity and frequency of impacts on a nationwide scale. Suspected local-scale impacts often require an extensive multiple lines-of-evidence investigation to determine their cause. Further, when investigations do take place, the lack of baseline water quality data can make it difficult to determine the cause and severity of the impact. In such cases, additional data are necessary to determine the full extent of the impact of releases of produced water.

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7. Produced Water Handling

7.1 Introduction

Water is a byproduct of oil and gas production. After the hydraulic fracturing of the formation is completed, the injection pressure is reduced, and a possible inactive period where the well is "shut in" is completed, water is allowed to flow back from the well to prepare for oil or gas production.[1] This return-flow water may contain chemicals injected as part of the hydraulic fracturing fluid, chemicals naturally occurring in the formation, or the products of reactions that take place in the formation. Initially this water, sometimes called flowback, is mostly hydraulic fracturing fluid, but as time goes on, water chemistry becomes more similar to water associated with the formation. For formations containing saline water (brine), the salinity of the returned water increases as time passes as the result of increased contact time between the hydraulic fracturing fluid and the formation and inclusion of an increased portion of formation water. For this assessment, and consistent with industry practice, the term produced water is used to refer to any water flowing from the oil or gas well.

Produced water is piped directly to an injection well or stored and accumulated at the surface for eventual management by injection into disposal wells, transport to wastewater treatment plants, reuse, or in some cases, placement in evaporation pits or permitted direct discharge. See Text Box ES-11 and Section 8.4 for discussion of these management practices.

Produced water spills and releases can occur due to several causes, including events associated with pipelines, transportation, blowouts, and storage. Impacts to drinking water resources can occur if this released water enters surface water bodies or reaches groundwater. Such impacts may result in the water becoming unfit for consumption, either through obvious taste and odor considerations or the constituents in the water exceeding hazard levels (Chapter 9). Once released to the environment, transport of chemical constituents depends on the characteristics of the:

- Spill (volume, duration, concentration);
- Fluid (density as influenced by salinity);
- Chemicals (volatility, sorption, solubility); and
- Site-specific environmental characteristics (surface topography and location of surface water bodies, the type of the soil and aquifer materials, layering and heterogeneity of rocks, and the presence of dissolved oxygen and other factors needed to support biodegradation, and the presence of inorganic species that affect metal transport).

This chapter provides characterization of produced water and also provides background information for the coverage of wastewater disposal and reuse in Chapter 8. Chapter 7 addresses the characteristics of produced water including per-well generation of produced water. Chapter 8 considers management of this water, now called wastewater, at an aggregate level, and thus

[1] There can be no shut-in period at all or it can last several weeks (Stepan et al., 2010).

discusses state, regional, and national estimates of treatment volumes. While Chapter 7 considers impacts from several types of unintentional releases, Chapter 8 focuses on impacts that are associated with wastewater management practices. One specific issue, leakage from pits and impoundments, is introduced in Chapter 7 as one of several avenues for accidental releases, with a more detailed exploration of the use of pits in wastewater management presented in Chapter 8.

Chapter 7 begins with a review of definitions for flowback and produced water in Section 7.1.1. Definitions are followed by a discussion in Section 7.2 of water volumes per well, first presenting data on the volume and percent of hydraulic fracturing fluid returned to the surface and then presenting data on the volume of water returned during production. These data all represent the response of individual wells. Because of the need to have aggregated volumes for estimating wastewater treatment loadings, estimates of total volumes are given in Section 8.2.

Chapter 7 continues with discussion of the chemical composition of produced water (Section 7.3). Because the composition of produced water is only known through analysis of samples, laboratory methods and their limitations are described in Section 7.3.1. Time-dependent changes in composition are discussed via three specific examples in Section 7.3.3, followed by discussion of five types of constituents: salts, metals, radioactive materials, organics, and known hydraulic fracturing additives in Section 7.3.4. The chemical and geological processes controlling the chemical composition of produced water are described in Appendix E. Spatial and temporal trends in the composition of produced water are illustrated with examples from the literature and data compiled for this report (Section 7.3.5).

The potential for impacts on drinking water resources of produced water releases and spills are described based on reported spill incidents (Section 7.4), and examples of spills from specific sources and data compilation studies are given in Section 7.4.2. The potential for impacts is described using contaminant transport principles in Section 7.6. The chapter concludes with a discussion of uncertainties and knowledge gaps, factors that influence the severity of impacts, and major findings (Section 7.7).

7.1.1 Definitions

Multiple definitions exist for the terms flowback and produced water. Appendix Section E.1 gives examples of definitions used by different organizations. These differing definitions reflect differing usage of the terms among various groups and that produced water reflects the continuously varying mixture between returning injection fluid and formation water. The majority of produced water definitions are fundamentally similar. The following definition is used in this report for produced water: any type of water that flows from the subsurface through oil and gas wells to the surface as a by-product of oil and gas production. Thus produced water can variously refer to returned hydraulic fracturing fluid, formation water alone, or a mixture of the two.

The term flowback has two major meanings. First is the process used to prepare the well for production by allowing excess liquids and proppant to return to the surface. The second use of the term is to refer to fluids predominantly containing hydraulic fracturing fluid that return to the surface. Because formation water can contact and mix with injection fluids, the distinction between returning hydraulic fracturing fluid and formation water is not clear. Definitions of flowback are operational in the sense that they include some characteristic of the oil and gas operation (i.e., fluids returning within 30 days). These reflect that during the early phases of operation, a higher concentration of chemical additives is expected and later, water is characteristic of the formation. Because we use existing literature in our review, we do not introduce a preferred definition of flowback, and describe all water flowing from the well as produced water.

7.2 Volume of Hydraulic Fracturing Flowback and Produced Water

Veil (2015) estimated that, in 2012, all types (i.e., from conventional and unconventional reservoirs) of U.S. onshore and offshore oil and gas production generated 8.90 x 10^{11} gal (3.37 x 10^{12} L) of produced water. More details and state-level estimates are given in Section 8.2. This section presents information on flowback and produced water volume over various time scales, and where possible, on a per-well and per-formation basis, because characteristics and volume of flowback and produced water vary by well, formation, and time.

The amount of produced water from a well varies and depends on several factors, including production, formation, and operational factors. Production factors include the amount of fluid injected, the type of hydrocarbon produced (gas or liquid), and the location within the formation. Formation factors include the formation pressure, the interaction between the formation and injected fluid (capillary forces), and reactions within the reservoir. Operational factors include the volume of the fractured production zone that includes the length of well segments and the height and width of the fractures. Certain types of problems also influence water production, including possible loss of mechanical integrity and subsurface communication between wells, both of which can result in an unexpected increase in water production (U.S. GAO, 2012; Byrnes, 2011; DOE, 2011a; GWPC and ALL Consulting, 2009; Reynolds and Kiker, 2003).

The processes that allow gas and liquids to flow are related to the conditions along the faces of fractures. Byrnes (2011) conceptualized fluid flow across the fracture face as being composed of three phases. The first is characterized by forced imbibition of fluid into the reservoir and occurs during and immediately following fracture stimulation.[1] Second is fluid redistribution within the reservoir rock, due to capillary forces. Estimates have shown that 50% or more of fracturing fluid could be captured within the Marcellus shale if imbibition drives water 2 to 6 in (5 to 15 cm) into the formation (Engelder, 2012; Byrnes, 2011; He, 2011). In the last phase, water flows out of the formation when the well is opened and pressure is reduced in the wellbore and fractures. The purpose of this phase is to recover as much of the injected fluid as possible (Byrnes, 2011) to allow higher oil or gas flow rates. The length of the last phase and, consequently, the amount of water removed, depends on factors such as the amount of injected fluid, the permeability and relative permeability of the reservoir, capillary pressure properties of the reservoir rock, and the pressure near the fracture faces.[2] The well can be shut in for varying time periods depending on operator scheduling, surface facility construction and connection thereto, or other reasons.

[1] The displacement of a non-wet fluid (i.e., gas) by a wet fluid (typically water). Adapted from Dake (1978).

[2] When multiple fluids (water, oil, gas) occupy portions of the pore space, the permeability to each fluid depends on the fraction of the pore space occupied by the fluid and the fluid's properties. As defined by Dake (1978), when this effective permeability is normalized by the absolute permeability, the resulting relationship is known as the relative permeability.

7.2.1 Flowback of Injected Hydraulic Fracturing Fluid

The amount of water produced by wells within the first few days following fracturing varies from formation to formation. Wells in the Mississippi Lime and Permian Basin can produce 1 million gal (3.8 million L) in the first 10 days of production. Wells in the Barnett, Eagle Ford, Granite Wash, Cleveland/Tonkawa Sand, Niobrara, Marcellus, and Utica Shales can produce 300,000 to 1 million gal (1.14 to 3.78 million L) within the first 10 days. Haynesville wells produce less, about 250,000 gal (950,000 L) (Mantell, 2013). Data show that the rate of water produced during the flowback period decreases as time passes (Ziemkiewicz et al., 2014; Hansen et al., 2013; Hayes, 2009).

It is not possible to specify precisely the amount of injected fluids that return in the flowback, because there is not a clear distinction between flowback and produced water, and the indicators (e.g., salinity and radioactivity, to name two) are not routinely monitored (GWPC and ALL Consulting, 2009). Rather, flowback estimates usually relate the amount of produced water measured at a given time after fracturing as a percentage of the total amount of injected fluid. Estimates of the fraction of injected hydraulic fracturing fluid that returns as flowback are highly variable (U.S. EPA, 2016d; Vengosh et al., 2014; Mantell, 2013; Vidic et al., 2013; Minnich, 2011; Xu et al., 2011). The maxima are less than 85% in all but one of the examples given in Table 7-1, Table 7-2, and Table 7-3, and most of the median values are less than 30%. In some cases, the amount of flowback is greater than the amount of injected hydraulic fracturing fluid, and the additional water comes from the formation (Nicot et al., 2014) or from a conductive pathway from an adjacent formation (Arkadakskiy and Rostron, 2013). See Appendix Section E.2.1 for more details.

Table 7-1. Data from one company's operations indicating approximate total water use and approximate produced water volumes within 10 days after completion of wells.
From Mantell (2013).

Formation	Approx. total average water use per well (million gal)	Produced water (flowback) within the first 10 days after completion		Produced water as a percentage of average water use per well	
		Low estimate (million gal)	High or only estimate (million gal)	Low estimate (% of total water use)	High or only estimate (% of total water use)
Gas shale plays (primarily dry gas)					
Barnett[a]	3.4	0.3	1.0	9%	29%
Marcellus[a]	4.5	0.3	1.0	7%	22%
Haynesville	5.4	--	0.25	--	5%
Liquid plays (gas, oil, condensate)					
Mississippi Lime	2.1	--	1.0	--	48%

Formation	Approx. total average water use per well (million gal)	Produced water (flowback) within the first 10 days after completion		Produced water as a percentage of average water use per well	
		Low estimate (million gal)	High or only estimate (million gal)	Low estimate (% of total water use)	High or only estimate (% of total water use)
Cleveland/ Tonkawa	2.7	0.3	1.0	11%	37%
Niobrara	3.7	0.3	1.0	8%	27%
Utica	3.8	0.3	1.0	8%	26%
Granite Wash	4.8	0.3	1.0	6%	21%
Eagle Ford	4.9	0.3	1.0	6%	20%

[a] Mantell (2011) reported produced water for the first 10 days at 500,000 to 600,000 gal for the Barnett, Fayetteville and Marcellus Shales.

Table 7-2. Additional short-, medium-, and long-term produced water estimates.

Location–formation	Produced water as percentage of injected fluid	Reference	Comment
Estimates without reference to a specific data set			
Unspecified Shale	5% – 35%	Hayes (2011)	
Marcellus Shale	10% – 25%	Minnich (2011)	Initial flowback
ND–Bakken	25%	EERC (2013)	
Estimates with reference to specific data evaluation			
Short duration			
Marcellus Shale	10%	Clark et al. (2013)	0 – 10 days
TX—Barnett	20%	Clark et al. (2013)	0 – 10 days
TX—Haynesville	5%	Clark et al. (2013)	0 – 10 days
AR—Fayetteville	10%	Clark et al. (2013)	0 – 10 days
Medium duration			
WV—Marcellus	8%	Hansen et al. (2013)	30 days
Marcellus Shale	24%	Hayes (2011, 2009)	Average from 19 wells, 90 days

Location–formation	Produced water as percentage of injected fluid	Reference	Comment
Long duration			
TX—Barnett	~100%[a]	Nicot et al. (2014)	72 months
WV—Marcellus	10% – 30%	Ziemkiewicz et al. (2014)	Up to 115 months
TX—Eagle Ford	<20%	Nicot and Scanlon (2012)	Lifetime
Unspecified duration			
PA—Marcellus	6%	Hansen et al. (2013)	

[a] Approximate median with large variability: 5th percentile of 20% and 90th percentile of 350%.

Table 7-3. Flowback water characteristics for wells in unconventional reservoirs.
Source: U.S. EPA (2016d). The formation-level data used to develop Tables 7-3 and 7-4 appear in Appendix Table E-1.

Resource type	Well type	Fracturing fluid (million gal)			Flowback (percent of fracturing fluid returned)		
		Weighted average	Range	Data points	Weighted average	Range	Data points
Shale	Horizontal	4.2	0.091–24	80,388	7%	0%–580%	7,377
	Directional	1.4	0.037–20	340	33%	1%–57%	36
	Vertical	1.1	0.015–19	5,197	96%	2%–581%	57
Tight	Horizontal	3.4	0.069–12	7,301	12%	0%–60%	75
	Directional	0.05	0.046–4	3,581	10%	0%–60%	342
	Vertical	1	0.016–4	10,852	4%	0%–60%	130

7.2.2 Produced Water Volumes

Mantell (2013, 2011) described the amount of produced water over the long term as high, moderate, or low for several formations. Wells in the Barnett Shale, Cleveland/Tonkawa Sand, Mississippi Lime, and the Permian Basin can produce more than 1,000 gal (3,800 L) of water per million cubic feet (MMCF) of gas. The most water-productive of these can be as high as 5,000 gal (19,000 L) per MMCF of gas. As a specific example, a high water producing formation in the western United States was described as producing 4,200 gal (16,000 L) per MMCF of gas for the life of the well (McElreath, 2011). The well was fractured and stimulated with about 4 million gal (15 million L) of water and returned 60,000 gal (230,000 L) per day in the first 10 days, followed by 8,400 gal (32,000 L) per day in the remainder of the first year. The Niobrara, Granite Wash, Eagle Ford, Haynesville, and Fayetteville Shales are relatively dry formations (with small amounts of naturally occurring formation water) and produce between 500 and 2,000 gal (1,900 to 7,600 L) of produced water per MMCF of gas (Mantell, 2013). The Utica and Marcellus Shales are viewed as drier still and produce less than 200 gal (760 L) per MMCF of gas.

Wells producing in various formation show high produced water volume variability, including the Barnett Shale, which was attributed by Nicot et al. (2014) to a few wells with exceptionally high water production. Some of these wells produced more than the amount of injected fracturing fluid.

Wells in conventional and unconventional reservoirs produce differing amounts of water. Individual hydraulically fractured wells producing gas from the Marcellus Shale produced more water than hydraulically fractured wells in conventional wells in Pennsylvania (Lutz et al., 2013). However, on a per-unit of gas produced basis, wells producing from the Marcellus Shale generate less water (35%), than those in the conventional formations.

The EPA (2016d) reported characteristics of long-term produced water for hydraulically fractured shale and tight formations (Table 7-4). For shale, horizontal wells produced more water (1,100 gal/day; 4,200 L/day) than vertical wells (500 gal/day; 1,900 L/day). Typically, this would be attributed to the longer length of the production zone in horizontal laterals than in vertical wells.

Table 7-4. Long-term produced water generation rates (gal/day per well) for wells in unconventional reservoirs.

Source: U.S. EPA (2016d). The formation-level data used to develop Tables 7-3 and 7-4 appear in Appendix Table E-1.

Resource type	Well type	Long-Term Produced Water Generation Rates (gal per day per well)		
		Weighted average	Range	Data points
Shale	Horizontal	1,100	0–29,000	43,893
	Directional	820	0.83–12,000	1,493
	Vertical	500	4.8–51,000	12,551
Tight	Horizontal	980	10–120,000	4,692
	Directional	390	15–8,200	10,784
	Vertical	650	0.71–2100	34,624

In an example from the Pennsylvania Marcellus Shale, the EPA determined that, for vertical wells in unconventional reservoirs, 6% of water came from drilling, 35% from flowback, and 59% from long-term produced water; for horizontal wells, the corresponding numbers were 9%, 33%, and 58% (U.S. EPA, 2016d). This result agrees with the U.S. Department of Energy (DOE, 2011a) who concluded that the characteristic small amount of produced water from the Marcellus Shale was due either to its low water saturation or low relative permeability to water (see Section 6.3.2.1). For these dry formations, low shale permeability and high capillarity cause water to imbibe into the formation, where some is retained permanently.

7.2.2.1 Time Trends

High rates of water production (flowback) typically occur in the first few months after hydraulic fracturing, followed by rates reduced by an order of magnitude (e.g., Nicot et al., 2014). In many cases half of the total produced water from a well is generated in the first year. Similarly, the EPA (2016d) reported a general rule of thumb that, for unconventional reservoirs, the volume of flowback (which occurs over a short period of time) is roughly equal to the volume of long-term produced water. These trends in produced water volumes occur within the timeline of hydraulic fracturing activities (Section 3.3), and show that the large, initial return volumes of flowback last for several weeks, whereas the lower-rate produced water phase can last for years (Figure 7-1).

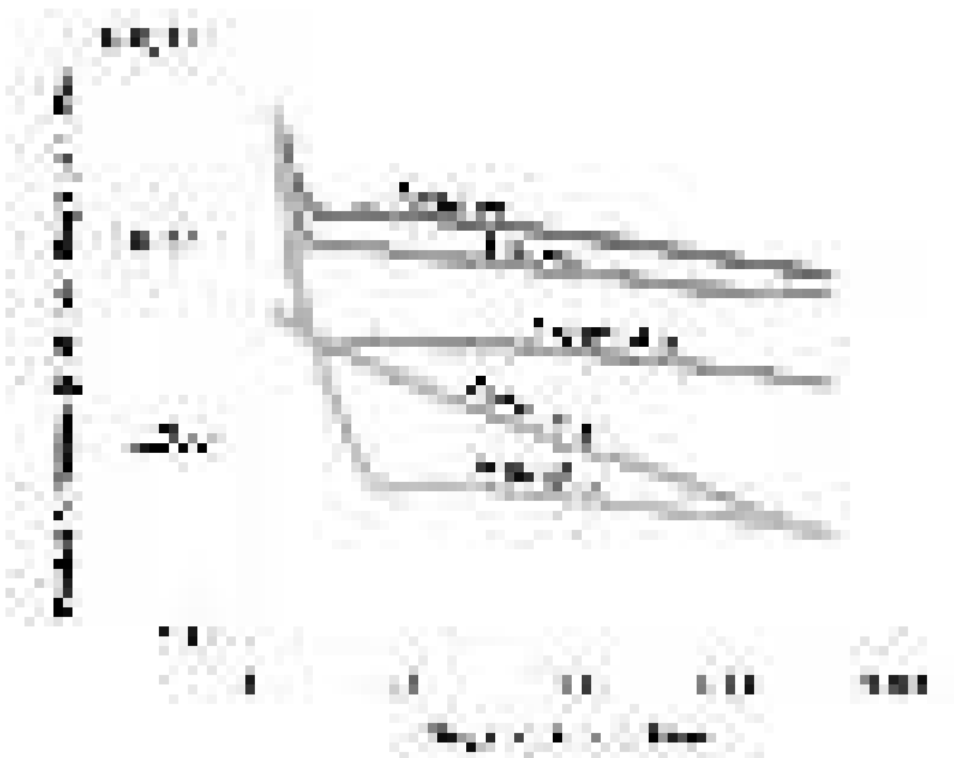


Figure 7-1. Generalized examples of produced water flow from five formations.
Actual produced water flows vary by location, play, basin, and amount of water used for hydraulic fracturing (EWI, 2015). Figure used with permission.

7.2.2.2 Coalbed Methane

Water is pumped from coal seams to reduce pressure so that gas adsorbed to the surface of the coal can flow to the production well (Guerra et al., 2011). Consequently, CBM tends to produce large volumes of water early on: more than conventional gas-bearing formations (U.S. GAO, 2012) (Figure 7-2). Within producing CBM formations, water production can vary for unknown reasons (U.S. GAO, 2012). As an example, data show that CBM production in the Powder River Basin produces 16 times more water than that in the San Juan Basin (U.S. GAO, 2012).

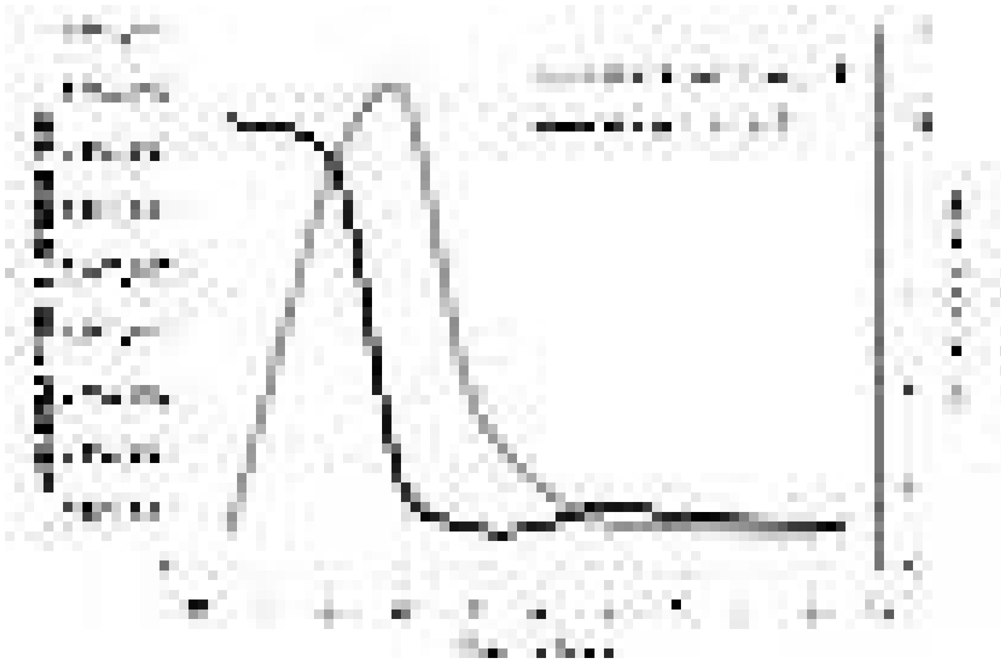


Figure 7-2. Typical produced water volume for a coal bed methane well in the western United States.
Source: Guerra et al. (2011).

7.3 Chemical Composition of Produced Water

For hydraulically fractured wells, the chemical composition of produced water changes from being similar to the injected hydraulic fracturing fluid to reflecting a mixture of hydraulic fracturing fluids, naturally occurring hydrocarbons, transformation products, and formation water. Initial produced water data show continuous changes in chemical composition and reflect processes occurring in the formation (Section 7.3.3). The data presented on longer-term produced water represent water that is primarily associated with the formation, rather than the hydraulic fracturing fluid (Section 7.3.4). Unlike the hydraulic fracturing fluid, the composition of which may be disclosed, compositional data on produced water comes from laboratory analysis of samples. Because of this reliance, we first discuss sampling and analysis of produced water, and especially note the limitations of existing analytical methods for organic chemicals and radionuclides.[1] It is important to note that the analytical methods can differ depending on the purpose of the analysis. Specifically, advanced laboratory methods have been used to identify unknown organic constituents of produced water (Section 7.3.1), routine methods are used for pre-drilling sampling, and a combination of methods may be needed for assessing environmental impacts (Section 7.4.2.5).

7.3.1 Determination of Produced Water Composition

Recent advances in analytical methods for produced water have allowed detection and quantification of a broad range of organic compounds, including those associated with hydraulic

[1] Chemical components of produced water are described below.

fracturing fluid (Section 7.3.4.7 and Appendix E.3.5.). These studies make clear that standard analytical methods are not adequate for detecting and quantifying the numerous organic chemicals, both naturally occurring and anthropogenic, that are now known to occur in produced water (Lester et al., 2015; Maguire-Boyle and Barron, 2014; Thurman et al., 2014). Similarly, methods commonly applied for the analysis of radionuclides in drinking water may suffer from analytical interferences that result in poor data quality (Maxwell et al., 2016; Ying et al., 2015; Zhang et al., 2015b; Nelson et al., 2014; U.S. EPA, 2014i, 2004b). In these instances, alternative methods that have been developed to support the nuclear materials production and waste industry provide more reliable approaches to ensure adequate detection limits and avoid sample matrix interferences that are anticipated for the high salinity and concentrations of organic constituents that may be present in produced water samples.[1] Development of advanced or non-routine methods for both organics and inorganics (especially radium) suggests that data generated from earlier methods may be less reliable that those developed by the new methods (Nelson et al., 2014), and that advanced analytical techniques are needed to detect or quantify some analytes.

The compositional data that follow in this chapter and Appendix E rely on the analytical procedures used in measurement and were summarized as noted from numerous produced water studies or compilations, such as the U.S. Geological Survey (USGS) produced water database (Blondes et al., 2014).

7.3.2 Factors Influencing Produced Water Composition

Several interacting factors influence the chemical composition of produced water: (1) the composition of injected hydraulic fracturing fluids, (2) the targeted geological formation and associated hydrocarbon products, (3) the stratigraphic environment, and (4) subsurface processes and residence time (Barbot et al., 2013; Chapman et al., 2012; Dahm et al., 2011; Blauch et al., 2009).

The mineralogy and structure of a formation are determined initially by deposition, when rock grains settle out of their transporting medium (Marshak, 2004). Generally, shale forms from clays that were deposited in deep, oxygen-poor marine environments, and sandstone can form from sand deposited in shallow marine environments (Ali et al., 2010; U.S. EPA, 2004a). Coal forms when carbon-rich plant matter collects in shallow peat swamps. In the United States, coal formed in both freshwater (northern Rocky Mountains) and marginal-marine environments (Alabama's Black Warrior formation) (NRC, 2010; Horsey, 1981). Consequently, shale and sandstone produced water are expected to be saline, and CBM water may be much less so.

7.3.3 Produced Water Composition During the Flowback Period

The chemistry of produced water changes over time, especially during the first days or weeks after hydraulic fracturing. Generally, produced water concentrations of cations, anions, metals, naturally occurring radioactive material (NORM), and organics increase as time goes on (Barbot et al., 2013; Haluszczak et al., 2013; Chapman et al., 2012; Davis et al., 2012; Gregory et al., 2011; Blauch et al.,

[1] For guidance in planning, implementing, and assessing projects that require laboratory analysis of radionuclides, see U.S. EPA (2004b).

2009). The causes include precipitation and dissolution of salts, carbonates, sulfates, and silicates; pyrite oxidation; leaching and biotransformation of organic compounds; and mobilization of NORM and trace elements. Concurrent precipitation of sulfates (e.g., $BaSO_4$) and carbonates (e.g., $CaCO_3$) alongside decreases in pH, alkalinity, dissolved carbon, and microbial abundance and diversity occur over time after hydraulic fracturing (Orem et al., 2014; Barbot et al., 2013; Murali Mohan et al., 2013; Davis et al., 2012; Blauch et al., 2009; Brinck and Frost, 2007). Leaching of organics appears to be a result of injected and formation fluids associating with shale and coal strata (Orem et al., 2014). Concentrations of organics in CBM produced water decrease with time, possibly due to the depletion of coal-associated water through formation pumping (Orem et al., 2007).

7.3.3.1 Total Dissolved Solids

Produced water total dissolved solids concentrations (TDS) increase by varying degrees because of the formation's geological origin. As an example, TDS concentrations increased to upper bound values in samples from four Marcellus Shale gas wells (Chapman et al., 2012) (Figure 7-3). The increased TDS was composed of increased sodium, calcium, and chloride (Chapman et al., 2012; Blauch et al., 2009). Similarly, TDS in flowback from the Westmoreland County wells started low and exceeded that of typical seawater (35,000 mg/L) within three days (Chapman et al., 2012). In a similar study, wells with hydraulic fracturing fluid containing less than 1,000 mg/L saw TDS concentrations increase above a median value of 200,000 mg/L within 90 days (Hayes, 2009).

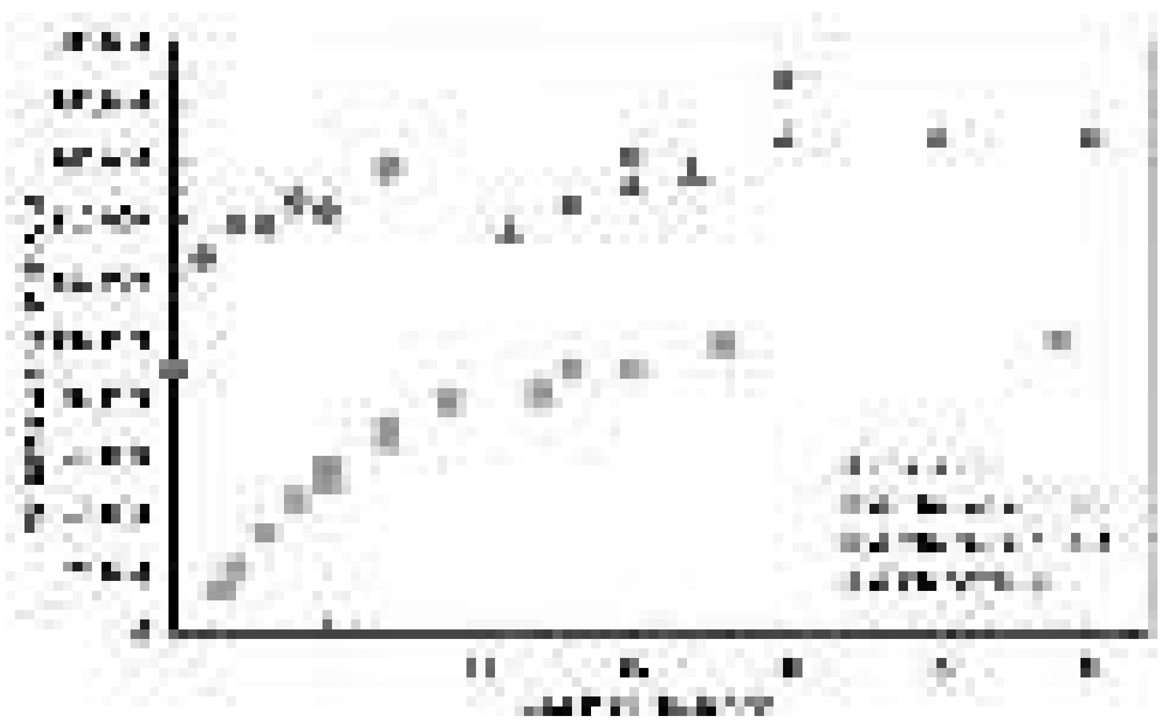


Figure 7-3. TDS concentrations measured through time for injected fluid (at 0 days), and produced water samples from four Marcellus Shale gas wells in three southwest Pennsylvania counties.
Data from Chapman et al. (2012).

7.3.3.2 Radionuclides

Shales and sandstones naturally contain various radionuclides (Sturchio et al., 2001).[1] Radium in pore waters or adsorbed onto clay particles and grain coatings can dissolve and return in produced water (Langmuir and Riese, 1985). Available data indicate that radium and TDS concentrations in produced water are positively correlated (Rowan et al., 2011; Fisher, 1998), likely because radium remains adsorbed to mineral surfaces when salinity is low, and then desorbs into solution with increased salinity (Sturchio et al., 2001). As an example, over the course of 20 days, radium concentration in flowback from a Marcellus Shale gas well increased by almost a factor of four (Chapman et al., 2012; Rowan et al., 2011) (Figure 7-4).

Figure 7-4. Total radium and TDS concentrations measured through time for injected (day 0), and produced water samples Greene County, PA, Marcellus Shale gas wells.
Data from Rowan et al. (2011) and Chapman et al. (2012).

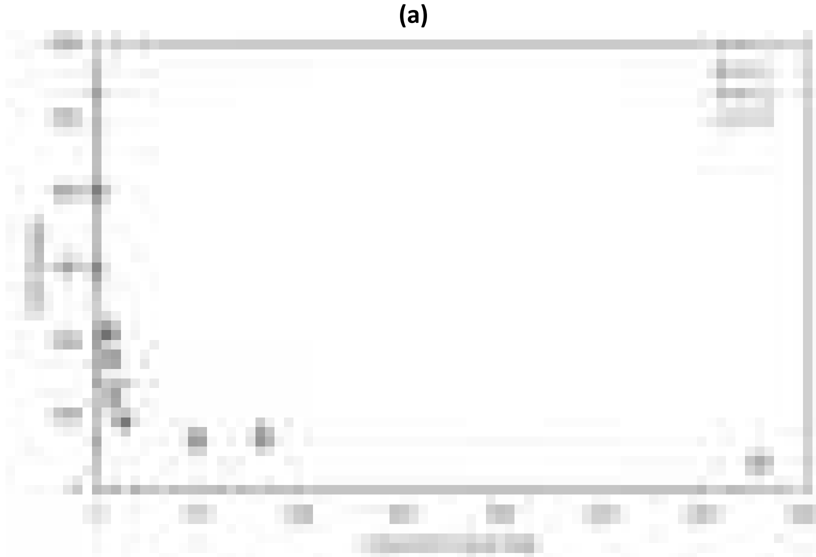
7.3.3.3 Dissolved Organic Carbon

Dissolved organic carbon (DOC) concentrations decrease from initial levels in shales and coalbeds (Murali Mohan et al., 2013; Orem et al., 2007). This occurs while TDS and chloride concentrations are increasing (Barbot et al., 2013; Chapman et al., 2012). DOC sorption, dilution with injected or formation water, biochemical reactions, and microbial transformation may all cause decreased concentrations of DOC during flowback. Injected organics can include gel polymer formulations, namely guar gum; petroleum distillates; and ethyl and ether glycol formulations, which can serve as food sources for microbes. (Wuchter et al., 2013; Arthur et al., 2009b; Hayes, 2009). In coalbeds,

[1] Hydraulic fracturing fluids typically do not contain radioactive material (Rowan et al., 2011). However, reusing produced water can introduce radioactive material into hydraulic fracturing fluid. See Section 7.3.4.6 and PA DEP (2015b).

water contacting the coal may become depleted in DOC to the degree that when outside water of lower DOC is produced, the resulting DOC concentrations in the produced water are reduced (Orem et al., 2014).

(a)



(b)

Figure 7-5. (a) Increasing chloride (Cl) and (b) decreasing DOC concentrations measured through time for samples from three Marcellus Shale gas wells on a single well pad in Greene County, PA.

Data from Cluff et al. (2014). Reprinted with permission from Cluff, M; Hartsock, A; Macrae, J; Carter, K; Mouser, PJ. (2014). Temporal changes in microbial ecology and geochemistry in produced water from hydraulically fractured Marcellus Shale Gas Wells. Environ Sci Technol 48: 6508-6517. Copyright 2014 American Chemical Society.

As an example, produced water DOC concentrations decreased from their initial levels twofold from the hydraulic fracturing fluid and initial samples (Figure 7-5b) followed by a decrease of 11-fold over nearly 11 months. The DOC leveled off several months after hydraulic fracturing, presumably as a result of in situ attenuation processes (Cluff et al., 2014). As DOC was decreasing, chloride concentrations increased five- to six-fold. These chloride concentrations increased linearly during the first two weeks (Cluff et al., 2014) and then later approached higher levels (Figure 7-5a). The pattern in the DOC and chloride levels reflected the changing composition of the produced water—initially high in DOC from hydraulic fracturing additives and low in salinity, then higher in salinity and lower in DOC reflecting the chemistry of formation water. The changing composition of produced water suggests that the potential concern for produced water spills also changes: initially the produced water may contain more hydraulic fracturing chemicals, and later the concern may shift to the impact of high salinity water.

7.3.4 Produced Water Composition

The chemical composition of produced water continues to change after the initial flowback period. Produced water may contain a range of constituents, but in widely varying amounts. Generally, these can include:

- Salts, including those composed from chloride, bromide, sulfate, sodium, magnesium and calcium;
- Metals including barium, manganese, iron, and strontium;
- Radioactive materials including radium (radium-226 and radium-228);
- Oil and grease, and dissolved organics (including BTEX);[1]
- Hydraulic fracturing chemicals, including tracers and their transformation products; and
- Produced water treatment chemicals.[2]

We discuss these groups of chemicals and then conclude by discussing variability within formation types and within production zones.

7.3.4.1 Similarity of Produced Water from Conventional and Unconventional Reservoirs

Produced water generated from unconventional reservoirs is reported to be similar to produced water from conventional reservoirs in terms of TDS, pH, alkalinity, oil and grease, TOC, and other organics and inorganics (Wilson, 2014; Haluszczak et al., 2013; Alley et al., 2011; Hayes, 2009; Sirivedhin and Dallbauman, 2004). Although produced water salinity varies within and among shales and tight formations, produced water is typically characterized as saline (Lee and Neff, 2011; Blauch et al., 2009). Produced water from coalbeds may have low TDS if the coal source bed was formed in freshwater. Saline produced water is also enriched in major anions (e.g., chloride, bicarbonate, sulfate); cations (e.g., sodium, calcium, magnesium); metals (e.g., barium, strontium);

[1] BTEX is an acronym representing benzene, toluene, ethylbenzene, and xylenes.

[2] Some chemicals are added to produced water for the purpose of oil/water separation, improved pipeline flow, or equipment maintenance, including prevention of corrosion and scaling in equipment (Cal/EPA, 2016). Generally the chemicals serve as clarifiers, emulsifiers, emulsion breakers, floating agents, and oxygen scavengers. Among proprietary formulations, a few specific chemicals have been disclosed including low concentrations of benzene, toluene, and inorganics (acetic acid, ammonium chloride, cupric sulfate, sodium hypochlorite).

naturally occurring radionuclides (e.g., radium-226, radium-228) (Chapman et al., 2012; Rowan et al., 2011); and organics (e.g., hydrocarbons) (Orem et al., 2007; Sirivedhin and Dallbauman, 2004).

7.3.4.2 Variability in Produced Water Composition Among Unconventional Reservoirs

Alley et al. (2011) compared geochemical parameters of shale gas, tight gas, and CBM produced water. This comparison aggregated data on produced water from original analyses, peer-reviewed literature, and public and confidential government and industry sources and determined the statistical significance of the results.

As shown in Table 7-5, Alley et al. (2011) found that of the constituents of interest common to all three types of produced water from unconventional reservoirs (calcium, chloride, potassium, magnesium, manganese, sodium, and zinc):

1. Shale gas produced water had significantly different concentrations from those of CBM;
2. Shale gas produced water constituent concentrations were significantly similar to those of tight gas, except for potassium and magnesium; and
3. Five tight gas produced water constituent concentrations (calcium, chloride, potassium, magnesium, and sodium) were significantly similar to those of CBM (Alley et al., 2011).

The degree of variability between produced waters of these three resource types is consistent with the degree of mineralogical and geochemical similarity between shale and sandstone formations, and the lack of the same between shale and coalbed formations (Marshak, 2004). Compared to the others, shale gas produced water tends to be more acidic, as well as enriched in strontium, barium, and bromide. CBM produced water is alkaline, and it contains relatively low concentrations of TDS (one to two orders of magnitude lower than in shale and sandstone). It also contains lower levels of sulfate, calcium, magnesium, DOC, sodium, bicarbonate, and oil and grease than typically observed in shale and sandstone produced waters (Alley et al., 2011; Dahm et al., 2011; Benko and Drewes, 2008; Van Voast, 2003).[1]

Table 7-5. Compiled minimum and maximum concentrations for various geochemical constituents in produced water from shale gas, tight gas, and CBM produced water.
Source: Alley et al. (2011).

Parameter	Unit	Shale gas[a]	Tight Gas Sands[b]	CBM[c]
Alkalinity	mg/L	160–188	1,424	54.9–9,450
Ammonium-N	mg/L	-	2.74	1.05–59
Bicarbonate	mg/L	ND–4,000	10–4,040	-
Conductivity	µS/cm	-	24,400	94.8–145,000
Nitrate	mg/L	ND–2,670	-	0.002–18.7

[1] Several regions had low representation in the Alley et al. (2011) data set, including the Appalachian Basin (western New York and western Pennsylvania), West Virginia, eastern Kentucky, eastern Tennessee, and northeastern Alabama.

Parameter	Unit	Shale gas[a]	Tight Gas Sands[b]	CBM[c]
Oil and grease	mg/L	-	42	-
pH	SU[d]	1.21−8.36	5−8.6	6.56−9.87
Phosphate	mg/L	ND−5.3	-	0.05−1.5
Sulfate	mg/L	ND−3,663	12−48	0.01−5,590
Radium-226	pCi/g	0.65−1.031	-	-
Aluminum	mg/L	ND−5,290	-	0.5−5,290
Arsenic	mg/L	-	0.17	0.0001−0.06
Boron	mg/L	0.12−24	-	0.002−2.4
Barium	mg/L	ND−4,370	-	0.01−190
Bromide	mg/L	ND−10,600	-	0.002−300
Calcium	mg/L	0.65−83,950	3−74,185	0.8−5,870
Cadmium	mg/L	-	0.37	0.0001−0.01
Chloride	mg/L	48.9−212,700	52−216,000	0.7−70,100
Chromium	mg/L	-	0.265	0.001−0.053
Copper	mg/L	ND−15	0.539	ND−0.06
Fluorine	mg/L	ND−33	-	0.05−15.22
Iron	mg/L	ND−2,838	0.015	0.002−220
Lithium	mg/L	ND−611	-	0.0002−6.88
Magnesium	mg/L	1.08−25,340	2−8,750	0.2−1,830
Manganese	mg/L	ND−96.5	0.525	0.002−5.4
Mercury	mg/L	-	-	0.0001−0.0004
Nickel	mg/L	-	0.123	0.0003−0.20
Potassium	mg/L	0.21−5,490	5−2,500	0.3−186
Sodium	mg/L	10.04−204,302	648−80,000	8.8−34,100
Strontium	mg/L	0.03−1,310	-	0.032−565
Uranium	mg/L	-	-	0.002−0.012
Zinc	mg/L	ND−20	0.076	0.00002−0.59

-, No value available; ND, non-detect. If no range, but a singular concentration is given, this is the maximum concentration.

[a] *n* = 541. Alley et al. (2011) compiled data from USGS (2006); McIntosh and Walter (2005); McIntosh et al. (2002) and confidential industry documents.

[b] *n* = 137. Alley et al. (2011) compiled data from USGS (2006) and produced water samples presented in Alley et al. (2011).

[c] Alley et al. (2011) compiled data from Montana GWIC (2009); Thordsen et al. (2007); ESN Rocky Mountain (2003); Rice et al. (2000); Rice (1999); Hunter and Moser (1990).

[d] SU = standard units.

7.3.4.3 General Water Quality Parameters

Data characterizing the content of produced water from unconventional reservoirs in 12 shale and tight formations and CBM basins were evaluated for this assessment. These reservoirs and basins include parts of 18 states, but the data do not allow for comparison of trends over time.

For most reservoirs, the amount of available general water quality parameter data is variable (see Appendix Table E-2 for an example). Average pH levels range from 5.87 to 8.19, with typically lower values for shales. Larger variations in average specific conductivity are seen among unconventional reservoirs and range from 213 microsiemens (μS)/cm in the Bakken Shale to 184,800 μS/cm in Devonian sandstones (Appendix Table E-2). Shale and tight formation produced waters are enriched in suspended solids, as reported concentrations for total suspended solids and turbidity exceed those of coalbeds by one to two orders of magnitude.

The average dissolved oxygen (DO) concentrations of CBM produced water range from 0.39-1.07 mg/L (Appendix Table E-3). By comparison, well-oxygenated surface water can contain up to 10 mg/L DO at 59 °F (15 °C) (U.S. EPA, 2012a). Thus, coalbed produced water is either hypoxic (less than 2 mg/L DO) or anoxic (less than 0.5 mg/L DO) and, if released to surface waters, could contribute to aquatic organism stress (USGS, 2010; NSTC, 2000).

7.3.4.4 Salinity and Inorganics

The TDS profile of produced water from unconventional reservoirs is dominated by sodium and chloride, with large contributions to the profile from mono- and divalent cations (Sun et al., 2013; Guerra et al., 2011). Shale and sandstone produced waters tend to be characterized as sodium-chloride-calcium water types, whereas CBM produced water tends to be characterized as sodium chloride or sodium bicarbonate water types (Dahm et al., 2011). Elevated levels of bromide, sulfate, and bicarbonate are also present (Sun et al., 2013). Elevated strontium and barium levels are characteristic of Marcellus Shale produced water (Barbot et al., 2013; Haluszczak et al., 2013; Chapman et al., 2012). Data representing shales and tight formations are presented in Appendix Table E-4.

Marcellus Shale produced water salinities range from less than 1,500 mg/L to over 300,000 mg/L, as shown by Rowan et al. (2011). By comparison, the average salinity concentration for seawater is 35,000 mg/L.

Of the CBM data presented in Appendix Table E-5, differences are evident between the Black Warrior and the three western formations (Powder River, Raton, and San Juan). The Black Warrior is higher in average chloride, specific conductivity, TDS, TOC, and total suspended solids, and lower in alkalinity and bicarbonate than the other three. These differences are due to the saline or brackish conditions during deposition in the Black Warrior, and its older geologic age that contrasts with the freshwater conditions for the younger western basins. The TDS concentration of CBM produced water can range from 170 mg/L to nearly 43,000 mg/L (range composited from Dahm et al. (2011) and Benko and Drewes (2008); see also Van Voast (2003)).[1]

7.3.4.5 Metals

The metals content of produced water from unconventional reservoirs varies by well and site lithology. Levels of iron, magnesium, and boron were within ranges known for conventional produced water (Hayes, 2009). Produced water from unconventional reservoirs may also contain low levels of heavy metals (e.g., chromium, copper, nickel, zinc, cadmium, lead, arsenic, and mercury as found by Hayes). Data illustrating metal concentrations in produced water appear in Appendix Tables E-6 and E-7.

7.3.4.6 Naturally Occurring Radioactive Material (NORM) and Technologically Enhanced Naturally Occurring Radioactive Material (TENORM)

Geologic environments contain naturally occurring radioactive material (NORM). Radioactive materials commonly present in shale and sandstone sedimentary environments include uranium, thorium, radium, and their decay products. Elevated formation uranium levels have been used to identify potential areas of natural gas production for decades (Fertl and Chilingar, 1988). Shales that contain significant levels of uranium include the Barnett in Texas, the Woodford in Oklahoma, the New Albany in the Illinois Basin, the Chattanooga Shale in the southeastern United States, and a group of black shales in Kansas and Oklahoma (Swanson, 1955).[2] When exposed to the environment in produced water, NORM is called *technologically enhanced* naturally occurring radioactive material (TENORM).[3] Water soluble forms of TENORM are present in most produced water from unconventional reservoirs, but particularly so in Marcellus Shale produced water (Rowan et al., 2011; Fisher, 1998).

Due to insolubility under prevailing reducing conditions encountered within shale formations, only low levels of uranium and thorium are found in produced water, typically in the concentrated form of mineral phases or organic matter (Nelson et al., 2014; Sturchio et al., 2001). Conversely, radium, a decay product of uranium and thorium, is known to be relatively soluble within the redox range encountered in subsurface environments (Sturchio et al., 2001; Langmuir and Riese, 1985). As noted in Section 7.3.3.2, radium and TDS produced water concentrations are positively correlated (Rowan et al., 2011; Fisher, 1998); therefore, in formations containing radium, increasing TDS concentration indicates likely increasing radium concentration.

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[1] From a similar dataset, Dahm et al. (2011) report TDS concentrations from a composite CBM produced water database (n = 3,255) for western basins that often are less than 5,000 mg/L (85% of samples).

[2] Marine black shales are estimated to contain an average of 15–60 ppm uranium depending on depositional conditions (Fertl and Chilingar, 1988).

[3] The U.S. EPA Office of Air and Radiation's website (https://www.epa.gov/radiation/technologically-enhanced-naturally-occurring-radioactive-materials-tenorm) states that TENORM is produced when activities such as uranium mining or sewage sludge treatment concentrate or expose radioactive materials that occur naturally in ores, soils, water, or other natural materials. Formation water containing radioactive materials contains NORM, because it is not exposed; produced water contains TENORM, because it has been exposed to the environment.

Median values of total radium in the Marcellus Shale ranged from about 1,000 pCi/L to less than 6,000 pCi/L, which are values far exceeding the industrial discharge limit of 60 pCi/L (Rowan et al., 2011) (Figure 7-6). In the Marcellus Shale, TENORM levels in produced water from unconventional reservoirs exceeded levels from conventional reservoirs levels by factors of 4 to 26 (PA DEP, 2015b) (Appendix Table E-8). The individual median concentrations in produced water from unconventional reservoirs of 11,300 pCi/L gross alpha, 3,445 pCi/L gross beta, and total radium of 7,180 pCi/L (Appendix Table E-8). TENORM has been identified in hydraulic fracturing fluid, presumably due to the reuse of produced water at levels from 2 to 4.5 times lower than produced water from unconventional reservoirs (PA DEP, 2015b) (Appendix Table E-8).

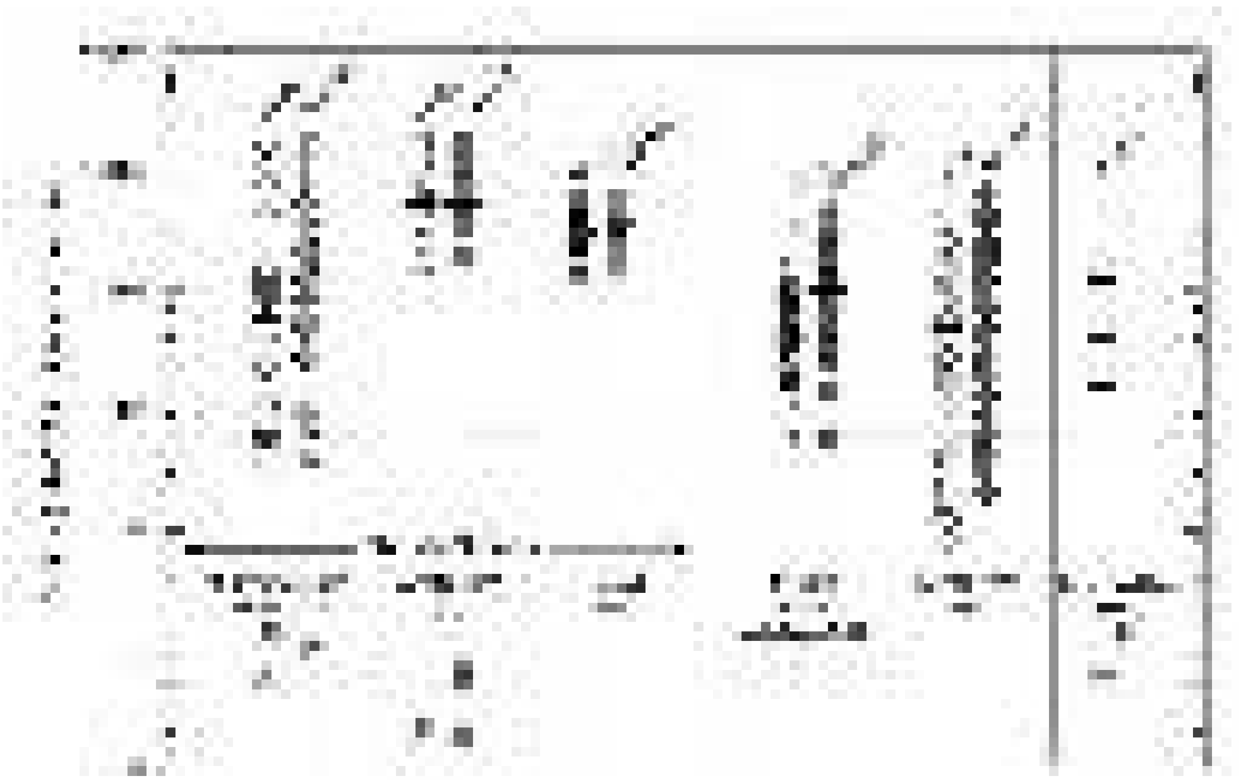


Figure 7-6. Data on radium 226 (open symbols) and total radium (filled symbols) for Marcellus Shale wells (leftmost three columns) and other formations (rightmost three columns).
Source: Rowan et al. (2011). The dashed line represents the industrial effluent discharge limit of 60 pCi/L set by the Nuclear Regulatory Commission. The black lines indicate the median concentrations, and the number of points in each dataset are shown in parentheses. Citations within the figure are provided in Rowan et al. (2011).

7.3.4.7 Organics

The organic content of produced water varies by well and lithology, but consists of naturally occurring and injected organic compounds (Lee and Neff, 2011). Of the organics detected by either routine or advanced analytical methods (Section 7.3.1), the majority are naturally occurring constituents of petroleum (Appendix Tables H-4 and H-5). These organics may be dissolved in water or, in the case of oil production, in the form of a separate or emulsified phase. Several classes of organic chemicals have been found in shale gas and CBM produced water, including aromatics, polyaromatic hydrocarbons, heterocyclic compounds, aromatic amines, phenols, phthalates, aliphatic alcohols, fatty acids, and nonaromatic compounds (list from Orem et al. (2014), see also: Hayes (2009), Benko and Drewes (2008), Orem et al. (2007), and Sirivedhin and Dallbauman (2004)). Compounds found in CBM waters included pyrene, phenanthrenone, alkyl phthalates, C_{12} through C_{18} fatty acids, and others. Similarly, compounds found in shale gas produced water included pyrene and perylene, ethylene glycol, diethylene glycol monododecyl ether, 2-(2-butoxyethoxy) ethanol, and others (Orem et al., 2014). Biomarkers—organic molecules characteristically produced by life forms, and unique to shale formations—have recently been suggested to fingerprint produced water (Hoelzer et al., 2016). More representative examples from five coal bed and two shale gas formations with reported concentrations are given in Appendix Tables E-9, E-11, and E-12, and the complete list of chemicals with CAS registry numbers identified by the EPA for this assessment appears in Appendix H. (See Appendix Table H-4 for chemicals with EPA-identified CAS numbers and Appendix Table H-5 for chemicals without.) Appendix Table E-13 lists concentrations of organic chemicals that were identified in three specific studies (Khan et al., 2016; Lester et al., 2015; Orem et al., 2007).

7.3.4.8 Hydraulic Fracturing Fluid Additives

Several chemicals used in hydraulic fracturing fluids have been identified in produced water. (Examples are shown in Table 7-6, Appendix Table E-10, and Appendix Tables H-4 and H-5.) Many of these chemicals were identified through advanced analytical procedures and equipment, and would not be expected to be found by routine analyses. Of note is that phthalates do not occur naturally. Their presence in produced water is due to either their use in hydraulic fracturing fluids; polyvinyl chloride (PVC) in well adhesives, valves, or fittings; or coatings on laboratory sample bottles (Orem et al., 2007).[1] Phthalates can also be used in drilling fluids, as breaker additives, or as plasticizers (Maguire-Boyle and Barron, 2014; Hayes and Severin, 2012a).[2] One of the produced water phthalates has been identified as a component of hydraulic fracturing fluid (di(2-ethylhexyl) phthalate) (Appendix Table H-2), while others have not, and those may originate from laboratory or field equipment.

Table 7-6. Examples of compounds identified in produced water that can be components of hydraulic fracturing fluid.

Appendix Tables H-4 and H-5 list chemicals identified in produced water and indicates those also identified as constituents of hydraulic fracturing fluid. Chemical or class designation in this table is taken directly from the text of the cited references except where noted, and may or may not reflect the chemical names from the Distributed Structure-Searchable Toxicity Database (DSSTox) show in Appendix Table H-4 or other chemicals listed in Appendix Table H-5.

Chemical or class	Use	Reference
2-Butanone	Solvent; microbial degradation product	Lester et al. (2015)

[1] Examples include di(2-ethylhexyl) phthalate, diisodecyl phthalate, and diisononyl phthalate (Orem et al., 2007).

[2] Specifically fatty acid phthalate esters (Maguire-Boyle and Barron, 2014).

Chemical or class	Use	Reference
2-Butoxyethanol	Acid dispersant, solvent, non-emulsifier	Thacker et al. (2015)
Acetone	Solvent; microbial degradation product	Lester et al. (2015)
Cocamidopropyl dimethylamine (C-7)	Foaming and lubrication enhancer	Lester et al. (2015)
Di(2-ethylhexyl) phthalate[a]	Derivative of polyvinyl chloride used in adhesives, valves, fittings or coatings of sample bottles	Orem et al. (2007)
Diethylene glycol monododecyl ether	Antifreeze, scale inhibitor, friction reducer	Orem et al. (2014)
Dioctadecyl ester of phosphate phosphoric acid	Common lubricant	Maguire-Boyle and Barron (2014)
Ethylene glycol	Antifreeze, scale inhibitor, friction reducer	Orem et al. (2014)
Fatty acid phthalate esters	(Related to) use in drilling fluids and breakers	Maguire-Boyle and Barron (2014)
Fluorocarbons	Tracers	Maguire-Boyle and Barron (2014)
Hexahydro-1,3,5-trimethyl-1,3,5-triazine-2-thione	Biocide	Orem et al. (2014)
Linear alkyl ethoxylates (C-4 to C-8, C-11 to C-14)	Enhancer of surfactant properties	Lester et al. (2015); Thurman et al. (2014)
Polyethylene glycol carboxylates (PEG-C-EO2 to PEG-C-EO10)	Friction reducer, clay stabilizer, surfactants	Thurman et al. (2016)
Polyethylene glycols (PEG-EO4 to PEG-EO10)	Friction reducer, clay stabilizer, surfactants	Thurman et al. (2016)
Polypropylene glycols (PPG-PO2 to PPG PO10)	Friction reducer, clay stabilizer, surfactants	Thurman et al. (2016)
Toluene	Solvent, scale inhibitor	Thacker et al. (2015)
Triethylene glycol monododecyl ether	Antifreeze, scale inhibitor, friction reducer	Orem et al. (2014)
Xylenes	Solvent, scale inhibitor	Thacker et al. (2015)

[a] Di(2-ethylhexyl) phthalate was named di-2-ethyl hexyl phthalate in Maguire-Boyle and Barron (2014).

7.3.4.9 Reactions within Formations

The introduction of hydraulic fracturing fluids into the target formation induces a number of changes to formation solids and fluids that influence the chemical evolution and composition of produced water. These changes can result from physical processes (e.g., rock fracturing and fluid mixing); geochemical processes (e.g., introducing oxygenated fluids of composition unlike that of the formation); and down hole conditions (elevated temperature, salinity, and pressure) that mobilize trace or major constituents into solution.

The creation of fractures exposes new formation surfaces to interactions involving hydraulic fracturing fluids and existing formation fluids. Formations in unconventional reservoirs targeted for development are composed of detrital, cement, and organic fractions. For example, elements potentially available for mobilization when exposed via fracturing include calcium, magnesium, manganese, and strontium in cement fractions, and silver, chromium, copper, molybdenum, niobium, vanadium, and zinc in organic fractions.

From organic compounds identified in five flowback samples and one produced water sample from the Fayetteville Shale, three possible types of reactions were identified by Hoelzer et al. (2016): hydrolysis of delayed acids, oxidant-caused halogenation reactions, and transformation of disclosed additives. First, delayed acids are used to "break" gel structures and would be intentionally introduced for their ability to cause in-formation reactions. Second, strong oxidants or other compounds introduced as breakers, along with elevated temperature and salinity, can trigger reactions between halogens (chloride, bromide, and iodide) and methane, acetone and pyrane resulting in halomethane compounds. A similar suggestion was made by Maguire-Boyle and Barron (2014). Low pH was found to promote oxidation of additives (Tasker et al., 2016). Third, known additives may react to form byproducts. Hoelzer et al. (2016) postulate examples from several types of compounds, two of these are the formation of benzyl alcohol from the hydraulic fracturing additive benzyl chloride, and abiotic and biotic reactions of phenols. In a study that used synthetic fracturing fluid, Tasker et al. (2016) reported that surfactants were recalcitrant to degradation under high pressure and temperature, which may explain the presence of the surfactant glycols in produced water as reported by Thurman et al. (2016) (Table 7-6), and the oxidation of other additives (gelling and some friction reducers (Table 5-1)) may explain their absence.

7.3.5 Spatial Trends in Produced Water Composition

As was reported for the volume of produced water (Section 7.2.2), the composition of produced water varies spatially on a regional to local scale according to the geographic and stratigraphic locations of each well within a hydraulically fractured production zone (Bibby et al., 2013; Lee and Neff, 2011). Spatial variability of produced water content occurs: (1) between plays of different rock sources (e.g., coal vs. sandstone); (2) between plays of the same rock type (e.g., Barnett Shale vs. Bakken Shale); and (3) within formations of the same source rock (e.g., northeastern vs. southwestern Marcellus Shale) (Barbot et al., 2013; Alley et al., 2011; Breit, 2002).

Geographic variability in produced water content has been established at a regional scale for conventional produced water. As an example, Benko and Drewes (2008) demonstrate TDS variability in conventional produced water among fourteen western geologic basins (e.g., Williston, San Juan, and Permian Basins). Median TDS in these basins range from as low as 4,900 mg/L in the Big Horn Basin to as high as 132,400 mg/L in the Williston Basin based on over 133,000 produced water samples from fourteen basins (Benko and Drewes, 2008).[1]

Average or median TDS of more than 100,000 mg/L has been reported for the Bakken (North Dakota, Montana) and Marcellus (Pennsylvania) formations; between 50,000 mg/L and 100,000 mg/L for the Barnett (Texas), and less than 50,000 mg/L for the Fayetteville (Arkansas) shale formations.[2] In tight formations, the average TDS was above 100,000 mg/L for the Devonian Sandstone (Pennsylvania) and Cotton Valley Group (Louisiana, Texas), between 50,000 mg/L and 100,000 mg/L for the Oswego (Oklahoma), and less than 50,000 mg/L for the Mesaverde Formation (Colorado, New Mexico, Utah, Wyoming). Maximum concentrations above 200,000 mg/L have been reported for the Marcellus, Bakken, Cotton Valley Group and Devonian Sandstone (Appendix Table E-2).

CBM produced waters had average TDS of less than 5,000 mg/L in the Powder River (Montana, Wyoming), Raton (Colorado, New Mexico), and San Juan (Arizona, Colorado, New Mexico, Utah) basins; while above 10,000 mg/L in the Black Warrior Basin (Alabama, Mississippi), which as noted above are due to the depositional history of these basins (Appendix Table E-3, Section 7.3.2).

Data further illustrating variability within both shale and tight gas reservoirs, as well as coalbed methane fields, at both the formation and local scales are presented and discussed in Appendix Section E.3.

7.4 Spill and Release Impacts on Drinking Water Resources

Surface spills of produced water from oil and gas production have occurred across the country and, in some cases, have caused impacts to drinking water resources. Released fluids can flow into nearby surface waters, if not contained on-site, or infiltrate into groundwater via soil. In this section, we first briefly describe the potential for spills from produced water handling equipment. Next, we address individually reported spill events. These have originated from pipeline leaks, well blowouts, well communication events, and leaking pits and impoundments. We then summarize several studies of aggregated spill data, which are based on state agency spill reports.

7.4.1 Produced Water Handling and Spill Potential

Throughout the production phase at oil and certain wet gas production facilities, produced water is stored in containers and pits that can contain free phase, dissolved phase, and emulsified crude oil. Since the crude oil is not efficiently separated out by the flow-through process vessels (such as

[1] Data were drawn from the USGS National Produced Water Geochemical Database v2.0. Published updates made in October 2014 to the database (v2.1) are not reflected in this document.

[2] Because publications we are comparing may report either average or median values (but not uniformly both), we combine average and medians in this paragraph.

three-phase separators, heater treaters, or gun barrels), this crude oil can remain present in the produced water container or pit.

Produced water can be transferred to surface pits for long-term storage and evaporation. Surface pits are typically uncovered, earthen pits that may or may not be lined.[1] Unlined pits can lead to contamination of groundwater, especially shallow alluvial systems. Recovered fluids can overflow or leak from surface pits due to improper pit design and weather events.

Produced water that is to be treated or disposed of off-site is typically stored in storage tanks or pits until it can be loaded into transport trucks for removal (Gilmore et al., 2013). Tank storage systems are typically closed loop systems in which produced water is transported from the wellhead to aboveground storage tanks through interconnecting pipelines (GWPC and IOGCC, 2014). Failure of connections and lines during the transfer process or the failure of a storage tank can result in a surface release of fluids.

Depending on its characteristics, produced water can be recycled and reused on-site. It can be directly reused without treatment (after blending with freshwater), or it can be treated on-site prior to reuse (Boschee, 2014). As with other produced water management options, these systems also can spill during transfer of fluids.

7.4.2 Spills of Produced Water

7.4.2.1 Pipeline Leaks

Produced water is typically transported from the wellhead through a series of pipes or flowlines to on-site storage or treatment units (GWPC and IOGCC, 2014), or nearby injection wells. Faulty connections at either end of the transfer process or leaks or ruptures in the lines carrying the fluid can result in surface spills. A field report from PA DEP (2009b) described a leak from a 90-degree bend in an overland pipe carrying a mixture of produced water and freshwater between two pits. The impact included a "dull sheen" on the water and measured chloride concentration of 11,000 mg/L. The leak impacted a 0.4 mi (0.6 km) length of a stream, and fish and salamanders were killed. Beyond a confluence at 0.4 mi (0.6 km) with a creek, no additional dead fish were found. The release was estimated at 11,000 gal (42,000 L). In response to the incident, the pipeline was shut off, a dam was constructed for recovering the water, water was vacuumed from the stream, and the stream was flushed with fresh water (PA DEP, 2009b).

Another example of a pipeline release occurred in January 2015, when 70,000 bbls (2,940,000 gal or 11,130,000 L) of produced water containing petroleum hydrocarbons (North Dakota Department of Health, 2015) were released from a broken pipeline that crosses Blacktail Creek in Williams County, ND. The response included placing absorbent booms in the creek, excavating contaminated soil, removing oil-coated ice, and removing produced water from the creek. The electrical conductivity and chloride concentration in the water along the creek, the Little Muddy River, and Missouri River were found to be elevated above background levels, as were samples

[1] The use of the terms "impoundments" and "pits" varies and is described in Chapter 8. For the purposes of this section, the term "pits" will be generally used to cover all below-grade storage (but not above ground closed or open tanks).

taken from groundwater recovery trenches. Remediation work on this site continues as of the date of this writing (August, 2016).

7.4.2.2 Well Blowouts

Spills of produced water have occurred as a result of well blowouts. Fingerprinting of water from two monitoring wells in Killdeer, ND, was used to determine that brine contamination in the two wells resulted from a well blowout during a hydraulic fracturing operation. See the discussion in Section 6.2.2.1 for more information.

Another example of a well blowout associated with a hydraulic fracturing operation occurred in Clearfield County, PA. The well blew out, resulting in an uncontrolled flow of approximately 35,000 gal (132,000 L) of brine and fracturing fluid; some of the liquids reportedly reached a nearby stream (Barnes, 2010). The blowout occurred during drilling of plugs that were used to isolate fracture stages from each other. An independent investigation found that the primary cause of the incident was that the sole blowout preventer on the well had not been properly tested. In addition, the company did not have certified well control experts on hand or a written pressure control procedure (Vittitow, 2010).

In North Dakota, a blowout preventer failed, causing a release of between 50 and 70 bbls per day (2,100 gal/day or 7,900 L/day and 2,940 gal/day or 11,100 L/day) of produced water and oil (Reuters, 2014). Frozen droplets of oil and water sprayed on a nearby frozen creek. Liquid flowing from the well was collected and trucked offsite. A 3-ft (0.9-m) berm was placed around the well for containment. Multiple well communication events have also led to produced water spills ranging from around 700 to 35,000 gal (2,600 L to 130,000 L) (Vaidyanathan, 2013a). Well communication is described in Section 6.3.2.3.

The Chesapeake Energy ATGAS 2H well, located in Leroy Township, Bradford County, PA, experienced a wellhead flange failure on April 19, 2011, during hydraulic fracturing operations. Approximately 10,000 gal (38,000 L) of produced water spilled into an unnamed tributary of Towanda Creek, a state-designated trout stock fishery and a tributary of the Susquehanna River (USGS, 2013b; SAIC and GES, 2011). Chesapeake conducted post-spill surface water and groundwater monitoring (SAIC and GES, 2011).

Chesapeake concluded that there were short-term impacts to surface waters of a farm pond within the vicinity of the well pad, the unnamed tributary, and Towanda Creek following the event (SAIC and GES, 2011). The lower 500 ft (200 m) of the unnamed tributary exhibited elevated chloride, TDS, and specific conductance, which returned to background levels in less than a week. Towanda Creek experienced these same elevations in concentration, but only at its confluence with the unnamed tributary; elevated chloride, TDS, and specific conductance returned to background levels the day after the blowout (SAIC and GES, 2011).

7.4.2.3 Leaks from Pits and Impoundments

Leaks of produced water from on-site pits have caused releases as large as 57,000 gal (220,000 L) and have caused surface water and groundwater impacts (Vaidyanathan, 2013b; Levis, 2011; 2010c; PADEP, 2010). VOCs have been measured in groundwater near the Duncan Oil Field in New Mexico downgradient of an unlined pit storing produced water. More example releases from pits are described in Section 8.4.5.

Two of EPA's retrospective case studies evaluated potential impacts from produced water pits. The EPA retrospective case studies were designed to determine whether multiple lines of evidence might be found that could specifically link constituent(s) found in drinking water to hydraulic fracturing activities using the tiered assessment framework presented in Appendix Section E.6. A multiple-lines-of-evidence approach was used to evaluate potential cause-and-effect relationships between hydraulic fracturing activities and contaminant presence in groundwater. Such an approach is needed, because the presence of a constituent in groundwater that is also found in hydraulic fracturing fluids or produced water does not necessarily implicate hydraulic fracturing activities as the cause. This is because some constituents of hydraulic fracturing fluids or produced water are ubiquitous in society (i.e., BTEX), and some constituents of produced water can be present in groundwater as background constituents (i.e., methane, iron, and manganese).

Elements of the assessment framework include gathering background information, including pre-drilling sample results; developing a conceptual model of the site; and assessing multiple analytes to develop lines of evidence. Development of these requires adherence to sampling and quality assurance protocols to generate defensible data. Among many other quality assurance requirements, proper well purging and analyses of field and laboratory blanks are needed (Appendix Table E-17 and Figure E-15).

In the EPA's *Retrospective Case Study in Southwestern Pennsylvania: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2015j), elevated chloride concentrations and their timing relative to historical data suggested a recent groundwater impact on a private water well occurred near a pit. The water quality trends suggested that the chloride anomaly was related to the pit, but site-specific data were not available to provide a definitive assessment of the cause(s) and the longevity of the impact. Evaluation of other water quality parameters did not provide clear evidence of produced water impacts.

In the EPA's *Retrospective Case Study in Wise County, Texas: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2015l), impacts to two water wells were attributed to brine, but the data collected for the study were not sufficient to distinguish among multiple possible brine sources, including reserve pits, migration from underlying formations along wellbores, migration from underlying formation along natural fractures and a nearby brine injection well.

To aid in assessing impacts, a number of geochemical indicators and isotopic tracers for identifying oil and gas produced water have been identified. These include (Lauer et al., 2016; Warner et al., 2014a, b):

- Common ion ratios, including bromide/chloride and lithium/chloride;
- Isotope ratios, especially Strontium isotope ratios ($^{87}Sr/^{86}Sr$); and

- Enrichment of certain isotopes: $\delta^{18}O$, $\delta^{2}H$, $\delta^{7}Li$, $\delta^{13}C$-DIC, $\delta^{11}B$.[1]

For the case study, twelve geochemical indicators, including the bromine/chlorine (Br/Cl) and strontium isotope ratios, were considered for the well-water samples.[2] The results were used to assess whether the likelihood that the observed values originated with produced water (the aforementioned sources of brine), sea water, road salt, landfill leachate, sewage/septic tank leachate, and animal waste. In each sample evaluated, it was found that the water could have originated with one or more of the six sources. Thus these lines of evidence did not allow identification of neither a specific source nor a hydraulic fracturing source (Appendix Table E-18). A third well experienced similar impacts, and a landfill leachate source could not be ruled out in that case.

The case studies illustrate how multiple lines of evidence were needed to assess suspected impacts and that no single constituent or parameter could be used alone to assess potential impacts.

7.4.2.4 Other Sources

In the EPA's *Retrospective Case Study in Northeastern Pennsylvania: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2014f), a pond was found to be impacted due to elevated chloride and TDS, along with strontium ratios ($^{87}Sr/^{86}Sr$) characteristic of Marcellus Shale produced water. Here, the suspected source of the impact was a well pad which had a hydrochloric acid spill, a possible produced water spill and been used for temporary storage of drill cuttings. The same mulidence fracturing impacts from constituents characteristic of produced water (TDS, chloride, sodium, barium, strontium and radium) found in three domestic wells located in an area with naturally occurring saline groundwater. Conversely, at a spring with organic chemical contamination but no associated chloride or TDS impacts, hydraulic fracturing activities were also ruled out.

An estimated 6,300 to 57,373 gal (24,000 to 217,280 L) of Marcellus Shale produced water was discharged through an open valve that drained a tank at XTO Energy Inc.'s Marquardt pad and flowed into a tributary of the Susquehanna River in November 2010 (U.S. EPA, 2016e; PA DEP, 2011c). Overland and subsurface flow of released fluids impacted surface water, a subsurface spring, and soil. Five hundred tons of contaminated soil were excavated, and an estimated 8,000 gal (30,000 L) of produced water was recovered (Science Applications International Corporation, 2010). Elevated levels of TDS, chloride, bromide, barium and strontium that indicated a release of produced water were present in the surface stream and a spring for roughly 65 days (U.S. EPA, 2016e). At that time the chloride concentration in the spring dropped below the state surface water standard of 250 mg/L. The impact extended a distance of approximately 1,400 ft (440 m) to the spring from the release point. Samples were taken in the tributary roughly 500 ft downstream from the spring, where chloride concentrations remained below the 250 mg/L standard throughout the sampling period, but were above the upstream concentrations (PA DEP, 2011c; Schmidley and Smith, 2011). Similarly, the total barium, total and dissolved iron, manganese and alkalinity concentrations remained below the Pennsylvania surface water quality standards at the downstream monitoring location throughout the monitoring period (Schmidley and Smith, 2011).

[1] DIC is dissolved inorganic carbon.
[2] The full list was: Br vs. B, Cl vs. Mg, Cl vs. Br, Cl vs. HCO_3,Cl vs. Ca, Cl vs. K, Cl vs. Na, Cl vs. SO_4, Cl/Br, Cl/I, K/Rb, $^{87}Sr/^{86}Sr$.

In Pennsylvania, discharges of brine were made into a storm drain that itself discharges to a tributary of the Mahoning River in Ohio. Analyses of the brine and drill cuttings that were discharged indicated the presence of contaminants, including benzene and toluene (U.S. Department of Justice, 2014). In California, an oil production company periodically discharged hydraulic fracturing wastewaters to an unlined sump for 12 days. It was concluded by the prosecution that the discharge posed a threat to groundwater quality (Bacher, 2013). These unauthorized discharges represent both documented and potential impacts on drinking water resources. However, data do not exist to evaluate whether such episodes are uncommon or whether they happen on a more frequent basis and remain largely undetected. Other cases of unpermitted discharges have been reported by various sources (Caniglia, 2014; Paterra, 2011).[1]

7.4.2.5 Data Compilation Studies

Three datasets were examined for produced water spill data. These included two published studies: a review of spills in Oklahoma that occurred prior to the onset of widespread high-volume hydraulic fracturing (Fisher and Sublette, 2005), and an EPA study of spills occurring between February 2006 and April 2012 on the well pads of hydraulically fractured wells (U.S. EPA, 2015m). The EPA spills study, *Review of state and industry spill data: characterization of hydraulic fracturing-related spills*, is described in Text Box 5-10. Because of data availability, EPA's study was dominated by data from Pennsylvania (21% of releases) and Colorado (48% of releases). Several difficulties are encountered in compiling and evaluating data on produced water spills and releases. Because states have differing minimum reporting levels, more spills are potentially reported in states with lower reporting limits.[2]

To include data from another state and to give results current to 2015, data from North Dakota were reviewed for this assessment.[3] Details on the procedures and results for non-produced water spills are given in Appendix Section E.5. The North Dakota Department of Health (NDDOH) collects data on environmental incidents and separately compiles oil field incidents; information is made available to the public at http://www.ndhealth.gov/EHS/Spills/. Of these incidents, most describe a release of oil, salt water, or other liquid. Of the remainder, a few describe releases of gas only.

For the period from November 2012 to November 2013, NDDOH reported 552 releases of produced water that were retained within the boundaries of the production or exploration facility and 104 that were not (North Dakota Department of Health, 2011). Thus, 16% of the releases were not contained within facility boundaries and had greater potential for impacting drinking water resources.

[1] Section 8.4 discusses permitted discharges of wastewater.

[2] For example, two agencies in the state of California manage different databases that both store information on spills associated with oil and gas production (CCST, 2015a). CCST (2015a) reported that the databases contain inconsistencies as to the number of spills and the details regarding those spills (e.g., quantity, chemical composition of the wastewater) resulting in uncertainty on the impacts spills have on the environment.

[3] Wirfs-Brock (2015) presented an analysis of North Dakota spill data through 2013.

7.4.2.6 Frequency of Spills and Releases

The EPA analyzed these data and found that, in recent years (2010-2015), there were between five and seven produced water spills per hundred active production wells (Figure 7-7). Spills declined between 2014 and 2015 (from 846 to 609), although the number of production wells increased. A study of 17 states indicated that there was an overall reduction of 8% in spills from 2014 to 2015, and an increase of 9% in Texas (King and Soraghan, 2016). More details on the data analysis are given in Appendix Section E.5, which includes results on North Dakota oil and spills of other types, including hydraulic fracturing fluids (as noted in Chapter 5).

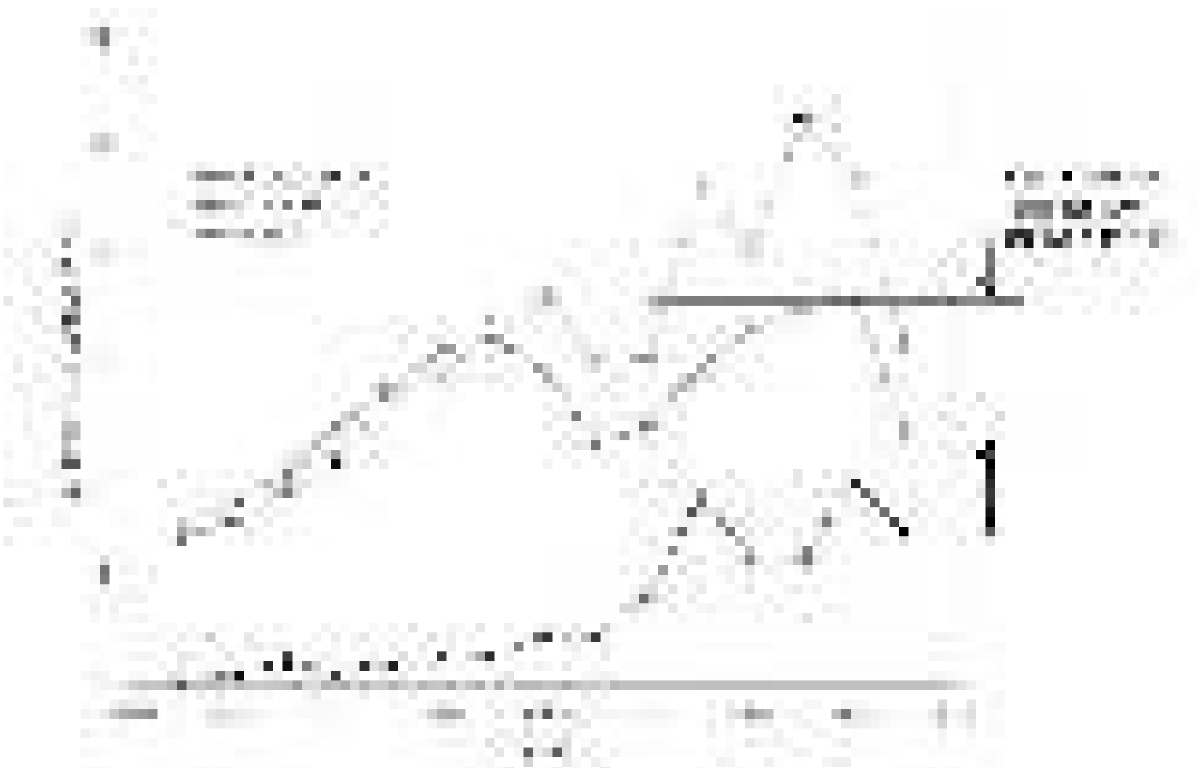


Figure 7-7. Produced water spill rates (spills per active wells) for North Dakota from 2001 to 2015 (Appendix Section E.5).

7.4.2.7 Produced Water Releases—Causes and Sources

The causes and sources identified for releases vary among the three datasets reviewed. North Dakota releases were dominated by leaks from various pieces of equipment, followed by "others," and various overflows (Figure 7-8). While the release rate declined from 2014 to 2015, the causes remained ranked relatively in the same order; notably fewer releases were attributed to "other" and more to equipment failure in 2015. The EPA's spills study found on- or near-well pad releases to be dominated by human error, unknown, and equipment failure (U.S. EPA, 2015m). The earlier

Oklahoma study was dominated by overflows, unpermitted discharges, and storms (Figure 7-9).[1] Storms can cause releases, as was noted after a major flood in northeastern Colorado that caused damage to produced water storage tanks releasing an estimated 43,000 gal (160,000 L) of produced water (COGCC, 2013).

The sources of releases are documented for the Oklahoma and EPA studies (Figure 7-10). The EPA cites storage, unknown, and hoses or lines as the major sources for its 225 well-pad releases. The earlier Oklahoma study cites unclassified, lines, and tanks as major sources of its 8,874 releases.

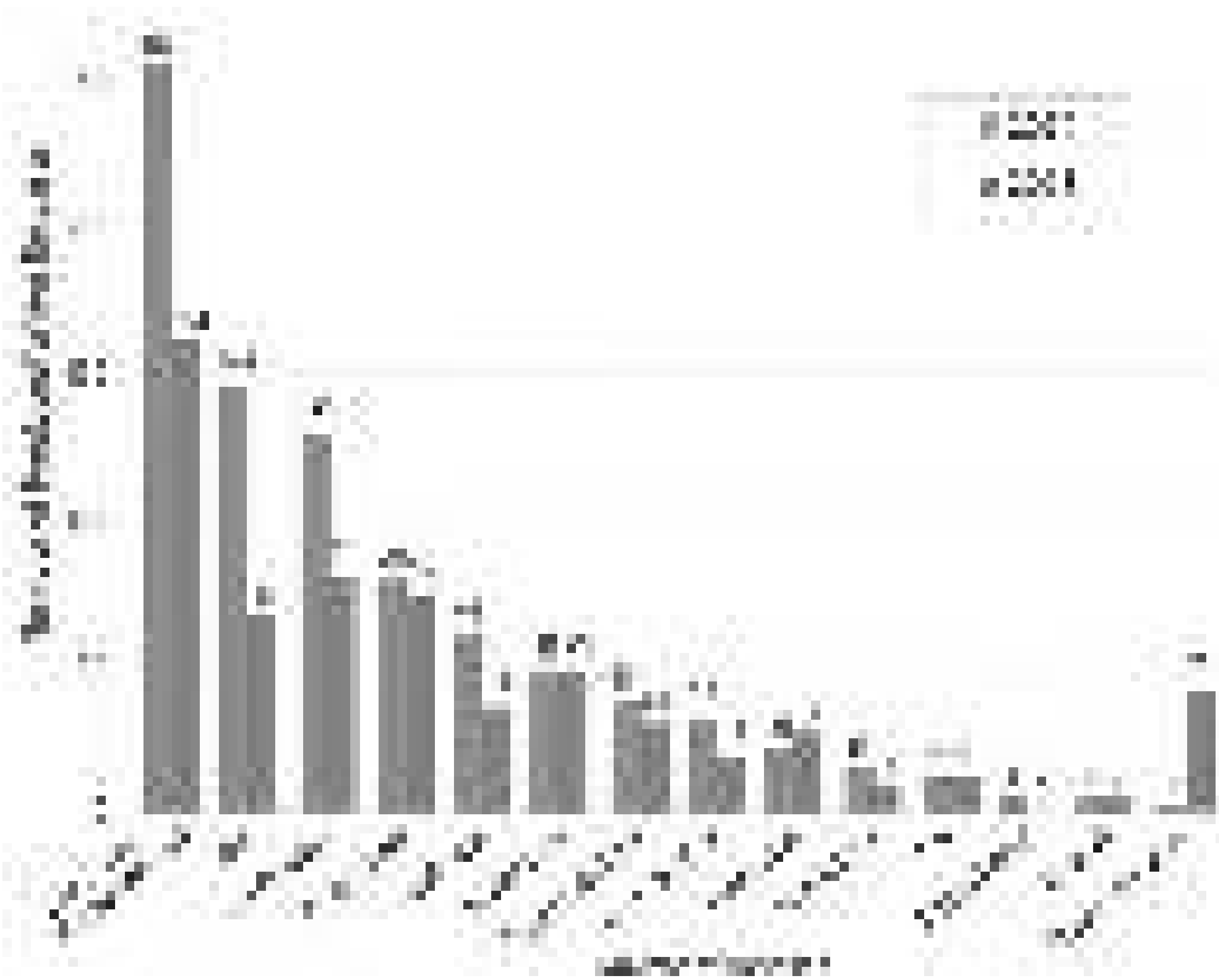


Figure 7-8. Number of produced water releases in North Dakota by cause for 2014 and 2015 (Appendix Section E.5).

[1] Some of the causes in the three studies may be more similar than they appear, because the categorization used in the different studies overlap. For example, the EPA categorized overflows as "human error;" blowouts, vandalism and weather as "other;" and corrosion as "equipment failure," while other studies listed these separately.

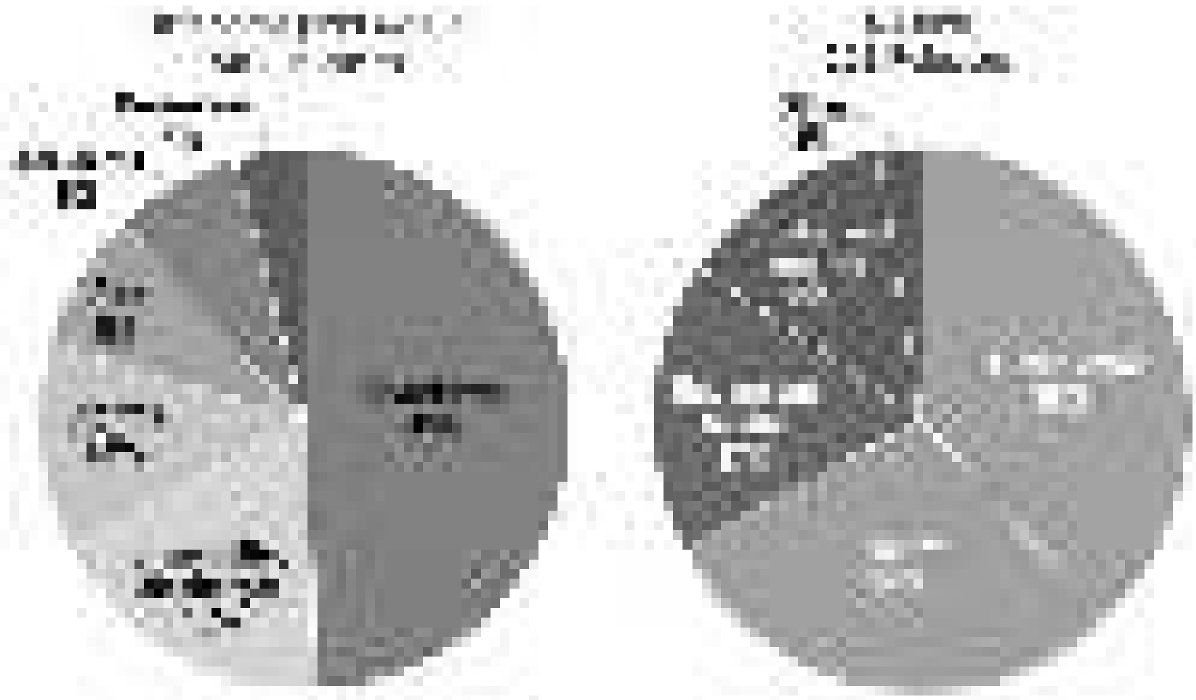


Figure 7-9. Distribution of spill causes in Oklahoma, pre-high volume hydraulic fracturing years of 1993-2003 (left) and in the EPA study of spills on production pads (right).
Data sources: left, Fisher and Sublette (2005); right, U.S. EPA (2015m).

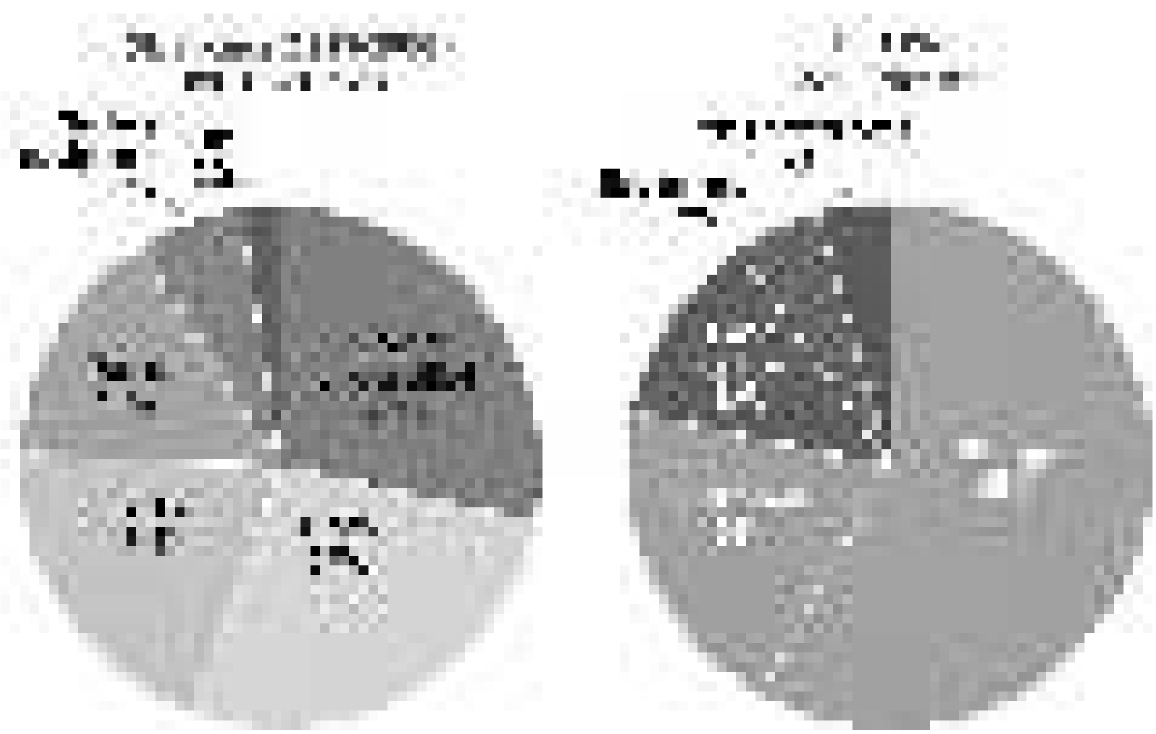


Figure 7-10. Distribution of spill sources in Oklahoma, pre-high volume hydraulic fracturing years of 1993-2003 (left) and in the EPA study of spills on production pads (right).
Data sources: left, Fisher and Sublette (2005); right, U.S. EPA (2015m).

7.4.2.8 The Volumes of Spilled Produced Water

The 2015 North Dakota spills were ranked from by the median volume, which is the level at which 50% of the spills are below this volume and 50% above (Figure 7-11).[1] Of the North Dakota spills in 2015, the highest median spill volume was caused by a blowout (2,400 gal, 91,000 L, left-most red box). The smallest median volume spill is approximately 10 times lower in volume (84 gal, 320 L). Spills larger than the median are of interest, because of their potential for impacting drinking water resources. The largest volume spill occurred from a pipeline break (2,900,000 gal, 11,000,000 L). The EPA spills study found the highest median volume spill was from equipment failure (1,700 gal, 6400 L), while the highest volume spill was due to container integrity (1,300,000 gal, 4,900,000 L) (Figure 7-12).

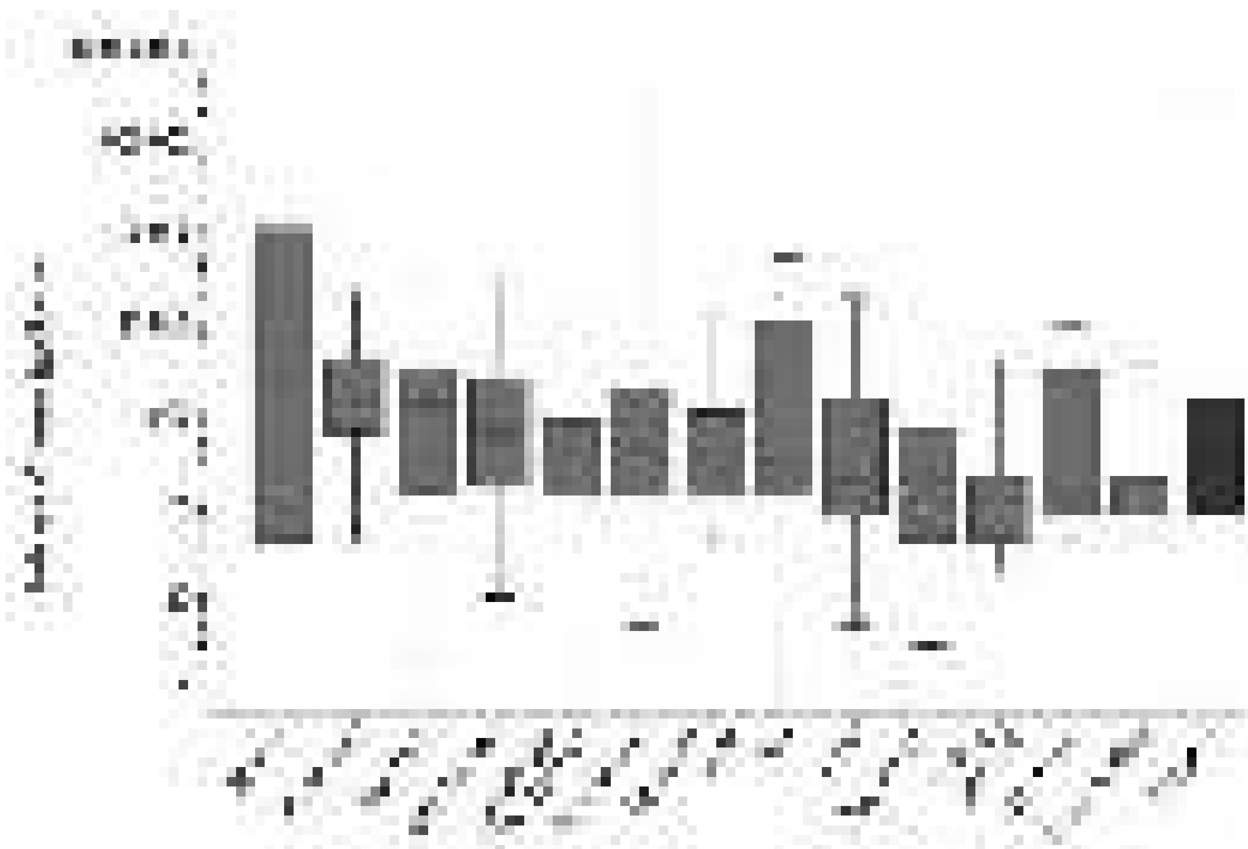


Figure 7-11. Volumes of 2015 North Dakota salt water releases by cause (leftmost 13 boxes in red), and all causes (last box in blue).

[1] These figures are called "box" plots or "box and whisker" plots. The rectangle in the middle represents the range of data from the 25th to 75th percentile. The line across the box represents the 50th percentile, also known as the median. Fifty percent of the data are below the median. The lines extending above and below the boxes represent the range of data from minimum to maximum. These concepts are illustrated in Appendix Figure E-6.

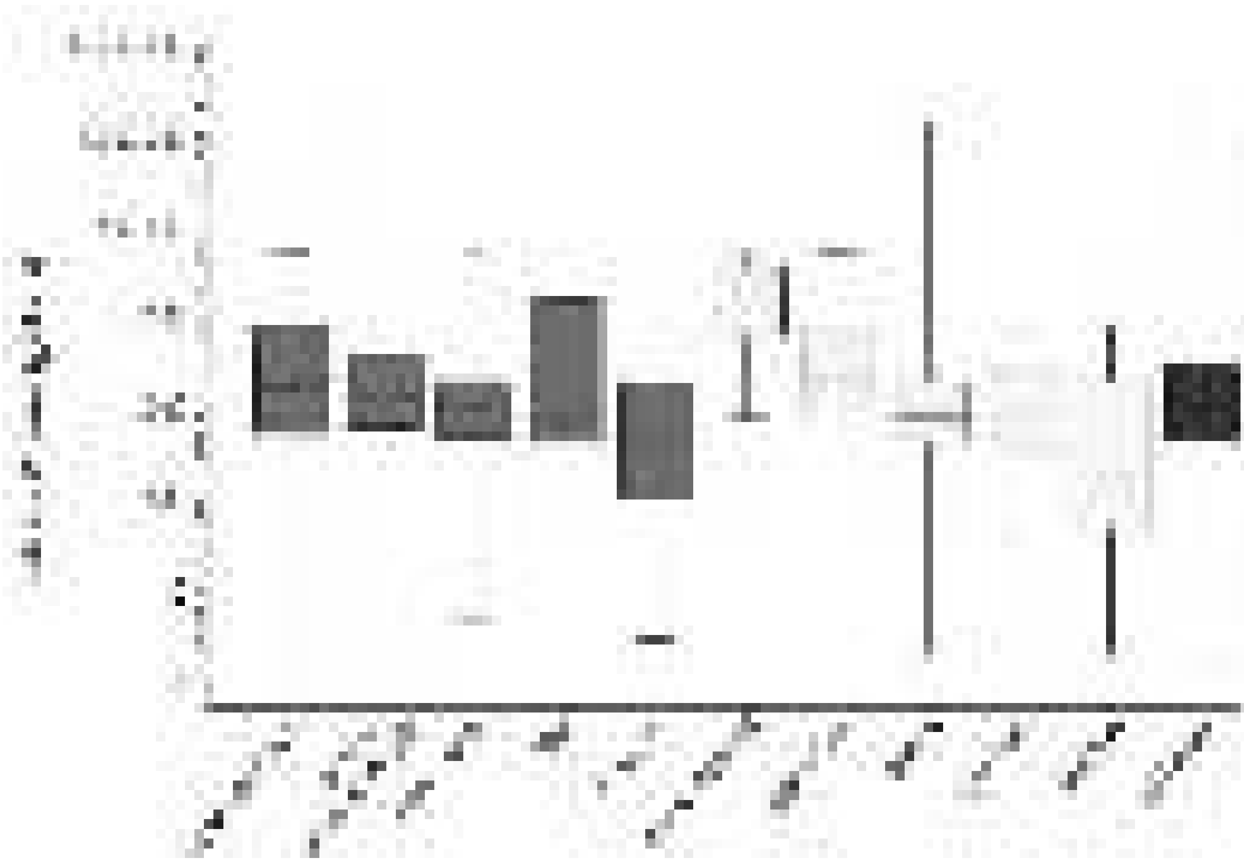


Figure 7-12. Volumes of produced water spills reported by the EPA for 2006 to 2012 by cause (the five left most boxes in red), source (the second five boxes in yellow), and all spills (blue).
Calculated from Appendix B of U.S. EPA (2015m).

From the analyses, half of the spills are less than 1,000 gal (3,800 L) (EPA) and 340 gal (1,300 L) (North Dakota) (Figure 7-12, Figure 7-13, and medians in Table 7-7). The medians for the Oklahoma study were higher (overall 1,700 gal or 6,400 L; see Table 7-7 for yearly values) (Fisher and Sublette, 2005). These occurred in a different state and over an earlier time period, so a direct connection with the recent North Dakota and EPA results has not been made.

The skewed nature of the distributions are noted by the mean values being considerably higher than these medians (see Figure 7-13). In each case, this is caused by a small number of large spills. For 2015 in North Dakota, for example, there were 12 releases of 21,000 gal (79,000 L) or more; 5 of 42,000 gal (160,000 L) or more; and one of greater than 420,000 gal (1,600,000 L) (Appendix Table E-15). The largest spills from these data sets ranged from 1,000,000 gal (3,800,000 L) to 2,900,000 gal (11,000,000 L).

The EPA results give insight into recovery and reuse. Of the volume of spilled produced water, 16% was recovered for on-site use or disposal, 76% was reported as unrecovered, and the rest was unknown. The fewest spills occurred from wells and wellheads, but these spills had the greatest median volumes. Failure of container integrity was responsible for 74% of the volume spilled (U.S. EPA, 2015m).

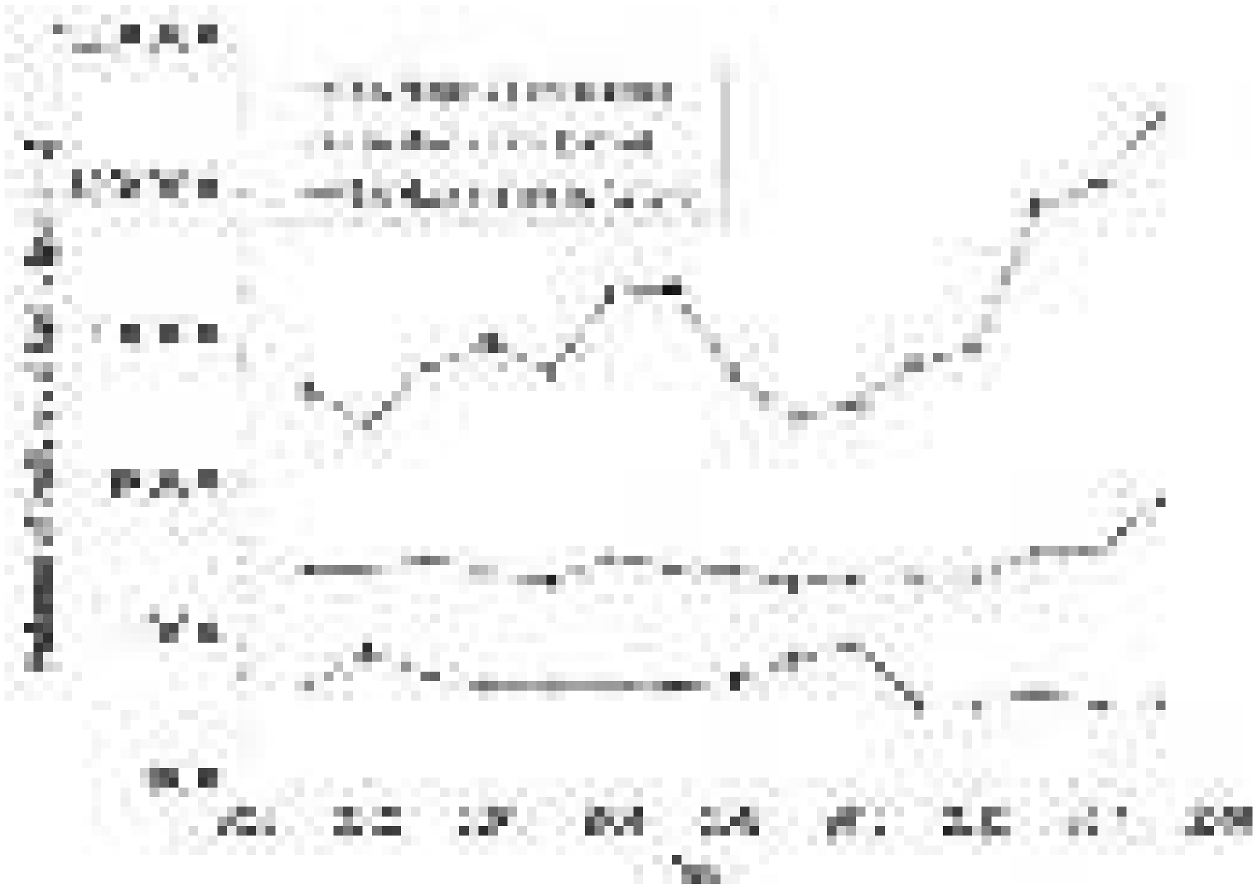


Figure 7-13. Median, mean, and maximum produced water spill volumes for North Dakota from 2001 to 2015.

Table 7-7. Summary of produced water release volumes.

Sources: U.S. EPA (2015m), Fisher and Sublette (2005), and Appendix Section E.5.

		Number		Minimum	25th percentile	Median	Mean	75th percentile	Maximum
Study	**Year(s)**	**Total**	**Quantified**	**(gal)**	**(gal)**	**(gal)**	**(gal)**	**(gal)**	**(gal)**
Oklahoma	1993-2002	7,916	2,365	0.0	630	1,700	7,000	4,200	3,400,000
	1993	373	161	0.4	420	1,500	3,900	4,200	46,000
	1994	844	333	0.4	420	1,600	5,400	4,200	84,000
	1995	913	333	0.0	420	1,500	3,700	4,200	63,000
	1996	880	333	4.2	630	2,100	6,500	4,200	420,000
	1997	806	270	0.4	630	1,900	6,000	4,200	120,000
	1998	825	236	2.1	798	4,900	2,100	4,200	105,000
	1999	886	218	10.5	840	2,100	6,600	4,200	120,000
	2000	853	155	4.2	840	2,100	5,600	5,040	210,000
	2001	826	144	21.0	840	2,100	31,000	6,510	3,400,000
	2002	710	182	0.8	630	1,700	5,500	3,276	130,000
U.S. EPA	2006-2012		225	2.1	420	1,008	10,920	2,982	1,344,000
North Dakota	2001		97	21.0	168	420	2,646	2,520	42,000
	2002		110	4.2	210	756	2,604	2,100	25,200
	2003		128	2.1	126	504	3,150	2,562	58,800
	2004		159	10.5	126	420	2,478	2,100	88,200
	2005		184	5.0	126	420	2,142	1,680	54,600
	2006		226	5.0	126	420	3,150	1,680	189,000
	2007		248	0.4	210	420	2,814	2,100	210,000

Study	Year(s)	Number		Minimum	25th percentile	Median	Mean	75th percentile	Maximum
		Total	Quantified	(gal)	(gal)	(gal)	(gal)	(gal)	(gal)
North Dakota, cont.	2008		248	8.4	84	504	2,520	2,058	54,600
	2009		208	2.1	126	630	2,100	2,100	27,300
	2010		255	0.1	126	840	2,478	2,310	34,020
	2011		381	2.1	126	336	2,436	1,680	58,800
	2012		543	7.1	84	336	2,310	1,260	84,000
	2013		700	2.1	126	378	3,402	1,428	714,000
	2014		846	0.8	84	336	3,528	1,470	1,008,000
	2015		609	0.8	84	336	7,560	1,386	2,940,000

7.4.2.9 Environmental Receptors and Transport

Data from the EPA (U.S. EPA, 2015m) were used to show that some spills were known to impact environmental receptors: soil (141 spills, 340,000 gal, or 1.3 million L); surface water (17 spills, 170,000 gal, or 640,000 L); surface water and soil (13 spills); and groundwater (1 spill, 130 gal, or 490 L).[1] Although 1 spill was identified as reaching groundwater, the possible groundwater impact of 107 of the spills was unknown.

In summary, 18 produced water spills reached surface water or groundwater, accounting for 8% of the 225 cases and accounting for approximately 170,000 gal (640,000 L) of produced water. Spills with known volumes that reached a surface water body ranged from less than 170 gal (640 L) to almost 74,000 gal (280,000 L), with median of 5,900 gal (22,000 L). In 30 cases, it is unknown whether a spill of produced water reached any environmental receptor.

An assessment conducted by the California Council on Science and Technology (CCST, 2015a) states that between January 2009 and December 2014, 575 produced water spills were reported to the California Office of Emergency Services of which nearly 18 percent impacted waterways (CCST, 2015a). These spills occurred in areas where production from both unconventional and conventional reservoirs occurs. Additional studies of spill impacts are presented in Appendix Section E.5.3.

Studies of Environmental Transport of Released Produced Water

The processes that affected the fate and transport of spilled produced water (Figure 7-14) are the same as those processes that impact the fate and transport of spilled chemicals (Section 5.8). Produced water spills differ from the chemical spills as they are always primarily spills of water containing multiple chemicals. Additionally, produced water of high salinity is denser than water and may alter transport and transformation properties of the chemicals and soils.[2] If a spill occurs prior to treatment in an oil and water separator, the produced water can be spilled along with oil. In the environment, oil is transported as a separate phase liquid as it is immiscible with water. The oil phase may become trapped (similarly to how oil is trapped in oil reservoirs) and serve as a slowly dissolving source of hydrocarbons to the environment.

For example, Whittemore (2007) described a site with relatively little infiltration due to moderate to low permeability of silty clay soil and low permeability of underlying shale units. Thus, most, but not all, of the historically surface-disposed produced water at the site flowed into surface drainages. Observed historic levels of chloride in receiving waters resulted from the relative balance of produced water releases and precipitation runoff, with higher concentrations corresponding to low stream flows. Persistent surface water chloride contamination was attributed to slow flushing and discharge of contaminated groundwater.

[1] Quoted volumes.

[2] Appendix Section E.7 describes the estimation of chemical properties for organic chemical constituents of produced water for baseline conditions of low TDS. Elevated salinity, as is common for produced water, would alter these values.

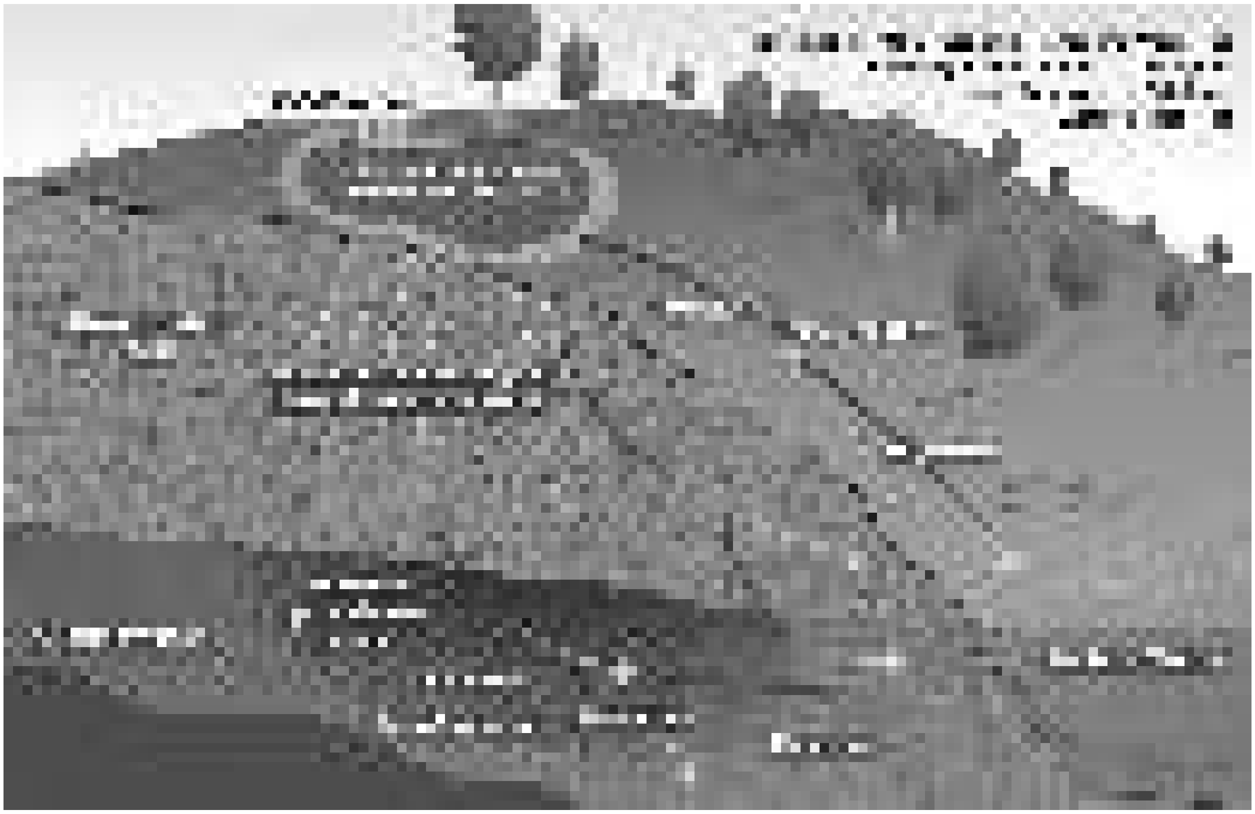


Figure 7-14. Schematic view of transport processes occurring during releases of produced water.

Because it is denser than freshwater, saline produced water can migrate downward through aquifers. Whittemore (2007) reported finding oilfield brine with a chloride concentration of 32,900 mg/L at the base of the High Plains aquifer. Where aquifers discharge to streams, saline stream water has been reported, although at reduced concentrations (Whittemore, 2007), likely due to diffusion within the aquifer and mixing with stream water. The stream flow rate, in part, determines mixing of substances in surface waters. High flows are related to lower chemical concentrations, and vice versa, as demonstrated for bromide in the Allegheny River (States et al., 2013).

7.5 Roadway Transport of Produced Water

Produced water is transported to treatment and disposal sites via pipeline, roadways, or railroad tankers. Accidents during transportation of hydraulic fracturing produced water are a possible mechanism leading to potential impacts to drinking water as truck-related releases have been reported. Nationwide data are not available, however, on the number of such accidents that result in impacts.

Crash rate estimates for Texas showed that commercial motor vehicle (CMV) crashes were correlated with oil and gas development activities over a recent period of increased oil and gas development (Quiroga and Tsapakis, 2015). As an example of the results, the number of new wells in the Permian Basin increased (by 61%) and so did rural CMV crashes (by 52%). For the Barnett Shale region, the number of new wells decreased (by 49%), and so did rural CMV crashes (by 34%). The correlations were strongest for the rural areas with oil and gas development (Permian and Eagle Ford).

Based on scenarios presented in Appendix Section E.8, the EPA estimated for this assessment the number of releases from truck crashes as having a chance of occurrence ranging between 1:110 and 1:13,000 over the lifetime of a producing well. The wide range of these estimates reflects both variable (distance and volume transported) and uncertain (crash rate) quantities. At 5,300 gal (20 m^3) per truckload, the volume from an individual spill would be low relative to the typical volume of water produced from a well. Several limitations are inherent in this analysis, including differing rural road and highway accident rates, differing transport distances, and differing amounts of produced water transported. Further, the estimates present an upper bound on impacts, because not all releases would reach or impact drinking water resources.

As for other types of impacts to drinking water resources, local effects can be significant despite the generally small numbers. For example, a brine-truck spill in Ohio resulted in concern for impacts to a drinking-water-source reservoir (Tucker, 2016).

7.6 Synthesis

Produced water is a by-product of oil and gas production and is that water that comes out of the well after hydraulic fracturing is completed and injection pressure is reduced. Produced water may contain hydraulic fracturing fluid, water from the surrounding formation, and naturally present hydrocarbons. Initially the chemistry of produced water reflects that of the hydraulic fracturing fluid. With time, the chemistry of the produced water becomes more similar to the water in the formation. Produced water is directly re-injected or stored at the surface for eventual reuse or disposal. Impacts to drinking water resources from produced water have been shown where spilled produced water entered surface water bodies or aquifers.

7.6.1 Summary of Findings

The volume and composition of produced water vary geographically, both within and among different production zones and with time and other site-specific factors. In most cases, there are high initial flow rates of produced water that last for a few weeks, followed by lower flow rates throughout the duration of gas production. The amount of fracturing fluid returned to the surface varies, and typically is less than 30%. In some formations (e.g., the Barnett Shale), the ultimate volume of produced water can exceed the volume of hydraulic fracturing fluid because of an inflow of water.

Knowledge of the composition of produced water comes from analysis of samples. Analysis of an individual sample is made much easier if the hydraulic fracturing and any equipment maintenance chemicals have been disclosed. Much of the chemical loading of produced water comes from naturally occurring material, both organic and inorganic, in the formation along with transformation products. As such, knowledge of produced water composition is uniquely dependent on sampling and analysis, which requires appropriate analytical methods. These are methods that can deal especially with high levels of TDS. Recently developed laboratory methods have greatly expanded the knowledge of organic chemicals in shale-gas and CBM produced waters, but these methods rely on advanced equipment and techniques. Routine methods of laboratory analysis do not detect many of the organic constituents of produced water.

The composition of produced water changes with time as the hydraulic fracturing fluid contacts the formation and mixes with the formation water. Typically it becomes more saline and more radioactive, if those constituents are present in the formation, while containing less DOC. The changing composition of produced water suggests that the potential concern for produced water spills also changes: initially the produced water may contain more hydraulic fracturing chemicals, later the concern may shift to the impact of high salinity water. Although varying within and between formations, shale and tight gas produced water typically contains high levels of TDS (salinity) and associated ionic constituents (bromide, calcium, chloride, iron, potassium, manganese, and sodium). Produced water can also contain toxic materials, including barium, cadmium, chromium, lead, mercury, nitrate, selenium, and BTEX. CBM produced water can have lower levels of salinity if its coal source was deposited under fresh water conditions, or if freshwater inflows to coal beds dilutes the formation water (Dahm et al., 2011). Many organic compounds have been identified in produced water. Most of these are naturally occurring constituents of petroleum. With the advent of advanced analytical techniques, more hydraulic fracturing fluid chemicals have been identified in produced water. These include some known tracer compounds, but others are known to exist whose identities have not yet been determined. Work has been done to identify environmentally benign tracers for assessing impacts, but these tracers have not been fully developed. Despite the presence in produced water of known hydraulic fracturing chemicals, the majority of organic and inorganic constituents of produced water come from the formation and cannot be minimized through actions of the operator. Throughout the formation-contact time, reactions occur between the constituents of the fracturing fluid and the formation.

Produced water spills have occurred across the country. From evaluation of data from across the United States and a focused study of North Dakota, the median produced water spill ranges from 336 to 1,000 gal (1,300 to 3,800 L). Although half of the spills are smaller than the median spill size, small numbers of much higher volume spills occur. In 2015, there were 12 spills in North Dakota greater than 21,000 gal (80,000 L), and one of 2,900,000 gal (11,000,000 L). From 2010 to 2015, there were approximately 5 to 7 produced water spills per hundred operating production wells. The major causes identified for these spills are container and equipment failures, human error, well communication, blowouts, pipeline leaks, and unpermitted discharges. Section 7.4.2 described impacts that were both of short and long term duration.

Highway transportation of produced water has resulted in crashes, but the impacts from these are unknown. Analysis of Texas crashes shows that as the oil and gas development activities increase, so do crashes, especially in rural areas. The EPA estimated the chance of a crash releasing produced water to range from 1:110 to 1:13,000.

7.6.2 Factors Affecting the Frequency or Severity of Impacts

The potential of spills of produced water to affect drinking water resources depends upon the release volume, duration, and composition, as well as watershed and water body characteristics. Larger spills of greater duration are more likely to reach a nearby drinking water resource than are smaller spills. Small releases, however, can impact resources where there are direct conduits from a source to receptor, such as fractures in rock. The composition of the spilled fluid also impacts the severity of a spill, as certain constituents are more likely to affect the quality of a drinking water resource.

Potential impacts to water resources from hydraulic fracturing related spills are expected to be affected by watershed and water body characteristics. For example, overland flow is affected by surface topography and surface cover. Infiltration of spilled produced water reduces the amount of water threatening surface water bodies. However, infiltration through soil can lead to groundwater impacts. Releases from pits can directly impact drinking water resources.

7.6.3 Uncertainties

The volume and some compositional aspects of produced water are known from published sources. The amount of hydraulic fracturing fluid returned to the surface is not well defined, because of the imprecise distinction between flowback and produced water. With regard to composition, TENORM and organics have the most limited data. Most of the available data on TENORM has come from the Marcellus Shale, where concentrations are typically high in comparison to the limited data available from other formations. Many organic constituents of produced water have been identified, and many of them are naturally occurring petroleum hydrocarbons. As methods improve and more data are collected, an increasing number of hydraulic fracturing fluid chemicals are being identified in produced water. Little is known concerning subsurface transformations and is reflected in only a few transformation products have been positively identified. Halogenation of organics has been noted, though.

Nationwide data on spills of produced water are limited in two primary ways: the completeness of reported data cannot be determined, and individual states' reporting requirements differ (U.S. EPA, 2015m). Therefore, the total number of spills occurring in the United States, their release volumes, and associated concentrations can only be estimated because of these underlying data limitations.

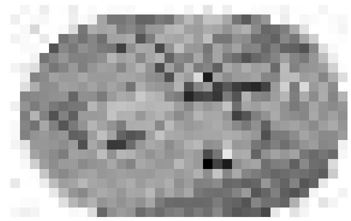
Spills vary in volume, duration, and composition, and most spill response focusses on immediate clean up, so several aspects of spills are not precisely characterized. The volume released is often a rough estimate, in part, because the spilled liquid spreads across the scene and is inherently difficult to measure. Simple measurements are often used to characterize the spill, rather than determining chemical concentrations (e.g., measuring electrical conductivity). As a consequence the suite of chemicals, and their concentrations, potentially impacting drinking water resources are usually unknown. Thus, the severity of impacts to drinking water resources is not usually well quantified.

Spills can originate from blowouts, well communication, aboveground or underground pipeline breaks, leaking pits, failed containers, human error (including unpermitted discharges, failure to detect spills, and failure to report spills) or unknown causes. The difference between these causes affects the location and size of the spill or release. For example, a container that fails may release a small amount of produced water, and be located on the well pad. A pipeline break may occur at a distance away from the well pad and release a larger amount of water from a bigger source (i.e., a pit). In addition, the factors governing transport of spilled fluid to a potential receptor vary by site: the presence and quality of secondary or emergency containment and spill response; the rate of overland flow and infiltration; the distance to a surface water body or drinking water well; and transport and fate processes. Impacts to drinking water resources from spills of produced water depend on environmental transport parameters, which can, in principle, be determined but are unlikely to be known or adequately specified in advance of a spill.

Because some constituents of produced water are constituents of natural waters (e.g., bromide in coastal surface waters) or can be released into the environment by other pollution events (e.g., benzene from gasoline releases, bromide from coal mine drainage), baseline sampling prior to impacts is one way to increase the certainty of an impact determination. Further sampling and investigation can be used to develop the linkage between a release and a documented drinking water impact. Appropriate sampling and analysis protocols, using quality assurance procedures, are essential for developing data that can withstand scrutiny. The EPA's northeastern Pennsylvania case study illustrates that the analytes that can be used to distinguish among types of water vary depending on the specifics of the situation. No single constituent or parameter could be used alone to assess impacts, and multiple lines of evidence were needed to assess the suspected impacts.

7.6.4 Conclusions

Produced water has the potential to affect the quality of drinking water resources if it enters into a surface water or groundwater body used as a drinking water resource. This can occur through spills at well pads or during transport of produced water. Specific impacts depend upon the spill itself, the environmental conditions surrounding the spill, water body and watershed characteristics, and the composition of the spilled fluid. The impacts from the majority of spills and releases is generally localized in extent as only the largest spills and releases impact large areas.



Chapter 8. Wastewater Disposal and Reuse

Abstract

This chapter addresses the practices and related impacts on drinking water resources that take place during the final stage of the hydraulic fracturing water cycle. This stage encompasses the management of wastewater, including disposal, reuse in hydraulic fracturing operations, or other uses. For this assessment, wastewater is defined as produced water from hydraulically fractured oil and gas wells that is managed by any of a number of strategies. The constituents of concern in hydraulic fracturing wastewaters that are most frequently noted include high total dissolved solids (TDS), chloride, bromide, and radionuclides (radium in particular). Other alkaline earth metals (e.g., barium), organics, and suspended solids, may be of concern as well.

Most hydraulic fracturing wastewater is managed by injection into Class II disposal wells. There are also "aboveground" management practices, which include reuse in subsequent hydraulic fracturing operations; treatment at a centralized waste treatment facility followed by reuse or discharge to surface water or a publicly owned treatment works; evaporation; irrigation; and direct discharge (under limited conditions). These practices can affect both surface water and groundwater.

Impacts on surface water arise from discharges of inadequately treated wastewater. In particular, bromide and iodide found in highly saline wastewaters can contribute to disinfection byproduct formation in downstream drinking water systems. If not removed during treatment, radium, metals, and organic compounds can also be discharged. Factors affecting the frequency and severity of impacts on surface waters include the wastewater's composition, its volume, and the processes used to treat it (common wastewater treatment processes do not significantly reduce the high TDS content in hydraulic fracturing wastewaters). In addition, site-specific factors such as local hydrology, size of the receiving water body, and other activities taking place in a watershed can affect the severity of the impact.

Pits and impoundments used for storage or disposal can impact surface water or groundwater through spills, leaks, and infiltration through soils. The frequency and severity of such impacts depend on pit construction and maintenance as well as proximity to drinking water resources. Unlined pits or those with compromised liners can cause long-lasting impacts on groundwater. Depth to the water table, soil properties, and the contaminants in the wastewater also affect the likelihood of impacts.

Characterizing the impacts from wastewater management associated with hydraulic fracturing is challenging given gaps in the data. Specifically, there are limited data on the wastewater volumes managed, on the influent and effluent concentrations and volumes from facilities that treat wastewater from hydraulic fracturing operations, and on wastewater residual characteristics and management of those residuals. Further, there is inadequate monitoring of drinking water resources for specific contaminants associated with hydraulic fracturing wastewater. However, the data that are available have shown that management of hydraulic fracturing wastewater through aboveground practices has affected the quality of water resources.

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8. Wastewater Disposal and Reuse

8.1 Introduction

The final stage of the hydraulic fracturing water cycle encompasses disposal and reuse of hydraulic fracturing wastewater. For the purposes of this assessment, "hydraulic fracturing wastewater" is defined as produced water from hydraulically fractured oil and gas wells that is being managed using practices that include, but are not limited to, reuse in subsequent hydraulic fracturing operations, treatment and discharge, and injection into disposal wells.[1,2,3] Although the term "wastewater" is generally used in this chapter, when more specific information about a wastewater is known (e.g., a source indicates the wastewater is flowback), that information is also noted.

Wells producing from oil and gas reservoirs generate produced water during the course of their productive lifespan. This produced water includes the often large volumes of flowback generated immediately after fracturing in deep wells with long horizontal sections. Flowback estimates vary by formation and are noted in Section 7.2.1 to range from about 300,000 to 10 million gal (1.14 to 37.8 million L) per well (Mantell, 2013; U.S. GAO, 2012). This large volume of initial flowback necessitates having a wastewater management strategy in place before hydraulic fracturing is initiated. Also, the longer-term generation of produced water requires ongoing wastewater management.

The majority of wastewater generated from all oil and gas operations in the United States is managed via Class II injection wells (Veil, 2015). Injection may be for either disposal or enhanced recovery. As hydraulic fracturing activity expands or diminishes, choices regarding disposal practices can change in a given region due to factors such as the quality and volume of the fluids; regulations; available infrastructure; and the feasibility and cost of treatment, reuse, and disposal options.

Several articles have noted potential effects of hydraulic fracturing wastewater on water resources (Vengosh et al., 2014; Olmstead et al., 2013; Rahm et al., 2013; States et al., 2013; Vidic et al., 2013; Rozell and Reaven, 2012; Entrekin et al., 2011). One study used probability modeling that indicated water pollution risk associated with gas extraction in the Marcellus Shale is highest for the wastewater disposal aspects of the operation (Rozell and Reaven, 2012). These concerns arise from

[1] The term "wastewater" is being used in this study as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes. This general description does not, and is not intended to, provide that the production, recovery, or recycling of oil, including the production, recovery, or recycling of flowback or produced water, constitutes "wastewater treatment" for the purposes of the Oil Pollution Prevention regulation (with the exception of dry gas operations), which includes the Spill Prevention, Control, and Countermeasure rule and the Facility Response Plan rule, 40 CFR 112 et seq.

[2] Disposal wells are Underground Injection Control (UIC) Class II wells, including those used for disposal (Class IID), enhanced oil recovery (Class IIR), and hydrocarbon storage (Class IIH).

[3] The term "reuse" is sometimes used to imply no treatment or basic treatment (e.g., media filtration) for the removal of constituents other than total dissolved solids (TDS), while "recycling" is sometimes used to convey more extensive treatment (e.g., reverse osmosis (RO)) to remove TDS (Slutz et al., 2012). In this document, the term "reuse" will be used to indicate use of wastewater for subsequent hydraulic fracturing, regardless of the level of treatment.

the elevated concentrations of chloride, bromide, radionuclides, and other constituents of concern found in many hydraulic fracturing wastewaters.

This chapter provides follow-on to Chapter 7, which discusses the per-well volumes of produced water (Section 7.2) and composition (Section 7.3), as well as the processes involved in its generation and impacts from a number of types of spills and releases. In this chapter, discussions are provided on management practices for hydraulic fracturing wastewater, available wastewater production information, and estimated aggregate volumes of wastewater generated for several states with active hydraulic fracturing (Section 8.2). As a complement to information on the composition of wastewaters in Chapter 7, Section 8.3 presents brief information on wastewater constituents and their relevance to wastewater management. Management methods used in recent years and their potential impacts on drinking water resources are described (Section 8.4). Based on background information provided in the earlier sections of the chapter, Section 8.5 discusses documented and potential impacts on drinking water resources from particular constituents, and a final synthesis discussion is provided (Section 8.6).[1]

8.2 Volumes of Hydraulic Fracturing Wastewater

This section provides a general overview of aggregate wastewater quantities generated in the course of hydraulic fracturing and subsequent oil and gas production, including estimates at regional and state levels. It also discusses methodologies used to produce these estimates and the associated challenges. (Chapter 7 provides a more in-depth discussion of the processes affecting produced water volumes and presents some typical per-well values and temporal patterns.) Wells also generate drilling fluid waste. Compared to produced water, however, drilling fluid wastewater can constitute a relatively small portion of the total wastewater produced (e.g., <10% in Pennsylvania during 2004-2013) (U.S. EPA, 2016d) and is not discussed further in this assessment.

Wastewater volume can be relevant to treatment costs, reuse options, and disposal capacities. IHS Global Insight suggests that as a general rule of thumb, the amount of flowback produced in the days or weeks after hydraulic fracturing is roughly comparable to the amount of produced water generated long-term over a span of years, which can vary considerably among wells (IHS, 2013). Thus, on a local level, operators can anticipate a relatively large volume of wastewater in the weeks following fracturing, with slower subsequent production of wastewater.

Wastewater volumes will most likely vary in the future as the amount and locations of hydraulic fracturing activities change and as existing wells age and move into the later phases of their production cycles. Substantial increases in wastewater production have occurred during times of increasing hydraulic fracturing activity. For instance, the average annual volume of wastewater

[1] This chapter makes use of background information collected by the EPA's Office of Water (OW) as part of the development of its recent pretreatment standards for wastewater from unconventional oil and gas formations (U.S. EPA, 2016d). The pretreatment standards apply to wastewater from crude oil and natural gas produced by a well drilled into shale and tight formations. Coalbed methane is beyond the scope of those standards. In this chapter, we consider wastewater generated by the hydraulic fracturing of those unconventional oil and gas formations included in the background research done by OW. But we also consider wastewater generated by hydraulic fracturing in coalbed methane and conventional formations.

generated by all gas production (both shale gas and conventional) in Pennsylvania quadrupled from the 2001-2006 period to the 2008-2011 period (Wilson and Vanbriesen, 2012).

However, although the total volume of wastewater might be expected to generally increase and decrease as oil and gas drilling and production changes, it is not necessarily a direct correlation. Data from the Pennsylvania Department of Environmental Protection (PA DEP) (PA DEP, 2016b) show trends in volumes of wastewater compared to gas produced from wells in the Marcellus Shale in Pennsylvania (Figure 8-1). Although the data show some variation, they demonstrate a general positive correlation between the volume of wastewater and the amount of produced gas until early 2015. At that time, Baker Hughes weekly rig counts also began to drop, declining from 85 in early January 2014 to 24 in early June 2016 (Baker Hughes, 2016). This suggests that a decline in overall drilling activity (generally a measure of new wells) can be associated with a decline in wastewater production, although the exact timing depends on whether there is a time delay between drilling and completion of a well and the start of production from that well.

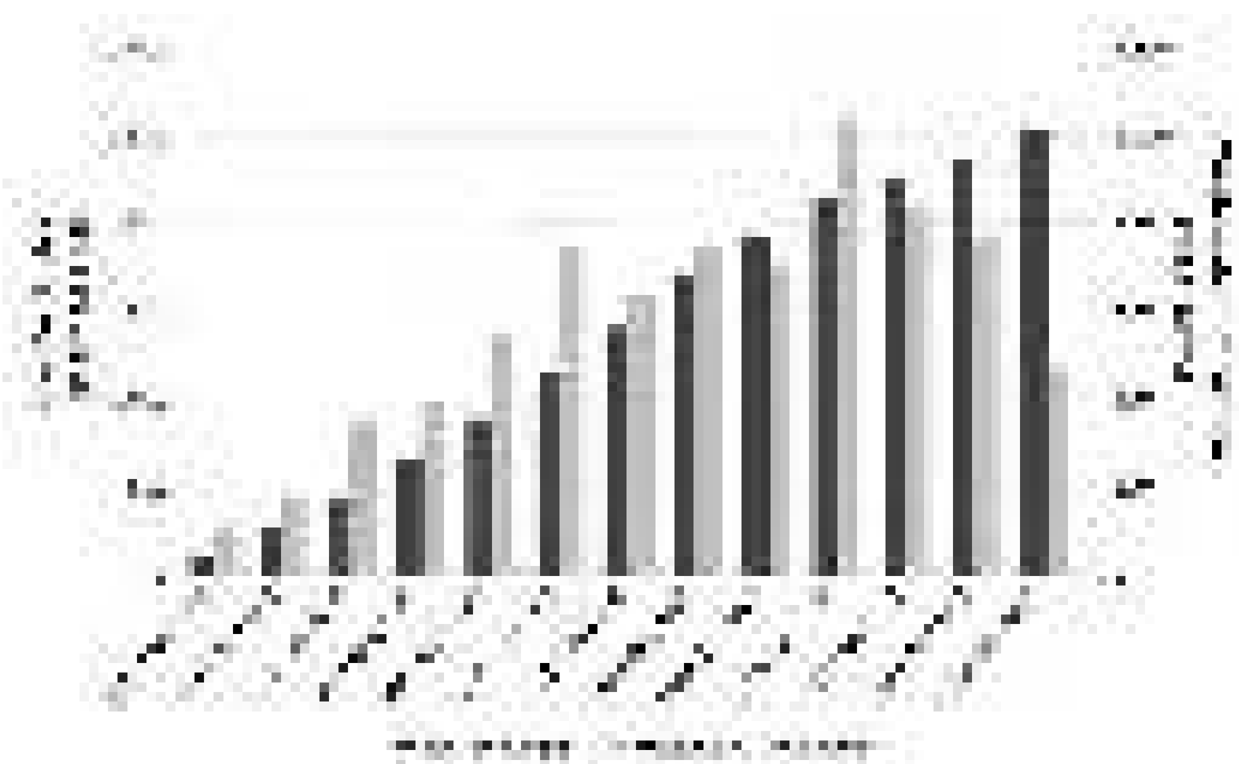

Figure 8-1. Wastewater (i.e., produced water and fracturing fluid waste) and produced gas volumes from unconventional (as defined by PA DEP) wells in Pennsylvania from January 2010 through June 2016.
Source: PA DEP (2016b).

Estimates of produced water compiled by Veil (2015) indicate that although oil and gas production in the United States increased by 29% and 22%, respectively, between 2007 and 2012, produced water volumes increased by less than 1%. There may be a number of factors contributing to this, including as noted by Veil (2015), a number of uncertainties associated with produced water estimates. First, wastewater generation varies from well to well, as do oil and gas production (see Chapter 7, Figure 7-1 for discussion of wastewater/produced water decline curves). The rates of decline in both wastewater volume and hydrocarbon production also vary among reservoirs. Additionally, some wells are drilled and completed but are not immediately put into production. Relationships between hydraulic fracturing activity, hydrocarbon production, and produced water volumes are likely reservoir- (and maybe production zone-) specific, and existing wells and production need to be considered to anticipate wastewater management needs.

8.2.1 National Level Estimate

Veil (2015) estimated that in 2012, U.S. onshore and offshore oil and gas production generated 889.59 billion gal (21.18 billion bbls) of produced water. This national-level estimate represents total oil and gas wastewater (from all oil and gas resources, and from wells hydraulically fractured and wells not hydraulically fractured). The estimate was compiled through a state-by-state analysis of survey data obtained from oil and gas agencies in the 31 states with active oil and gas production as well as the Department of Interior and U.S. EPA. However, Veil notes several issues with the data used for these estimates, including variability among states in data reporting, availability, and completeness. These issues may result in underestimation of the volumes of water produced (U.S. GAO, 2012). See Section 8.2.3 for more discussion on methods of estimating wastewater volumes.

8.2.2 Regional/State Level Estimates

A limited number of studies have described the geographic variability in oil and gas wastewater volumes. Veil (2015) reported that the top ten states nationwide for wastewater production in 2012 included Texas (35% of total), California (15% of total), Oklahoma (11% of total), and Wyoming (11% of total). A study by the Bureau of Land Management (BLM) (Guerra et al., 2011) states that in 2008, more than 80% of all oil and gas wastewater was generated in the western United States, with Texas producing the highest volume and Wyoming the second highest. The BLM report notes substantial wastewater from CBM wells, in particular those in the Powder River Basin (Wyoming). Figure 8-2 summarizes the wastewater volumes for these western states, demonstrating the wide variability from state to state (likely reflecting differences in the number of oil and gas production wells/production activity and reservoir geology). Although the authors do not identify all wastewater contributions from production involving hydraulic fracturing, the regions with established oil and gas production are likely to have methods and infrastructure available for management of hydraulic fracturing wastewater.

Figure 8-2. Wastewater quantities in the western United States (billions of gal per year).
Data from Guerra et al. (2011).

In the Marcellus region, waste data made public by the PA DEP have enabled analyses of wastewater volumes and trends since 2009. Estimates of produced water (including flowback or "fracing fluid waste" as well as "produced fluid") by Wunz (2015) and Shale Alliance for Energy Research Pennsylvania (SAFER PA, 2015) for 2014 are 1.73 and 1.64 billion gal (41.19 MMbl and 39.05 MMbl, respectively). The estimate compiled for this assessment is 0.65 billion gal (15.48 million bbls) for the first half of 2014 (Table 8-1). Variations among estimates reflect, among other factors, challenges in working with a dynamic database for which changes and corrections are ongoing.

Table 8-1 presents estimates of the volumes of hydraulic fracturing wastewater generated and the associated numbers of wells in North Dakota, Ohio, Pennsylvania, Texas (injected flowback only), and West Virginia. The data shown in this table were compiled for this assessment (except for West Virginia) and come primarily from records of produced water made publicly available on state websites.[1] These states are represented in Table 8-1 because the produced water volumes associated with hydraulic fracturing were readily identifiable. The data show that the increase in

[1] Data used for Table 8-1 were downloaded from state agency websites and compiled as needed (in either Microsoft Excel or Microsoft Access) for each state except West Virginia. Once compiled, data were filtered if needed and summed to produce estimates of wastewater production by year and a count of the numbers of wells generating the wastewater. Data were downloaded up through 2014. (Note that 2014 data for Pennsylvania and Texas are for partial years.) Differences in the years presented for the states are due to differences in data availability from the state agency databases. For West Virginia, data are from a report by Hansen et al. (2013) that compiled available flowback data from West Virginia.

the number of wells producing wastewater and the volumes of wastewater produced are generally consistent with the timing of the expansion of high-volume hydraulic fracturing and track with the increase in horizontal wells seen in Figure 3-20.

Several states with mature oil and gas industries (California, Colorado, New Mexico, Utah, and Wyoming) make produced water volumes publicly available by well as part of their oil and gas production data, but they do not directly indicate which wells have been hydraulically fractured. Some states (Colorado, Utah, Wyoming, and New Mexico) specify the producing formation or the basin along with volumes of hydrocarbons and produced water. Determining volumes of hydraulic fracturing wastewater for these states is challenging because there is a possibility of either inadvertently including wastewater from wells not hydraulically fractured or of missing volumes that should be included. This may be a particular problem where state terminology regarding what constitutes an unconventional resource or hydraulically fractured well is ambiguous or possibly different from other states. Appendix Table F-1 provides estimates of wastewater volumes in California, Colorado, New Mexico, Utah, and Wyoming in regions where hydraulic fracturing activity is taking place, along with notes on data limitations. The data in Table 8-1 and Appendix Table F-1 illustrate the challenges in both compiling a national estimate of hydraulic fracturing wastewater and comparing wastewater production among states due to dissimilar data types, presentation, and availability.

8.2.3 Estimation Methodologies and Challenges

Compiling and comparing data on wastewater production at the wide array of oil and gas locations in the United States presents challenges associated with data reporting and availability. Different approaches have been used to estimate state-specific and national wastewater volumes. Data from state agency websites and databases can be a ready source of information, whether publicly available and downloadable or provided directly by agencies upon request.

Veil (2015) notes that the reported volumes of produced water (e.g., reported by well in state production data) can be inaccurate or imprecise because produced water is not monitored continuously. Therefore, reported volumes may be estimates. Other issues such as data transcription errors or extrapolation of data can also affect reported volumes (Veil, 2015).

Using produced water volumes from state records to estimate the volume of wastewater regionally or nationally presents additional challenges due to a lack of consistency in data collection, availability, usability, completeness, and accuracy (Malone et al., 2015; Veil, 2015; U.S. GAO, 2012). Due to what are sometimes significant differences in the types of data collected and the mechanisms, formats, and definitions used, data cannot always be directly compared from state to state. This makes it difficult to aggregate volume data, and estimates may be better and more useful at a local or state level. Larger-scale estimates across regions or between states should be interpreted carefully.

Table 8-1. Estimated volumes (millions of gal) of wastewater based on state data for selected years and numbers of wells producing fluid.

State	Basin	Principal lithologies	Data type	2000	2004	2008	2010	2011	2012	2013	2014	Comments
North Dakota	Williston	Shale	Produced water	2	3	130	790	1,900	4,500	8,500	9,700	From North Dakota Oil and Gas Commission website[a]. Data provided for six members of the Bakken Shale. Produced water includes flowback (reports are submitted within 30 days of well completion.)
			Wells	*161*	*152*	*844*	*2,083*	*3,303*	*5,036*	*6,913*	*8,039*	
Ohio	Appalachian	Shale	Primarily flowback	-	-	-	-	3	29	110	-	Data from Ohio DNR Division of Oil and Gas[b]. Utica data for 2011 and 2012. Utica and Marcellus data for 2013. Brine is noted by agency to be mostly flowback.
			Wells	-	-	-	-	*9*	*86*	*400*	-	
Pennsylvania	Appalachian	Shale	Flowback plus produced water (% flowback; % produced water)	-	-	-	180 (51%; 49%)	740 (46%; 54%)	1,100 (36%; 64%)	1,300 (27%; 73%)	650 (32%; 68%)	Waste data from PA DEP[c]. Second half of 2010 and first half of 2014. Data described as unconventional as defined by PADEP. Separate codes are provided by PA DEP for flowback and produced water.
			Wells	-	-	-	*1,232*	*2,434*	*4,039*	*5,015*	*5,150*	

State	Basin	Principal lithologies	Data type	2000	2004	2008	2010	2011	2012	2013	2014	Comments
Texas	Unspecified (entire state)	Shale, Sandstone	Flowback - injected volumes	-	-	-	-	490	2,200	3,100	2,000	Waste injection data from Texas Railroad Commission[d]. Monthly totals are provided for entire state. Oct - Dec for 2011, full years for 2012 and 2013, and Jan - Oct for 2014
West Virginia	Appalachian	Shale	Flowback - Estimated total disposed	-	-	-	120	110	59	-	-	Estimates from Hansen et al. (2013).[e]

[a] North Dakota Industrial Commission. Department of Mineral Resources. Bakken Horizontal Wells By Producing Zone: https://www.dmr.nd.gov/oilgas/bakkenwells.asp.

[b] Ohio Department of Natural Resources, Division of Oil and Gas Resources. Oil and Gas Well Production. http://oilandgas.ohiodnr.gov/production#ARCH1.

[c] PA DEP Oil and Gas Reporting website, https://www.paoilandgasreporting.state.pa.us/publicreports/Modules/Welcome/Agreement.aspx.

[d] Railroad Commission of Texas, Injection Volume Query, http://webapps.rrc.state.tx.us/H10/searchVolume.do;jsessionid=J3cgVHhK9nkwPrC7ZcWNMgyzF9LCYyR1NmvDy3F1QQ5wqXfcGNGN!1841197795?fromMain=yes&sessionId=143075601021612. Texas state data provide an aggregate total amount of flowback injected for the past few years. (Data on brine volumes injected do not differentiate hydraulically fractured wells and, therefore, well counts are not presented here.) These values are interpreted as estimates of generated flowback as based on reported quantities of “fracture water flow back” injected into Class IID wells.

[e] West Virginia flowback estimates from Hansen et al. (2013) are based on state data. Well counts that are explicitly associated with the flowback and total disposed volumes were not available.

To compile estimates of the production and management of hydraulic fracturing wastewater, there are additional challenges. Reporting of wastewater volumes may or may not include information that helps determine whether the producing well was hydraulically fractured (e.g., an indicator of resource type or formation). It also might not be clear whether volumes listed as 'produced water' include the flowback component. Some states (e.g., Colorado and Pennsylvania) include information on disposal and management methods along with production data, and others do not.

Given the limitations of comparing state databases, some studies have generated estimates of wastewater volume using water-to-gas and water-to-oil ratios along with the reports of hydrocarbon production (Murray, 2013). The reliability of any wastewater estimates made using this method would need to be evaluated in terms of the quality, timeframe, and spatial coverage of the available data, as well as the extent of the area to which the estimates will be applied. Water-to-hydrocarbon ratios are empirical estimates. Because these ratios show a wide variation among formations, reliable data are needed to formulate a ratio in a particular region.

Another approach to estimating wastewater volumes would entail multiplying per-well estimates of produced water production rates by the numbers of wells in a given area. Challenges associated with this approach include obtaining accurate estimates of the number of new and existing wells, along with accurate estimates of per-well water production both during the flowback period and during the production phase of the well's lifecycle. In particular, it can be challenging to correctly match per-well wastewater production estimates, which will vary by formation, with counts of wells, which may or may not be clearly associated with specific formations. Temporal variability in wastewater generation would also be difficult to capture and would add to uncertainty. Such an approach, however, may be attempted for order of magnitude estimates if the necessary data are available and reliable.

8.3 Wastewater Characteristics

Along with wastewater volumes, wastewater characteristics and the characteristics of residuals produced during treatment or storage are important for understanding the potential impacts of management and disposal of hydraulic fracturing wastewater on drinking water resources. This section provides brief highlights on several important constituents known to exist in hydraulic fracturing wastewaters and residuals. Chapter 7 provides more in-depth detail on the chemistry of produced water, and Chapter 9 discusses reference values and health effects associated with hydraulic fracturing wastewater constituents.

8.3.1 Wastewater

Wastewater composition is the result of naturally-occurring constituents originating in the formation solids and fluids as well as chemicals associated with the fracturing fluid. Discussion in this chapter focuses on constituents in hydraulic fracturing wastewater for which adequate information is available to assess documented and potential impacts on drinking water resources. There may also be unknown constituents in wastewaters for which analyses have not been performed. This is due, in part, to a lack of information on fracturing fluid ingredients identified as confidential business information (CBI). In addition, there are uncertainties about how fracturing fluid ingredients are degraded or removed in the subsurface. (See Chapter 5, Section 5.8 for a discussion of processes that can cause chemicals to degrade or transform in the subsurface.)

8.3.1.1 Total Dissolved Solids and Inorganics

Hydraulic fracturing wastewaters are generally high in total dissolved solids (TDS), especially those from shales and tight formations, with TDS values ranging from less than 1,000 mg/L to hundreds of thousands of mg/L (Section 7.3.4.4). The TDS in wastewaters from shale formations is typically dominated by sodium and chloride and may also include elevated concentrations of bromide, bicarbonate, sulfate, calcium, magnesium, barium, boron, strontium, radium, organics, and heavy metals (Chapman et al., 2012; Rowan et al., 2011; Blauch et al., 2009; Orem et al., 2007; Sirivedhin and Dallbauman, 2004).

Within each formation, the minimum and maximum values presented in Section 7.3.4.4 suggest spatial variation in TDS content that may need to be accommodated when considering management strategies such as reuse or treatment. In contrast to shales and sandstones, TDS values for wastewater from CBM formations are generally lower, with reported concentrations ranging from approximately 150 mg/L to 62,000 mg/L (DOE, 2014b; Dahm et al., 2011) (Appendix Table E-3). This results in fewer treatment challenges and a wider array of management options.

Constituents commonly found in TDS from hydraulic fracturing wastewaters may have potential health impacts or create treatment burdens on downstream drinking water systems if discharged at high concentrations to drinking water resources. Bromide, for example, can contribute to the increased formation of disinfection byproducts (DBPs) during drinking water treatment (Hammer and VanBriesen, 2012); see Section 8.5.1.

8.3.1.2 Organics

Less information is generally available about organic constituents in hydraulic fracturing wastewaters than about inorganic constituents, but there are now several studies reporting analyses of organic constituents (Chapter 7). The organic content in flowback waters can vary based on the chemical additives (e.g., biocides, antiscalants, gelling agents, breakers) used in hydraulic fracturing fluids and the chemistry of the formation, but the organics generally include polymers, oil and grease, volatile organic compounds (VOCs), and semi-volatile organic compounds (SVOCs) (Akob et al., 2016; Walsh, 2013; Hayes, 2009). Examples of other constituents detected include alcohols, naphthalene, acetone, and carbon disulfide, compounds that may be remnants of hydraulic fracturing fluid chemicals (Hayes and Severin, 2012b; Hayes, 2009) (Appendix E). Wastewater associated with CBM wells may have high concentrations of aromatic and halogenated organic contaminants potentially requiring treatment depending on how the wastewater will be managed (Pashin et al., 2014; Sirivedhin and Dallbauman, 2004). Concentrations of BTEX (benzene, toluene, ethylbenzene, and xylenes) in CBM produced waters are lower than in shale produced waters (Appendix Table E-9).

New research is focusing on transformation products generated in the subsurface; experimental work by Kahrilas et al. (2015) suggests that the biocide glutaraldehyde can be present in wastewaters along with its transformation products. Low molecular weight organic acids such as acetate, formate, and pyruvate have been detected in Marcellus wastewater, indicating microbial degradation of organic compounds in the fracturing fluid or formation (Akob et al., 2015).

8.3.1.3 Radionuclides

Radionuclides are constituents of concern in some hydraulic fracturing wastewaters, with most of the available data obtained for the Marcellus Shale in Pennsylvania (Appendix Table E-8). Results from a United States Geological Survey (USGS) report (Rowan et al., 2011) indicate that the predominant radionuclides in Marcellus Shale wastewater are radium-226 and radium-228. Radionuclides in produced fluids are considered ‘technologically enhanced naturally-occurring radioactive material’ (TENORM) because they have been exposed to the accessible environment.[1]

Although data regarding radionuclides in wastewater from formations other than the Marcellus Shale are limited, there is information on the naturally occurring radioactive material (NORM) in the formations themselves.[2] In particular uranium and thorium can be found in certain organic-rich black shales. High uranium content has been measured in the Marcellus, Barnett, Woodford, and other black shales (Swanson, 1955) (Section 7.3.4.6). Radium-226 and -228 are decay products of uranium and thorium and are soluble (Sturchio et al., 2001; Langmuir and Riese, 1985). Therefore wastewater from shales with high concentrations of uranium and thorium can contain radium, especially where TDS concentrations are also high (Rowan et al., 2011; Sturchio et al., 2001; Fisher, 1998). Section 7.3.3.2 provides further information on radionuclides in produced waters and in formations.

8.3.2 Constituents in Residuals

Depending on the wastewater and the treatment processes used, treatment residuals can consist of sludges, spent media (used filter materials), or brines. Residuals may require further treatment (e.g., dewatering sludges) prior to disposal (see Section 8.4.7 for further discussion on management of residuals). Residuals can contain constituents such as total suspended solids (TSS), TDS, metals, radionuclides, and organics. These constituents will be concentrated in the residuals, with the degree of concentration depending on the type of treatment employed. Processes such as electrodialysis and mechanical vapor recompression have been found to yield residuals with TDS concentrations in excess of 150,000 mg/L after treating waters with influent TDS concentrations of approximately 50,000 – 70,000 mg/L (Hayes et al., 2014; Peraki and Ghazanfari, 2014).

Also, TENORM in wastewaters can cause residual wastes to have gamma radiation emissions (Kappel et al., 2013). A laboratory study by Zhang et al. (2014b) estimated that the barium sulfate solids precipitated during treatment to remove barium and strontium from Marcellus Shale wastewater would also contain between 2,571 and 18,087 pCi/g of radium due to coprecipitation. Another similar study using mass balances calculated that sludge from a sulfate precipitation

[1] Technologically Enhanced Naturally Occurring Radioactive Material (TENORM) is defined by the EPA as naturally occurring radioactive materials (NORM) that have been concentrated or exposed to the accessible environment as a result of human activities such as manufacturing, mineral extraction, or water processing.

[2] Naturally Occurring Radioactive Materials (NORM) are radioactive materials found in nature that have not been moved or concentrated by human activities.

process would have an average radium concentration of 213 pCi/g (Silva et al., 2012). In sludge from lime softening processes, Silva et al. (2012) estimated a radium-226 concentration of 58 pCi/g, a level that would necessitate disposal as a low-level radioactive waste.

8.4 Wastewater Management Practices and Their Potential Impacts on Drinking Water Resources

Operators have several strategies for management of hydraulic fracturing wastewaters (Figure 8-3), with the most common choice being disposal via Class IID wells (Veil, 2015; Clark et al., 2013; Hammer and VanBriesen, 2012). Other practices include reuse in subsequent hydraulic fracturing operations (with varying levels of treatment), treatment at a centralized waste treatment facility (CWT) (often followed by reuse), evaporation (in arid regions), irrigation (with no discharge to waters of the United States), and direct discharge for livestock or agricultural use (allowed west of the 98th meridian). Up until 2011, treatment of unconventional oil and gas wastewaters (as defined by PA DEP) at publicly owned treatment works (POTWs) was a common practice for wastewater management in the Marcellus region (Lutz et al., 2013); this is discussed further in Text Box 8-1.

The methods shown in Figure 8-3 represent wastewater management strategies, not all of which would be used at the same facility. Descriptions of incidents of unpermitted disposal and resulting legal actions have also been publicly reported (Chapter 7). However, such events are not generally described in the scientific literature, and the prevalence of this type of activity is unclear. Additional sources of information about potential impacts exist, but some records are sealed (e.g., litigation records) and are not publicly accessible.

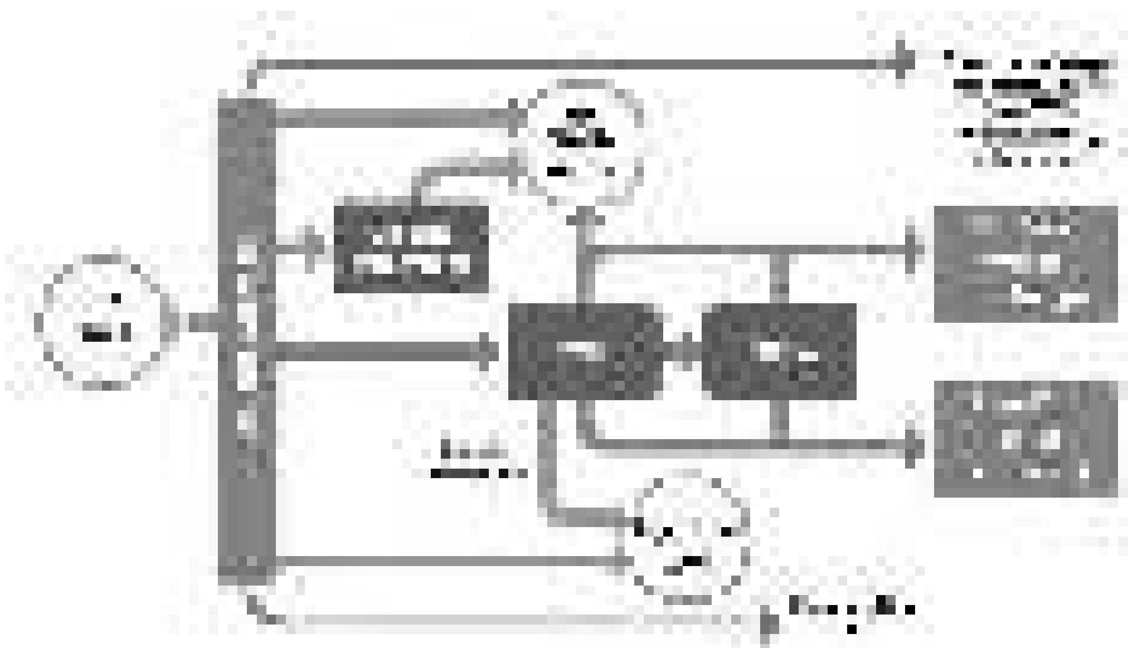

Figure 8-3. Schematic of wastewater management strategies.
Gray lines indicate management strategies that involve injection, either for reuse or disposal, and blue lines indicate management strategies that lead to other end points such as discharge, evaporation, landfills, or other uses.

Each of the wastewater management strategies can potentially lead to impacts on drinking water resources during some phase of their execution. Such impacts include, but are not limited to: accidental releases during transport (Chapter 7); discharges of treated wastewaters from CWTs or POTWs where treatment for certain constituents has been inadequate; migration of constituents to groundwater due to leakage from pits or land application of wastewaters; leakage from pits that reach surface waters (Chapter 7, Section 8.4.5); inappropriate management of liquid or solid residuals (e.g., leaching from landfills); or accumulation of constituents in sediments near outfalls of CWTs or POTWs that are treating or have treated hydraulic fracturing wastewater.[1]

A reliable census of oil and gas wastewater management practices nationwide is difficult to assemble due to a lack of consistent and comparable data among states. In addition, we do not know how often operators use more than one wastewater management strategy at a site (e.g., evaporation and injection), further complicating the tracking of wastewater management. As part of a data survey conducted by Veil (2015), some state agencies provided estimates of oil and gas wastewater volumes handled by several management practices (Table 8-2). These estimates illustrate how widespread injection for both enhanced recovery and for disposal is for managing oil and gas wastewater. The data also show regional differences in reuse and other practices. For hydraulic fracturing wastewaters, Table 8-3 illustrates nationwide variability in the primary wastewater management methods using qualitative and quantitative sources. Similar to Table 8-2, Table 8-3 shows disposal via underground injection predominates in most regions, and reuse is predominant in the Marcellus Region. (Table 8-3 does not include wastewater management in areas of CBM production.)

Management choices are affected by cost and a number of directly and indirectly related factors, including the chemical properties of the wastewater; the volume, duration, and flow rate of the wastewater generated; the feasibility of each option; the availability of necessary infrastructure; local, state, and federal regulations (Text Box 8-2); and operator discretion (U.S. GAO, 2012; NPC, 2011a). The economics (such as transport, storage, and disposal costs) and availability of treatment and disposal methods are of primary importance (U.S. GAO, 2012). For wastewater composition, there is limited information on the degradation or removal of fracturing fluid ingredients in the subsurface. Chemical disclosure requirements vary among states, and some fracturing fluid ingredients are claimed to be CBI. Therefore, the possible presence of unknown chemical constituents in wastewater contributes to uncertainty about the effectiveness and potential impacts of management strategies, particularly with regard to treatment efficacy.

[1] The term surface water as used in this assessment refers to surface waters that meet the definition of waters of the United States under the CWA (House Bill No. 1950, 2011).

Table 8-2. Estimated percentages of wastewater managed by practice and by state.
Source: Veil (2015). Estimates do not identify interstate transport (e.g., wastewater transported from PA to OH or WV for injection into disposal wells). Thus, there may have been some double counting of volumes in both the generating and receiving states.

Management practice	Percentage of produced water managed by practice and state											
	AR	CA	CO	NM	ND	OH	OK	PA	TX	UT	WV	WY
Injection for enhanced oil recovery	22	46	32	50 [d]	18	4.0	47	0	48	40	27	73
Injection for disposal	76	20	32	50 [d]	56	91	47	12	37	47	25	27
Surface discharge	0	2	10	no data	0	0	0	2.3	5.0 [f]	6	0	uncertain
Evaporation	0	21	9.0	no data	0	0		0	0	0	0	uncertain
Offsite commercial disposal	0.1 [a]	9	5.7 [c]	no data	26	Included in injection for disposal	6.0 [e]	0	10 [e]	7 [g]	28 [h]	uncertain
Beneficial reuse	1.1 [b]	no data	12 [b]	no data	0	5.0	0	85 (includes reuse for HF)	Est. 15-20 (flowback fluid)	0.5	uncertain	uncertain

[a] Land farm.

[b] Reuse for HF.

[c] Pits.

[d] Assumes even split with injection for enhanced oil recovery and injection for disposal.

[e] Injection.

[f] Fresh produced water.

[g] Evaporation ponds.

[h] Disposal wells.

Table 8-3. Management practices for wastewater from unconventional oil and gas resources.
Source: U.S. EPA (2016d).

Basin	Formation	Resource type	Reuse	Injection for disposal	CWT facilities	Notes	Available data[b]
Michigan	Antrim	Shale gas		XXX			Qualitative
Appalachian	Marcellus/Utica (PA)	Shale gas	XXX	XX	XX	Limited disposal wells in east	Quantitative
	Marcellus/Utica (WV)	Shale gas/oil	XXX	XX	X		Quantitative
	Marcellus/Utica (OH)	Shale gas/oil	XX	XXX	X		Mixed
Anadarko	Granite Wash	Tight gas	XX	XXX	X[a]		Mixed
	Mississippi Lime	Tight oil	X	XXX		Reuse/recycling limited but is being evaluated	Qualitative
	Woodford, Cana, Caney	Shale gas/oil	X	XXX	X[a]		Qualitative
Arkoma	Fayetteville	Shale gas	XX	XX	X[a]	Few existing disposal wells; new CWT facilities are under construction	Mixed
Fort Worth	Barnett	Shale gas	X	XXX	X[a]	Reuse/recycle not typically used due to high TDS early in flowback and abundance of disposal wells	Mixed
Permian	Avalon/Bone Springs, Wolfcamp, Spraberry	Shale/tight oil/gas	X	XXX	X[a]		Mixed
TX-LA-MS Salt	Haynesville	Tight gas	X	XXX		Reuse/recycle not typically used due to high TDS early in flowback and abundance of disposal wells	Mixed
West Gulf	Eagle Ford, Pearsall	Shale gas/oil	X	XXX	X		Mixed
Denver Julesburg	Niobrara	Shale gas/oil	X	XXX	X		Mixed
Piceance; Green River	Mesaverde/Lance	Tight gas	X	XX	X	Also managed through evaporation to atmosphere in ponds in this region	Qualitative

Basin	Formation	Resource type	Reuse	Injection for disposal	CWT facilities	Notes	Available data[b]
Williston	Bakken	Shale oil	X	XXX		Reuse/recycling limited but is being evaluated	Mixed

[a] CWT facilities identified in these formations are all operator-owned.

[b] This column indicates the type of data the EPA based the number of Xs on. In most cases, the EPA used a mixture of qualitative and quantitative data sources along with engineering judgment to determine the number of Xs.

XXX—The majority (≥50%) of wastewater is managed with this management practice; XX—A moderate portion (≥10% and <50%) of wastewater is managed with this management practice; X—This management practice has been documented in this location, but for a small (<10%) or unknown percent of wastewater. Blanks indicate the management practices have not been documented in the given location.

The availability and use of wastewater management strategies in a region can change over time as oil and gas production increases or decreases, regulations change, costs shift, and technologies evolve. Text Box 8-1 and Figure 8-4 illustrate shifting wastewater management practices in Pennsylvania as gas development in the Marcellus Shale increased and concerns over high-TDS discharges prompted a regulatory response. Reuse has increased substantially at well sites in Pennsylvania (labeled as "Reuse HF" in Figure 8-4) and wastewater management at CWTs has moved toward more facilities that provide wastewater for reuse and do not discharge (termed "zero-discharge facilities"). The estimated total reuse rate in Pennsylvania was 80% in 2012 and 90% in 2013 (PA DEP, 2015a). In contrast, wastewater disposal data in areas of Colorado where hydraulic fracturing takes place show a steady use of injection wells, an increase in surface water discharges, and a decrease in the use of on-site pits for evaporation since 2000 (Figure 8-5).

Another factor influencing reuse is the pace of hydraulic fracturing in the area. When hydraulic fracturing is active, demand for reuse is high. Some formations that are hydraulically fractured such as the Marcellus Shale and the Utica Shale are still in the early stages of development, with large potential resources not yet developed. For these plays, the need for wastewater treatment and/or reuse may remain high for decades to come, and the long-term wastewater management needs must be considered and addressed (SAFER PA, 2015).[1]

Researchers have developed optimization models to aid in the minimization of wastewater management costs as a part of comprehensive water management planning. For example, Yang et al. (2014) suggest an approach for reusing flowback in scheduled hydraulic fracturing events to minimize the operational costs of transportation, treatment, storage, and wastewater disposal. Another modeling study proposes an approach to minimize the total cost of water usage and wastewater treatment and disposal by optimizing capital costs (such as the costs of treatment units and storage pits) and operating costs for flowback management, treatment, storage, reuse, and wastewater disposal (Lira-Barragan et al., 2016).

Text Box 8-1. Temporal Trends in Wastewater Management – Experience of Pennsylvania.

Gross natural gas withdrawals from shale formations in the United States increased 518% between 2007 and 2012 (EIA, 2014b). This production increase has led to larger volumes of wastewater requiring appropriate management (Vidic et al., 2013; Gregory et al., 2011; Kargbo et al., 2010). The rapid increase in wastewater generated from hydraulically fractured oil and gas wells has led to many changes in wastewater disposal practices in the oil and gas industry. Changes have been most evident in Pennsylvania, which has experienced a more than 1,400% increase in natural gas production since 2000 (EIA, 2014b).

Lutz et al. (2013) estimated that total wastewater generation in the Marcellus region increased 570% between 2004 and 2013. The authors concluded that this increase has created stress on the existing wastewater disposal infrastructure. In 2010, concerns arose over elevated TDS in the Monongahela River

(*Text Box 8-1 is continued on the following page.)*

[1] As noted in Chapter 3, oil and gas prices influence new drilling activity. However, the links between oil and gas prices and the generation of wastewater (as a byproduct of production) appear to be less direct.

Text Box 8-1 (continued). Temporal Trends in Wastewater Management – Experience of Pennsylvania

basin, and studies linked high TDS (and, in particular, high bromide levels) to elevated DBP levels in drinking water systems (PA DEP, 2011a). In response, PA DEP amended Chapter 95 Wastewater Treatment Requirements under the Clean Streams Law for new discharges of TDS in wastewaters. This regulation is also informally known as the 2010 TDS regulation. The regulation disallowed any new direct discharges to streams as well as direct disposal at POTWs of hydraulic fracturing wastewater and set limits on treated discharges from new CWTs of 500 mg/L TDS, 250 mg/L chloride, 10 mg/L barium, and 10 mg/L strontium. Existing discharges were exempt.

In April 2011, PA DEP announced a request that by May 19, 2011, gas drilling operators voluntarily stop transporting wastewater from shale gas extraction (i.e., unconventional resources as defined by PA DEP) to the eight CWTs and seven POTWs that were exempt from the 2010 TDS regulation.[1] Follow-up letters from PA DEP to the owners of the wells specified that the role of bromides from Marcellus Shale wastewaters in the formation of total trihalomethanes (TTHM) was of concern due to the their potential public health impacts (PA DEP, 2011a).

In response to the request, the oil and gas industry in Pennsylvania accelerated the switch of wastewater deliveries from POTWs to CWTs for better removal of metals and suspended solids (Schmidt, 2013). Effluent sampling at two POTWs that had accepted Marcellus Shale wastewater showed that concentrations of bromide, chloride, barium, strontium, and sulfate dropped after the April 2011 request (Ferrar et al., 2013); data based on two sampling events, one before and one after May 2011).

Between early and late 2011, although reported wastewater production more than doubled, Marcellus Shale drilling companies in Pennsylvania reduced their use of CWTs that were exempt from the 2010 TDS regulation by 98%, and direct disposal of Marcellus Shale wastewater at POTWs was "virtually eliminated" (Hammer and VanBriesen, 2012).

Along with the decreased discharges from POTWs, there has been increased reuse of wastewater in the Marcellus Shale region. From 2008-2011, reuse of Marcellus wastewater for hydraulic fracturing increased, POTW treatment volumes decreased, tracking of wastewater improved, and wastewater transportation distances decreased (Rahm et al., 2013). Maloney and Yoxtheimer (2012) analyzed data from 2011 and found that reuse of flowback increased to 90% by volume. Eight percent of flowback was sent to CWTs. Brine water, which was defined as formation water, was reused (58%), disposed via injection well (27%), or sent to CWTs (14%). Of all the fluid wastes in the analysis, brine water was most likely to be transported to other states (28%). Maloney and Yoxtheimer (2012) also concluded that wastewater disposal to municipal sewage treatment plants declined nearly 100% from 47,221 bbls in the first half of 2011 to 408 bbls in the second half.

[1] An unconventional formation was defined in 2011 by the state of Pennsylvania as "A geological shale formation existing below the base of the Elk Sandstone or its geologic equivalent stratigraphic interval where natural gas generally cannot be produced at economic flow rates or in economic volumes except by vertical or horizontal wellbores stimulated by hydraulic fracture treatments or by using multilateral wellbores or other techniques to expose more of the formation to the wellbore." The EPA defines unconventional oil and gas as crude oil and natural gas produced by a well drilled into a shale and/or tight formation (including, but not limited to, shale gas, shale oil, tight gas, and tight oil). For the purpose of the rule, the definition of UOG does not include CBM (U.S. EPA, 2016d).

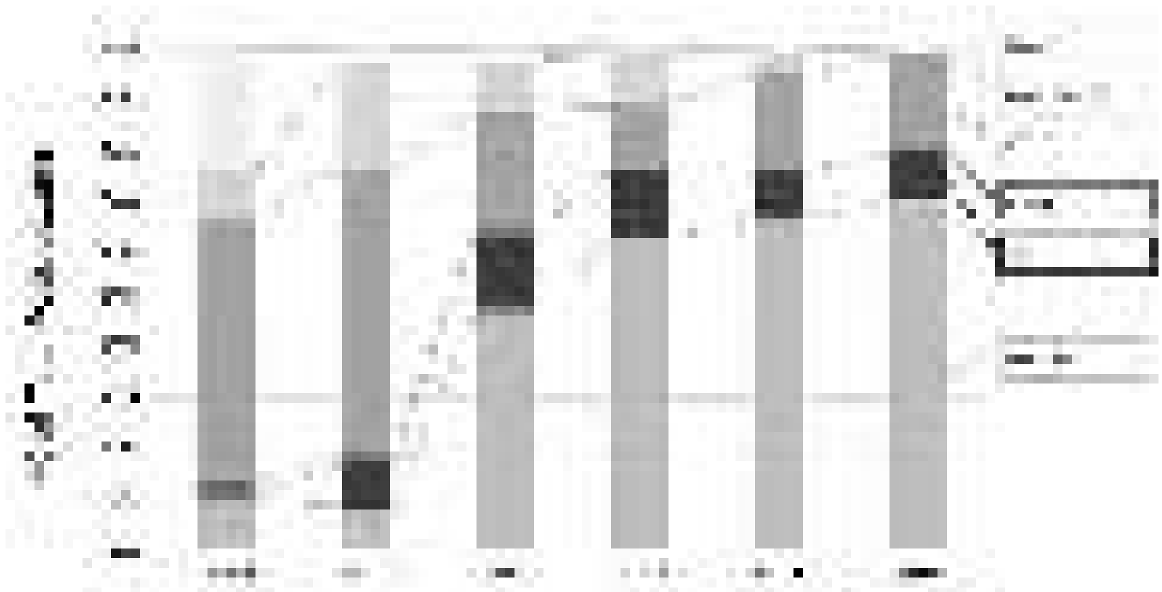

Figure 8-4. Percentages of total unconventional wastewater (as defined by PA DEP) managed via various practices for the second half of 2009 through the first half of 2014.
The volume sent to POTWs in 2013 was 0%. Note also that a majority of wastewater sent to CWTs is subsequently reused, so that when combined with "Reuse HF," the total reuse rate was approximately 90% in 2013. "Reuse HF" indicates on-site reuse. Source: Waste data from PA DEP (2015a).

Text Box 8-2. Regulations Affecting Wastewater Management.

Regulations affect wastewater management options and vary geographically as well as over time. At the Federal level, the EPA has promulgated national technology-based regulations, known as effluent limitations guidelines and standards (ELGs), for the oil and gas extraction industry, which can be found in 40 U.S. Code of Federal Regulations (CFR) Part 435. These ELGs do not apply to CBM discharges which are subject to technology based limits developed by permit writers on a case-by-case "best professional judgment" basis. The Onshore subcategory of the oil and gas, ELGs 40 CFR 125.3, Subpart C, prohibits the discharge of wastewater pollutants to waters of the United States from onshore oil and gas extraction facilities, with one exception in the arid west as discussed below. This "zero-discharge standard" means that, unless the exception applies, oil and gas wastewater pollutants cannot be discharged directly to waters of the United States. Operators have met this requirement through underground injection, reuse, or transfer of wastewater to POTWs and/or CWTs. The EPA finalized a rule in June 2016 that would prohibit operators from sending wastewater from unconventional oil and gas extraction to POTWs. Operators can continue to send wastewater to CWTs, which are subject to regulation under a separate set of ELGs in 40 CFR Part 437.

In addition, Subpart E of the oil and gas ELGs establishes an exception to the zero discharge standard west of the 98th meridian (the arid western portion of the continental United States), allowing discharges of produced water from onshore oil and gas extraction facilities to waters of the United States if the produced water has a use in agriculture or wildlife propagation when discharged into navigable waters. The term "use in agricultural or wildlife propagation" means that: (1) the produced water is of good enough quality to be used for wildlife or livestock watering or other agricultural uses; and (2) the produced water is actually put to

(*Text Box 8-2 is continued on the following page.)*

Text Box 8-2 (continued). Regulations Affecting Wastewater Management.

such use during periods of discharge (40 CFR 135.51(c)). Produced water discharged under this exception must not exceed an oil and grease concentration of 35 milligrams per liter (mg/L). Subpart E does not allow for discharge from sources other than produced water (i.e., drilling muds, drill cuttings, produced sands) to waters of the United States.

In addition to the technology-based limitations discussed above, the Clean Water Act (CWA) and the EPA's implementing regulations also require that permits include more stringent limits as necessary to meet applicable water quality standards. CWA Section 301(b)(1)(C); 40 CFR 122.44(d)(1).

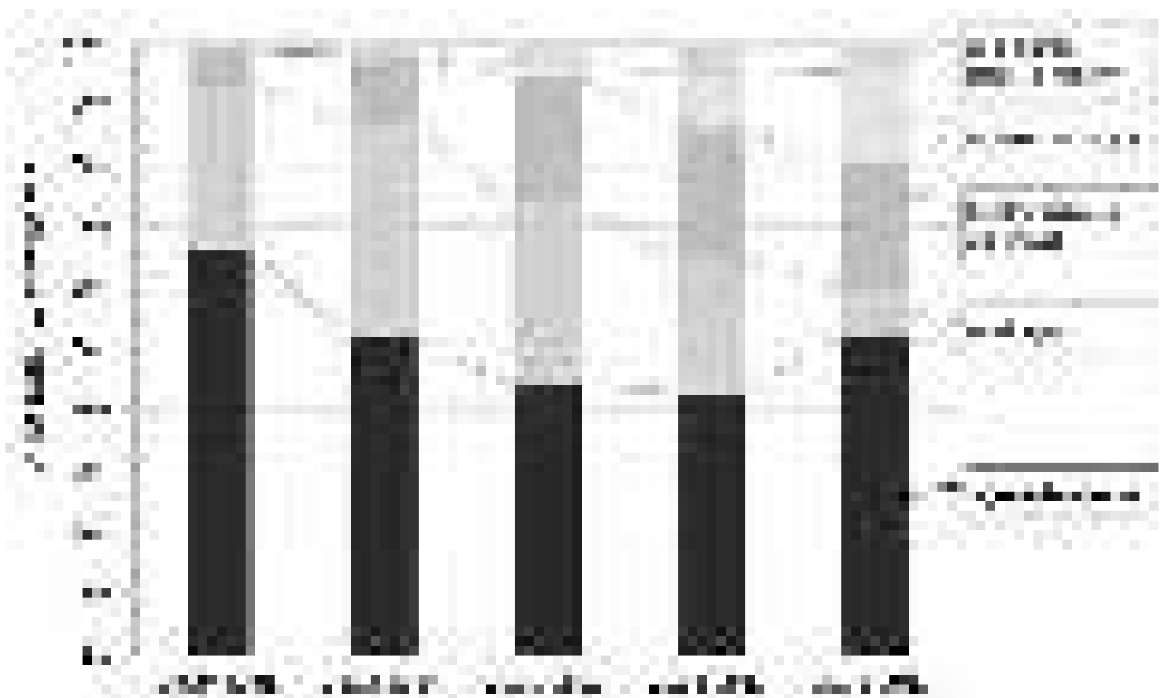

Figure 8-5. Management of wastewater in Colorado in regions where hydraulic fracturing is being performed.

See footnote for details on disposal codes.[1] Production data from Colorado Oil and Gas Conservation Commission (COGCC, 2015).

The following sections provide an overview of hydraulic fracturing wastewater management methods, with some discussion of the geographic and temporal variations in practices and their impacts on drinking water resources. In addition to currently used treatment and disposal methods, this section also briefly describes past treatment of hydraulic fracturing wastewater at

[1] Codes for wastewater disposal from COGCC are described by Veil (2015) as follows:

- Commercial disposal facility: water sent to commercial pits.
- On-site pit: most water evaporates, or excess water is hauled to disposal wells.
- Central disposal pit: Central facilities owned by a single producer to handle water from multiple wells (some recycled, much is injected).
- Injected on lease: Injected into wells, roughly half for enhanced recovery.
- Surface discharge: water is either fresh or treated to acceptable standards and discharged to surface water.

POTWs. More in-depth descriptions of treatment technologies applicable to hydraulic fracturing wastewater are available in Appendix F.

8.4.1 Underground Injection

Oil- and gas-related wastewater may be disposed of via Class II injection wells (disposal wells are referred to as Class IID whereas enhanced recovery wells are referred to as Class IIR) regulated by the UIC Program under the SDWA.[1] Nationwide, injection wells receive a large percentage of wastewater from the oil and gas industry, including wastewater associated with hydraulic fracturing. Veil (2015) estimates that in 2012, U.S. oil and gas production from onshore wells generated over 863 billion gal (20.56 billion bbls or 3.27 trillion L) of produced water, and of that volume, information on management was available for 97%. The study estimated that about 93% was injected into Class II wells, with about 47% injected into Class IID wells and 46% injected into Class IIR wells.[2]

The above national estimates are for the oil and gas industry as a whole. A good national estimate of the amount of hydraulic fracturing wastewater injected into Class II wells is difficult to develop due to lack of available information and data on injection of hydraulic fracturing wastewater. Management of hydraulic fracturing wastewater is not well tracked or made publicly available in many states (Pennsylvania being a notable exception). The local availability of Class IID wells along with generally low reuse rates, however, are consistent with Class IID wells being a primary means of wastewater management in many areas with hydraulic fracturing activity.

According to recently released data from 2012 and 2013, there are about 26,400 active Class IID wells in the United States, with more than 65% of these located in Texas, Oklahoma, and Kansas (Table 8-4). In Pennsylvania, on the other hand, there are currently nine operating disposal wells, and only three of these are commercially operated wells (at one facility) (SAFER PA, 2015). The location and number of Class IID wells is in part determined by geology (including depth and permeability of geologic formations appropriate for injection), permitting, and historical demand for disposal of oil and gas wastewater. The large Class IID well capacity in Texas, for example, is consistent with the availability of formations with suitable geology and the demand for wastewater disposal associated with a mature and active oil and gas industry. In contrast, injection plays a relatively small role in Marcellus Shale wastewater management in Pennsylvania—about 10% in 2013 and the first half of 2014 (PA DEP, 2015a).

[1] States may be given federal approval to run a UIC program under SDWA. UIC Class II wells include those used for disposal (Class IID), enhanced oil recovery (Class IIR), and hydrocarbon storage (Class IIH).

[2] Because some states surveyed by Veil (2015) do not distinguish between volumes injected for disposal versus enhanced recovery, assumptions and analyses were used in the study to estimate the two types of injection in some states.

Table 8-4. Distribution of active Class IID wells across the United States.
Data are primarily from 2012 and 2013. Source: U.S. EPA (2016d).

Geographic region (from the EIA)	State	Number of active disposal wells[a]	Average disposal rate per well (gpd/well)[b]	State disposal rate (MGD)
Alaska	Alaska	45	182,000	8.2
East	Illinois	1,054	—[c]	—[c]
	Michigan	772	16,200	13
	Florida	14	246,000	3.4
	Indiana	208	7,950	1.7
	Ohio	190	8,570	1.6
	West Virginia	64	6,970	0.45
	Kentucky	58	4,650	0.27
	Virginia	12	17,500	0.21
	Pennsylvania	9	6,380	0.057
	New York	10[d]	33.7	0.00034
Gulf Coast/Southwest	Texas	7,876	52,100	410
	Louisiana	2,448	40,300	99
	New Mexico	736	48,600	36
	Mississippi	499	24,200	12
	Alabama	85	53,300	4.5
Mid-Continent	Kansas	5,516	25,600	140
	Oklahoma	3,837	35,900	140
	Arkansas	640[e]	25,400	16
	Nebraska	113	19,100	2.2
	Missouri	11	2,270	0.025
	Iowa	3	—[c]	—[c]
Northern Great Plains	North Dakota	395	53,300	21
	Montana	199	32,700	6.5
	South Dakota	15	17,400	0.26

Geographic region (from the EIA)	State	Number of active disposal wells[a]	Average disposal rate per well (gpd/well)[b]	State disposal rate (MGD)
Rocky Mountains	Wyoming	335	107,000	36
	Colorado	292	48,800	14
	Utah	118	83,400	9.8
West Coast	California	826	86,800	72
	Nevada	10	54,600	0.55
	Oregon	9	—[c]	—[c]
	Washington	1	—[c]	—[c]
Total		**26,400**	**41,300**	**1,050**

Abbreviations: gpd—gal per day; MGD—million gal per day.

[a] Number of active disposal wells is based primarily on data from 2012 to 2013.

[b] Typical injection volumes per well are based on historical annual volumes for injection for disposal divided by the number of active disposal wells during the same year (primarily 2012 to 2013 data).

[c] Disposal rates and volumes are unknown.

[d] These wells are not currently permitted to accept extraction wastewater from production in unconventional reservoirs.

[e] Only 24 of the 640 active disposal wells in Arkansas are in the northern half of the state, close to the Fayetteville Shale.

The decision to inject hydraulic fracturing wastewater into Class IID wells depends in part on cost, including transportation costs. Therefore, the distance between the production well and a disposal well is an important consideration. For oil and gas producers, underground injection is a low cost management strategy unless significant trucking is needed to transport the wastewater to a disposal well (U.S. GAO, 2012).

Evaluation of documented or potential impacts on drinking water resources associated with disposal at Class IID injection wells is outside of the scope of this assessment. However, disposal wells play a significant role in the overall management of hydraulic fracturing water nationwide, and their availability and capacity are integral factors in determining which wastewater management strategies are used by operators in a given region. Should the feasibility of managing wastewater via underground injection become limited or less economically advantageous, operators will need to adjust their wastewater management programs. They may evaluate and implement other local practices such as sending wastewater to a CWT for treatment and discharge or reuse.

Recent events and studies, for example, have documented a link between wastewater injection and seismic activity in some locations in several states, including Oklahoma, Colorado, New Mexico, Arkansas, and Ohio (Weingarten et al., 2015; Wong et al., 2015). The Oklahoma Geological Survey (Andrews and Holland, 2015) "considers it very likely that the majority of recent earthquakes, particularly those in central and north-central Oklahoma, are triggered by the injection of produced water in disposal wells." Walsh and Zoback (2015) correlated wastewater injection from production wells (including hydraulically fractured wells) into Oklahoma's Arbuckle formation to the steep increase in seismic events observed in that state. Farther west, in the Raton Basin of southern Colorado and northern New Mexico, Rubinstein et al. (2014) presented several lines of evidence linking injection well disposal of CBM produced water to seismic events. Horton (2012) attributed a swarm of earthquakes in Northern Arkansas to hydraulic fracturing wastewater injection, and in a study evaluating multiple states in the mid-continent region, Weingarten et al. (2015) demonstrated a relationship between Class II wells (including both Class IID and Class IIR wells) and seismicity.

The local availability of Class IID wells and the capacity to accept large volumes of wastewater could be affected by these recent findings concerning seismic activity associated with injection (U.S. EPA, 2014c). Between 2011 and 2016, some state UIC programs modified their Class II wastewater injection regulations and permitting requirements. At least eight states (Arkansas, Colorado, Illinois, Kansas, Ohio, Oklahoma, Texas, and West Virginia) consider an assessment of seismicity in their Class II programs and have regulatory provisions for banning or shutting injection wells and/or modifying injection volumes and pressures if evidence indicates that a well is near a fault and/or is contributing to seismic activity.

As an example, Oklahoma has recently taken steps to reduce the risk of induced seismicity by implementing a regional strategy intended to reduce wastewater injection in certain regions (OCC OGCD, 2016). These actions affect over 10,000 square miles and 600 wastewater injection wells in western and central Oklahoma. The measures are intended to reduce wastewater injection in the area by 40% below 2014 totals, which will affect wastewater management and disposal practices across this large area.[1]

In terms of potential impacts on drinking water resources, Class IID facilities are subject to the same general considerations regarding wastewater storage and handling as other wastewater management sites and facilities (e.g., CWTs). Changes in surface water or groundwater quality due to general wastewater handling at these facilities may be another factor affecting wastewater management practices in some locations or regions. For example, Kell (2011) identified eight groundwater contamination incidents in Texas between 1993 and 2008 due to water releases from storage facilities associated with Class II well sites. A recent study by the United States Geological Survey documented impacts on surface water from hydraulic fracturing wastewater at a Class II disposal well site in central West Virginia (Akob et al., 2016). Water samples collected downstream from the facility were indicative of wastewater from hydraulic fracturing operations handled at the site. The authors documented elevated specific conductance and elevated TDS, sodium, chloride, barium, bromide, strontium, and lithium concentrations, and different strontium isotope ratios compared to those found in upstream, background waters. The study concluded that activities at the wastewater facility have affected water quality in a nearby stream. The pathways for the movement of wastewater into the local stream include several possibilities (e.g., leaks from storage ponds and tanks, transportation activities, previous site history).

[1] For additional information on strategies and initiatives regarding wastewater injection and inducted seismicity, see the following: KDHE (2014), States First Initiative (2014), and U.S. EPA (2014c).

8.4.2 Publicly Owned Treatment Works

POTWs are designed to treat local municipal wastewater and indirect discharges from industrial users. POTWs are also used to treat wastewater and other wastes from oil and gas operations in some eastern states. Although this is not a common method of treatment for oil and gas wastewaters in the United States, the scarcity of injection wells for waste disposal in Pennsylvania drove the need for disposal alternatives (Wilson and Vanbriesen, 2012). When development of the Marcellus Shale began, POTWs were used to treat wastewater originating from these oil and gas wells (Kappel et al., 2013; Soeder and Kappel, 2009). However, elevated concentrations of constituents in wastewater from the Marcellus region (halides, heavy metals, organic compounds, radionuclides, and salts) can pass through the treatment processes commonly used in POTWs and be discharged to receiving waters (Cusick, 2013; Kappel, 2013; Lutz et al., 2013; Schmidt, 2013). In addition, sudden, extreme salt fluctuations can disturb POTW biological treatment processes (Linarić et al., 2013; Lefebvre and Moletta, 2006).

The annual reported volume of oil and gas wastewater treated at POTWs in the Marcellus Shale region peaked in 2008 and has since declined significantly (Figure 8-6). As discussed in Text Box 8-1, this was in response to an April 2011 request from PA DEP asking operators to cease sending Marcellus Shale wastewater to 15 POTWs and CWTs that were exempt from the 2010 TDS regulation (Rahm et al., 2013). Although operators complied with the request in May 2011, non-Marcellus oil and gas produced water continued to be processed at these facilities (Ferrar et al., 2013; Lutz et al., 2013; Wilson and Vanbriesen, 2012).[1] In August 2016, the EPA finalized pretreatment standards prohibiting discharges of unconventional wastewater pollutants to POTWs (U.S. EPA, 2016d).

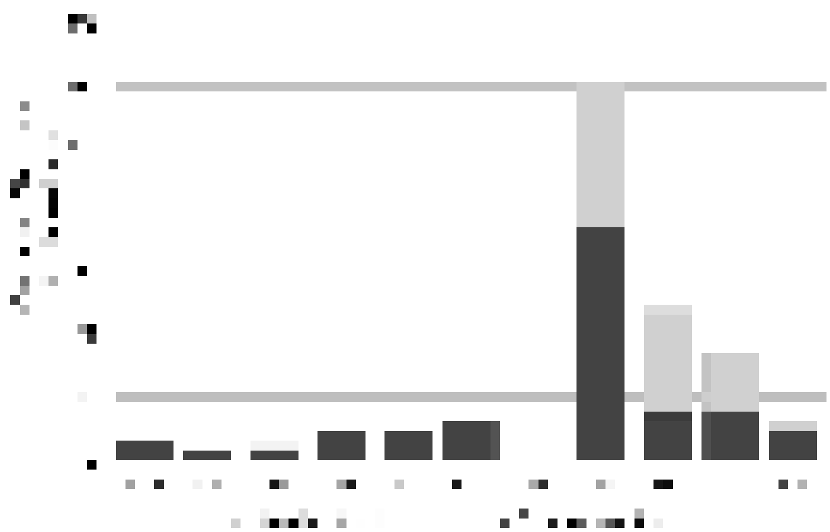

Figure 8-6. Oil and gas wastewater volumes discharged to POTWs from 2001-2011 in the Marcellus Shale. ("Conventional" is indicated by the authors as non-Marcellus wells and described as vertically drilled to shallower depths in more porous formations.)
Due to an unrecoverable data loss at the PA DEP, records for 2007 were not available. Source: Lutz et al. (2013).

[1] POTWs in Pennsylvania have likely been accepting waste considered conventional by Pennsylvania but unconventional by others based on the EPA's broader definition (Text Box 8-1).

8.4.3 Centralized Waste Treatment Facilities

A CWT facility is generally defined as one that accepts industrial materials (hazardous or non-hazardous, solid, or liquid) generated at another facility (off-site) for treatment or recovery (EPA, 2000). (Wastewater may also be treated at on-site mobile or semi-mobile facilities; see Appendix F for additional information.) The decision to treat hydraulic fracturing wastewater at a CWT and the level of treatment used depends upon several factors, such as a lack of proximity to Class II disposal wells; whether the wastewater might be reused for additional hydraulic fracturing jobs; the water quality needed if it will be reused; whether the treated wastewater can be discharged under the Subpart E agricultural and wildlife use exception in the arid west; and the water quality needed if it will be discharged to the waters of the United States. As a group, CWTs that accept oil and gas wastewater offer a wide variety of treatment capabilities and configurations (Text Box 8-3 and Appendix F).

Text Box 8-3. Wastewater Treatment Processes.

The constituents prevalent in hydraulic fracturing wastewater include TDS, TSS, radionuclides, organic compounds, and metals (Section 8.3 and Chapter 7). If the ultimate disposal or use of the wastewater necessitates treatment, a variety of technologies can be employed to remove or reduce these constituent concentrations.

The most basic treatment needed for oil and gas wastewaters, including those from hydraulic fracturing operations, is separation to remove TSS and oil and grease. This is accomplished through separation technologies including settling, skimming, hydrocyclones, dissolved air or induced gas flotation, media filtration, or biological aerated filters (Igunnu and Chen, 2014; Duraisamy et al., 2013; Barrett, 2010; Shammas, 2010).

Other treatment processes that may be used include media filtration after chemical precipitation for hardness and metals (Boschee, 2014); adsorption technologies for organics, heavy metals, and some anions (Igunnu and Chen, 2014); a variety of membrane processes (microfiltration, ultrafiltration, nanofiltration, reverse osmosis (RO)); and distillation technologies for metals and organics (Drewes et al., 2009).

Advanced processes, such as RO, or distillation methods, such as mechanical vapor recompression (MVR), are needed if the system requires significant reduction in TDS (Drewes et al., 2009; LEau LLC, 2008; Hamieh and Beckman, 2006). However, RO is typically only capable of treating TDS concentrations less than 35,000 mg/L (Shaffer et al., 2013), whereas distillation can effectively treat higher TDS waters (Hayes et al., 2014; Drewes et al., 2009). Extremely high TDS waters may require a series of advanced treatment processes, which can be very costly.

An emerging technology in hydraulic fracturing wastewater treatment is electrocoagulation, which has been used in mobile treatment systems to remove organics, TSS, and metals (Halliburton, 2014; Igunnu and Chen, 2014).

Appendix F provides more in-depth descriptions of technologies used to treat for hydraulic fracturing wastewaters and the constituents they remove. Also, Appendix Table F-4 provides an overview of influent and effluent results and removal percentages for constituents of concern at oil and gas treatment facilities reported in the literature (both conventional and unconventional) and the specific technology(ies) used to remove them. Section 8.4.7 discusses solid and liquid residuals, including treatment-related wastes.

The treated effluent from a CWT can be reused in hydraulic fracturing operations (also called zero-discharge), discharged directly to a receiving water under a National Pollutant Discharge Elimination System (NPDES) permit, discharged indirectly to a POTW, or a combination of these. Some CWTs may be configured so that they can either (1) partially treat the waste stream to suit the needs of operators who reuse it or (2) use more advanced treatment (i.e., TDS removal) if the treated wastewater will be discharged. Generally, the former option is less costly for the CWT, and some facilities that have permits to discharge do not do so continuously, opting to direct as much of the wastewater as possible for reuse. There are also CWTs permitted to discharge that do not have TDS removal capabilities. However, these facilities must still meet TDS discharge limits specified by their state. Appendix F contains additional information on treatment configurations, including examples of processes at several facilities treating oil and gas wastewater.

Facilities discharging treated wastewater to waters of the United States or POTWs are regulated under the Clean Water Act (CWA). For zero-discharge facilities, some states, including Pennsylvania and Texas, have adopted regulations to control permitting of these facilities or to encourage treatment and reuse. The PA DEP issues permits that allow zero-discharge CWTs to treat and release water back to oil and gas industries for reuse (see the Eureka Resources Facility in Williamsport, PA listed in Appendix Table F-6 as an example of a zero-discharge facility).[1]

In developing this assessment, we looked at NPDES permit information for several CWTs in the eastern United States treating wastewater from the Marcellus region and one near the Fayetteville Shale in Arkansas. The facilities include those with and without TDS removal capabilities, and some are undergoing upgrades to implement TDS removal. Some of the permits reviewed for this assessment are current, and others are expired and may be in the process of renewal. The permits require monitoring (with or without limits) for a range of constituents that may include chloride, TDS, TSS, total strontium, total barium, oil and grease, heavy metals, 5-day biological oxygen demand (BOD5), and a range of organic compounds (e.g., phenol, cresol, BTEX, phthalates), with the specific constituents varying by permit. Sample types for the facilities are generally 24-hour composites. The newer permits set limits for several important constituents such as chloride, TDS, TSS, total barium, total strontium, oil and grease, and a number of heavy metals. Bromide is generally either not included or is required to be reported but with no limit specified. However, limits on TDS will reduce bromide concentrations. Some permits require monitoring for total radium, uranium, and gross alpha, but no limits are specified. Note that these facilities do not necessarily discharge consistently because treated wastewater can be sent for reuse.

Although there are CWTs serving hydraulic fracturing operations throughout the country, the majority serve Marcellus Shale operations in Pennsylvania (Boschee, 2014). Of the 74 CWT facilities identified by the EPA (U.S. EPA, 2016d) as having accepted or having the ability to accept hydraulic fracturing wastewater (not counting facilities treating CBM wastewater), 40 are located in Pennsylvania (Table 8-5). Most are zero-discharge facilities, and many do not have treatment processes for TDS removal. Although several Pennsylvania facilities are permitted to discharge, Wunz (2015) found few that currently discharge (two CWTs in Pennsylvania, one in West Virginia,

[1] The facility is also permitted for indirect discharge to the Williamsport Sewer Authority.

Table 8-5. Number, by state, of CWT facilities that have accepted or plan to accept wastewater from unconventional oil and gas activities.
Source: U.S. EPA (2016d).

State	Unconventional formation(s) served	Zero discharge CWT facilities[a]		CWT facilities that discharge to a surface water or POTW[a]		CWT facilities with multiple discharge options[a]		Total known facilities
		Non-TDS removal	TDS removal	Non-TDS removal	TDS removal	Non-TDS removal	TDS removal	
AR	Fayetteville	2	0	0	0	0	1	**3**
CO	Niobrara, Piceance Basin	3 (1)	0	0	0	0	0	**3**
ND	Bakken	0	1 (1)	0	0	0	0	**1**
OH	Utica, Marcellus	10 (7)	0	1	0	0	0	**11**
OK	Woodford	2	0	0	0	0	0	**2**
PA	Utica, Marcellus	22	7(3)	8	0	0	3 (1)	**40**
TX	Eagle Ford, Barnett, Granite Wash	1	3	0	0	0	0	**4**
WV	Marcellus, Utica	4 (2)	0	0	0	1	1	**6**
WY	Mesaverde and Lance	0	2	0	0	0	2	**4**
Total		**44**	**13**	**9**	**0**	**1**	**7**	**74**

[a] Information is current as of 2014; it is possible that since 2014, some listed CWT facilities have closed and/or new CWT facilities not listed have begun operation. The number of facilities includes those that have not yet opened but are under construction, pending permit approval, or are in the planning stages. Facilities that are not accepting hydraulic fracturing wastewater but plan to accept it in the future are noted parenthetically and not included in the sum of total known facilities. Facilities handling CBM wastewater are not represented here.

and one in Ohio). According to EPA research (U.S. EPA, 2016d), the number of CWT facilities serving operators in the Marcellus and Utica Shales has increased since the mid-2000s, growing from roughly five in 2004 to over 40 in 2013. A similar trend has been noted for the Fayetteville Shale region in Arkansas, where there are fewer Class IID injection wells compared to the rest of the state (U.S. EPA, 2016d).

In other regions, a small number of newer facilities have emerged in the last several years, most often with TDS removal capabilities. In Texas, for example, two zero-discharge facilities with TDS removal capabilities are available to treat wastewater from the Eagle Ford Shale (beginning in 2011 and 2013), and one zero-discharge facility with TDS removal is located in the Barnett Shale region (operational since 2008). In Wyoming, there are four facilities in the region of the Mesaverde/Lance formations that started operating between 2006 and 2012. Two are zero-discharge facilities, and two have multiple discharge options; all are capable of TDS removal (U.S. EPA, 2016d).

Few states maintain a comprehensive list of CWT facilities, and the count provided by the EPA (U.S. EPA, 2016d) includes facilities that do not currently but plan to accept wastewater from unconventional formations. Therefore, the data in Table 8-5 do not precisely reflect the number of facilities currently handling hydraulic fracturing wastewaters. Other sources indicate either use of, or interest in, development of treatment facilities in other regions such as the Barnett Shale region (Hayes and Severin, 2012b), the Fayetteville (Veil, 2011), and other areas in Texas and Wyoming (Boschee, 2014, 2012). In addition, news releases and company announcements indicate that other wastewater treatment facilities are being planned (Greenhunter, 2014; Geiver, 2013; Purestream, 2013; Alanco, 2012; Sionix, 2011).

Use of specific types of CWTs has and will continue to shift due to drivers such as availability and cost of other disposal options (e.g., disposal wells), operator demand for reuse and the associated quality needed, developments in treatment, treatment costs, and regulatory changes. Practices in Pennsylvania over the last several years provide such an example. Between 2010 and 2013, the percentage of Marcellus wastewater treated at CWTs dropped from 52% to 20% (Figure 8-4), and the percentage of wastewater reused on-site rose to 65%, reflecting a shift in practice among operators. Among the percentage of the wastewater sent to CWTs, the portion sent to zero-discharge facilities for subsequent reuse rose from 10% to 65%. This is consistent with an increased emphasis on reuse in Pennsylvania. (See Section 8.4.4 for a discussion on reuse as a waste management practice.)

8.4.3.1 Relationship to Potable Surface Waters

Figure 8-7 shows the relationship between Pennsylvania potable water supplies and the CWTs that lie in their upstream watersheds. These surface waters, including streams, rivers, and waterbodies (e.g., lakes and reservoirs) have been evaluated by the PA DEP for attainment of a designated use of potable water supply as per the CWA Section 305(b) reporting and Section 303(d) listing. Ninety-four percent of the waterbodies and 98% of the streams and rivers were attaining their designated use in 2016. These stream segments may or may not currently have intakes for drinking water treatment plants. The map also shows the locations and types of CWTs that either currently accept unconventional oil and gas wastewater (as defined by PA DEP) or have accepted such wastewater

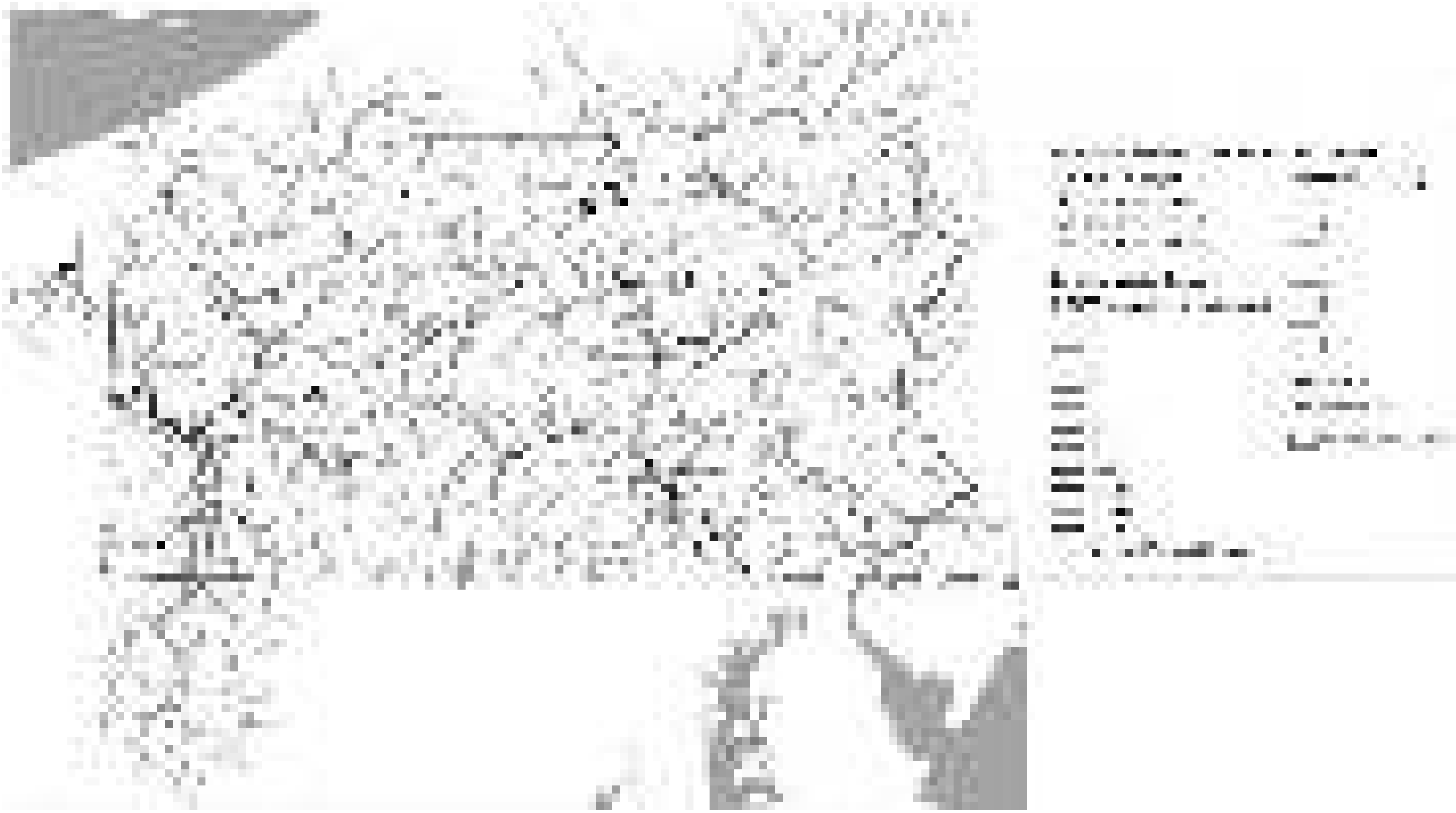

Figure 8-7. Map showing Pennsylvania surface water designated as potable water supplies and upstream CWTs.
Surface waters are colored orange to red to indicate the number of CWTs located in the upstream watershed. Blue surface waters have no upstream CWTs, and light gray lines show those not designated as potable water supplies. Symbols show the locations of CWTs that currently accept or have accepted unconventional oil and gas wastewater. Data sources: U.S. EPA (2016d), U.S. EPA (2016f), and PA DEP (2016b).

within approximately the last five years.[1] CWTs represented include both dischargers (direct and indirect) as well as zero-discharge facilities. For some facilities, we were not able to determine if the facility was zero-discharge or if it has a NPDES permit. The surface waters have been color-coded to indicate the number of CWTs that are located upstream. Darker red indicates more CWTs located in the upstream watershed, while blue indicates no upstream CWTs. Softer grey lines show portions of the stream network not designated for potable water supply. The thickness of the line indicates the size of the stream or river, categorized by the "stream order" designation.

The map provides a general illustration of how CWTs are situated within catchments in Pennsylvania, showing their spatial and general hydrologic relationships to streams that can serve as potable water supplies. The map shows that a given stream or waterbody may have a number of CWTs upstream, potentially contributing to combined impacts on surface water if there are spills or inadequately treated discharges. Note that the upstream catchment areas are large for the major rivers. Therefore, some rivers, such as the Ohio or Susquehanna, have as many as 15 or 16 upstream CWTs, although most are located far away. The map does not represent the effects of dilution on either discharges or spills; such an evaluation would necessitate currently unavailable data required to do a complete analysis of water quality. Note that many of the CWTs are zero-discharge facilities, and those that are permitted to discharge may do so intermittently. However, the storage and handling of wastewater at CWTs could impact nearby surface water through leaks and spills.

To more completely place these facilities in a watershed context, other types of discharges that could be occurring upstream should be taken into consideration. Impacts from hydraulic fracturing wastewater may be more problematic if there are additional pollutant sources within the watershed, increasing the cumulative effects of discharges and spills. For example, an EPA source apportionment study (U.S. EPA, 2015o) evaluated the relative contributions of bromide, chloride, nitrate, and sulfate from CWTs primarily treating hydraulic fracturing wastewater to the Allegheny River Basin and to two downstream public water system intakes. The study considered that the Allegheny River and its tributaries also receive runoff and discharges from an array of sources that include acid mine drainage and mining operations, coal-fired electric power stations, industrial wastewater treatment plants, and POTWs. It was concluded that CWTs treating oil and gas wastewater and coal-fired power plants with flue gas desulfurization were the primary contributors of bromide and chloride at the intakes (see Section 8.5.1 for further discussion), while nitrate and sulfate contributions were from POTWs and Acid Mine Drainage (U.S. EPA, 2015o).

8.4.3.2 Potential Impacts from CWTs

The potential impacts of managing hydraulic fracturing wastewater at CWTs depend on whether the CWT adequately treats for constituents of concern prior to discharge to surface water or a POTW, and whether treatment residuals are managed appropriately. Historically, CWTs have not

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[1] The list of CWTs used to develop this map is based on best available data, including information in the technical development document supporting the new EPA unconventional oil and gas effluent limitation guidelines (U.S. EPA, 2016d) as well as data from PA DEP waste records. This information was supplemented with other publicly available descriptions of the facilities. The information may, however, not be complete, and the symbols may not definitively reflect the discharge status of a facility.

included processes to treat for constituents that are difficult to remove, such as the high concentrations of TDS found in wastewater from unconventional reservoirs. As a result, impacts on drinking water resources have included increased suspended solids and chloride concentrations downstream of discharging facilities that were accepting hydraulic fracturing wastewater (Olmstead et al., 2013) and elevated bromide concentrations and radium concentrations in CWT effluent (Warner et al., 2013a); see Sections 8.5.1 and 8.5.2. In addition, spills and leaks can occur in pits or impoundments associated with the storage of treated wastewater at CWTs (impacts related to spills and leaks from pits and impoundments are discussed in Section 8.4.5). Wastewater being transported by truck or pipeline to and from a CWT can also present a vulnerability for spills or leaks (Easton, 2014) (Chapter 7).

While selection of appropriate treatment processes is critical for CWTs that discharge to surface waters, there are also two important issues related to completeness of treatment that can have an impact. First, there may be unknown constituents in the wastewater. The effectiveness of treatment cannot be evaluated for constituents for which the wastewater has not been tested. This makes it challenging to know the degree to which effluent from a CWT is protective of public health. Second, even an efficient treatment process may not be able to reduce the concentrations of some constituents to levels that allow for discharge to a drinking water resource if influent concentrations are so high that they exceed the capabilities of the treatment technology(ies) to meet those discharge limits. For example, a facility described by Kennedy/Jenks Consultants (2002) removed a high percentage of boron (88%), but the effluent concentration of 1.9 mg/L (average influent concentration of 16.5 mg/L) was not low enough to meet California's action level of 1 mg/L. Thus, the influent concentration must be considered together with removal efficiency to determine whether the effluent quality will meet the requirements dictated by end use or by regulations.

Relatively few studies describe the ability of individual treatment processes to remove constituents from hydraulic fracturing wastewater. For this assessment, simple estimated effluent concentrations were calculated for several combinations of unit treatment processes, wastewater constituents, and influent concentrations (details are given in Appendix Table F-3). The purpose of the analysis was to illustrate the relative capabilities of a number of treatment processes and not to represent a complete treatment system. As an example, the estimates suggest that if wastewater contains radium with a concentration in the thousands of pCi/L, a 95% removal rate with chemical precipitation may result in an effluent that exceeds 100 pCi/L. Treatment of the same wastewater via distillation or reverse osmosis could result in effluent concentrations in the tens of pCi/L. This analysis suggests that attention should be paid to the capabilities of a planned treatment system for the full range of anticipated wastewater compositions.

To gain a better understanding of impacts, the USGS has conducted sampling for a wide array of water quality parameters in surface water and groundwater in the Monongahela River Basin in West Virginia to establish baseline water-quality conditions (Chambers et al., 2014). Future water quality sampling can be compared to this baseline to assess impacts from hydraulic fracturing activities. To address past impacts, Pennsylvania, having experienced water quality impacts on receiving streams due to discharges of high-TDS effluent modified their regulations to address these issues by setting water quality standards for CWT dischargers (Mauter and Palmer, 2014; Shaffer et al., 2013). (See Text Box 8-1.)

8.4.4 Wastewater Reuse for Hydraulic Fracturing

The reuse of hydraulic fracturing wastewater for subsequent hydraulic fracturing operations has increased in some regions of the country in recent years (Boschee, 2014, 2012; Gregory et al., 2011; Rassenfoss, 2011).[1] This practice is driven by factors that include cost (including treatment costs), the lack of availability of other management options (e.g., Class II disposal wells), and changes to state regulations (Boschee, 2014; Shaffer et al., 2013). Wastewater quality is a consideration; some constituents pose challenges for reuse and may necessitate treatment. For example, high concentrations of barium and sulfate can lead to scaling, and the presence of some constituents in wastewater can hinder crosslinking (Akob et al., 2016; Boschee, 2014). Hydraulic fracturing fluid formulations that can use high TDS waters (e.g., as high as 150,000 mg/L to over 300,000 mg/L) facilitate reuse with minimal treatment (Boschee, 2014; Mauter and Palmer, 2014). See Chapter 5 for more information regarding the chemical composition of hydraulic fracturing fluids and Appendix F for more discussion of considerations for reuse.

Reuse can be accomplished by blending either untreated or minimally treated hydraulic fracturing wastewater with fresh water to lower the TDS content (Boschee, 2014). Wastewater may be reused at a site with multiple wells, eliminating the need for transport to a CWT (Lester et al., 2015; Easton, 2014). Alternatively, wastewater can be treated at a CWT and then taken by operators for mixing with other water sources for reuse (Easton, 2014). Flowback may be preferable to later-stage produced water for reuse because of its lower TDS concentration. Also, it is typically generated in larger quantities from a single location as opposed to water produced later on, which is generated in smaller volumes over time from many different locations (Barbot et al., 2013; Maloney and Yoxtheimer, 2012). Reuse can reduce the costs associated with water acquisition and produced water management. Such economic and logistical benefits can be expected to inform ongoing wastewater management decisions.

Costs can be the most significant driver for reuse. For example, the costs of transporting wastewater from the generating well to the treatment facility and then to the new well can be weighed against the costs for transport to alternative locations (e.g., a disposal well). Trucking large quantities of water can be relatively expensive—from \$0.01 to \$0.19 per gallon (\$0.50 to \$8.00 per bbl)—rendering on-site treatment technologies and reuse economically competitive in some settings (Dahm and Chapman, 2014; Guerra et al., 2011). Reuse rates may also be driven by wastewater production rates compared to the demand for reuse, with both production and demand increasing in a region if more wells go into production or decreasing as plays mature (Lutz et al., 2013; Hayes and Severin, 2012b; Slutz et al., 2012). Other logistics to consider include proximity of the water sources for aggregation and sequencing of completion schedules (Mauter and Palmer,

[1] Reused hydraulic fracturing wastewater is discussed in Chapter 4 of this report (Water Acquisition) as well as in this chapter, though in a different context. The wastewater reuse rate described in this chapter is the amount or percentage of generated hydraulic fracturing wastewater that is managed through use in subsequent hydraulic fracturing operations. In contrast, Chapter 4 discusses reused wastewater as a source water and as one part of the base fluid for new fracturing fluid.

2014). A small survey by Mauter and Palmer (2014) indicates that the scheduling of well completions is complex, requiring optimization of labor, contractual issues, equipment usage, and water storage capacity among other factors. Boschee (2014) notes that in the Permian Basin, older conventional wells are linked by pipelines to a central disposal facility, facilitating movement of treated water to areas where it is needed for reuse. Companies drilling fewer wells or located in more remote areas may find reuse difficult because of challenges in consolidating wastewater from their wells or accessing wastewater from centralized facilities.

Regulations may also encourage reuse. For example, in 2013, the Texas Railroad Commission adopted rules eliminating the need for a permit when operators reuse on their own lease or transfer the fluids to another operator for reuse (Rushton and Castaneda, 2014). Any information on wastewater management practices in Texas that becomes available for the years after 2013 will allow evaluation of whether reuse has in fact increased.

A summary of reuse practices throughout the United States is hampered by the limited amount of data available for many regions of the country. However, current data indicate that extensive reuse takes place in the Marcellus region. Several studies using data from PA DEP data show that total reuse rates of oil and gas wastewater in Pennsylvania have risen over the last several years to between 85 and 90% (Table 8-6). This includes wastewater sent to CWTs to treat for reuse as well as reuse at the well sites without transfer to a CWT (labeled as "Reuse HF" in Figure 8-4). In particular, reuse of Marcellus wastewater at well sites in Pennsylvania has risen from about 8% in the second half of 2010 to nearly 70% in the first half of 2014 (PA DEP, 2015a). Schmid and Yoxtheimer (2015) report more recent data stating that in 2014, approximately 85% of Marcellus hydraulic fracturing wastewater was reused. Of that amount, 78% occurred on-site, and 22% was via CWTs.

Table 8-6. Estimated percentages of reuse of hydraulic fracturing wastewater.

Play or basin	Source and year	2008	2009	2010	2011	2012	2013	2014
East Coast[a]								
Marcellus, PA	Rahm et al. (2013)	9	8	25 – 48	67 – 80			
Marcellus, PA	Ma et al. (2014)		15 - 20				90	
Marcellus, PA	Shaffer et al. (2013)					90		
Marcellus, PA	Schmid and Yoxtheimer (2015)							85
Marcellus, PA	Hansen et al. (2013)	9	6	20	56			
Marcellus, PA	Maloney and Yoxtheimer (2012)				71.6			
Marcellus, PA	Tiemann et al. (2014)				72	87		

Play or basin	Source and year	2008	2009	2010	2011	2012	2013	2014
Marcellus, PA	Rassenfoss (2011)			~67 (general estimate) 96 (one company)				
Marcellus, PA	Wendel (2011)			75-85	90			
Marcellus, PA	Lutz et al. (2013)	13 (prior to 2011)			56			
Marcellus, PA (SW region)	Rahm et al. (2013)	~10	~15	~25-45	~70-80			
Marcellus, PA (NE region)	Rahm et al. (2013)	0	0	~55-70	~90-100			
Marcellus, WV	Hansen et al. (2013)			88	73	65 (partial year)		
Gulf Coast and Midcontinent								
Fayetteville	Veil (2011)			20 (single company target)				
Barnett	Rahm and Riha (2014), Nicot et al. (2012)				5 (general estimate – appears to cover recent years)			
Eagle Ford	Nicot and Scanlon (2012)				0	20 (estimate based on interviews)		
East Texas	Nicot and Scanlon (2012)				5			
Haynesville	Horner et al. (2014)						0	
Haynesville	Rahm and Riha (2014)				5 (general estimate – appears to cover recent years)			

Play or basin	Source and year	2008	2009	2010	2011	2012	2013	2014
West Coast and Upper Plains								
Denver-Julesburg (Weld County), CO	Sumi (2015)					54 (flow-back only)		
Bakken	Horner et al. (2014)						0	

[a] Studies focusing on the Marcellus Shale use waste data reports from PA DEP.

Reuse in the Marcellus region is higher in the northeastern part of Pennsylvania than in the southwestern portion where easier access to Class IID wells in Ohio makes disposal by injection more feasible (Rahm et al., 2013). Outside of the Marcellus region, reuse rates are lower. Ma et al. (2014) note that only a small amount of reuse is occurring in the Barnett Shale. Reuse has not yet been pursued aggressively in New Mexico or in the Bakken (North Dakota) (Horner et al., 2014; LeBas et al., 2013). Other sources, however, indicate growing interest in reuse, as evidenced in specialized conferences (e.g., "Produced Water Reuse Initiative 2014" on produced water reuse in Rocky Mountain oil and shale gas plays), and available state-developed information on reuse (e.g., fact sheet by the Colorado Oil and Gas Conservation Commission) (Colorado Division of Water Resources et al., 2014).

If hydraulic fracturing activity slows in an area that is currently reusing wastewater, demand for the wastewater may decrease and wastewater management practices may shift. Analysis by Wunz (2015) and data in Figure 8-1 suggest a decline in wastewater production in Pennsylvania. Wunz (2015) also notes that in the future, there could be a trend of more wastewater coming from late-stage produced water and less from flowback as more wells are in the production phase and fewer wells are being fractured. If the demand drops relative to production due to fewer wells being drilled and fractured, then the "excess" produced water will need to be managed by other means. Alternatives to reuse may include increased transport to disposal wells (e.g., those in Ohio), development of more disposal wells in Pennsylvania, or advanced treatment and discharge to surface water via CWTs that have TDS removal capabilities (SAFER PA, 2015; Wunz, 2015; Silva et al., 2014a).

8.4.4.1 Potential Impacts from Reuse

For companies employing reuse as a wastewater management strategy, surface spills and leaks can occur during wastewater transport to and from a treatment facility or from storage tanks/pits located at the treatment facility or at the well site. Releases may be due to failed infrastructure such as tank or pipe ruptures, from natural disasters such as floods or earthquakes, or incidents such as overfills, improper operations, or vandalism (CCST, 2015a; NYSDEC, 2011). If the spill or leak is not contained or otherwise mitigated, these releases could reach groundwater or surface water (CCST, 2015a; NYSDEC, 2011). See Chapter 7 for more discussion on types of spills associated with hydraulic fracturing activities, including storage and transport. See Section 8.4.5 for discussion of storage pits and associated impacts on drinking water resources.

With reuse there is the potential for accumulation of dissolved solids such as salts and TENORM in the wastewater over successive reuse cycles (see Section 7.3.4.6 and Section 8.5.2 for more information about TENORM). Because wastewater is often reused with minimal treatment, constituents resulting from time spent in the subsurface remain in the wastewater and can increase during additional hydraulic fracturing. This potentially concentrated wastewater can pose a bigger issue if a breach occurs in an on-site pit or tank storing this wastewater while awaiting reuse (Section 8.4.5; Chapter 7).

The issue of concentrating contaminants during reuse has not yet been quantitatively evaluated in the literature. Also, it is not known how much this problem would be mitigated due to the dilution of wastewater when reused as new fracturing fluid. Estimates of the percentages of reused wastewater in new fracturing fluids in Pennsylvania range from about 2% in 2009 to as much as 22% in 2013 (SRBC, 2016; Schmid and Yoxtheimer, 2015) (Chapter 4). However, data from Pennsylvania's TENORM study (PA DEP, 2015b) showed radium in some hydraulic fracturing fluids, presumably from a reused wastewater component. As reused wastewater continues to accumulate contaminants, the water will eventually need to be managed, either through treatment or injection.

8.4.5 Storage and Disposal Pits and Impoundments

The use of pits and impoundments as part of a wastewater management strategy is a historic as well as current practice in the oil and gas industry. These structures are either used for temporary storage (on-site at oil and gas production wells or off-site at CWTs or disposal wells) or they are intended for permanent disposal (evaporation or percolation). There are a variety of terms to describe these structures depending upon their use (Richardson et al., 2013); "pits," "impoundments," and "reserve pits" are some of the more common terms associated with wastewater management. The terms "impoundment" or "pond" are often used to refer to large area holding structures and are also used by some states for specific applications such as holding "freshwater" for fracturing fluid formulation (Quaranta et al., 2012). Definitions and terminology are not standardized and vary from state to state (Richardson et al., 2013). For the purposes of this section, the nomenclature will defer to the term used by the original author/regulating authority.

States govern the use and permitting of pits under their jurisdiction. Regulations vary from state to state regarding the circumstances in which pits can be used (e.g., chemical composition of the fluid), how they should be constructed, and whether they must be lined (e.g., proximity to drinking water resources and/or chemical composition of the fluid) (Richardson et al., 2013). Most states restrict the use of wastewater pits in environmentally sensitive areas. To avoid contamination events, some states are moving toward requiring closed loop systems (i.e., tanks) or injection wells rather than using pits for hydraulic fracturing wastewater storage. For example, Pennsylvania has modified their regulations (published October 8, 2016) to ban the use of pits for temporary storage of unconventional (as defined by PA DEP) wastewaters; many operators have already moved to closed-loop systems (PA DEP, 2016a). This development is particularly notable because of

Pennsylvania's heavy reliance on reuse for wastewater management, necessitating both on-site and off-site storage.

8.4.5.1 Locations and Numbers of Pits

The locations and number of existing pits (both for storage and for disposal) are not well documented in all states, and in the data found, pits associated with hydraulic fracturing operations were not specifically identified. With respect to larger pits for storage or disposal of wastewater, some states (e.g., Utah and Oklahoma) provide locational data on their websites. In 2016, the state of California began posting the number of active and inactive oil field produced water "ponds" (defined as unlined surface impoundments), both permitted and unpermitted, on their website. The July 2016 posting showed that 64% (682) of the 1,065 unlined ponds identified in the Central Valley and Central Coast of California were active. Of the active ponds, 21% (144) were not permitted (CA Water Board, 2016). Active ponds are primarily found in the southern San Joaquin Valley (CCST, 2015a). The EPA Region 8 conducted a survey of pits associated with oil and gas operations in Colorado, Montana, North Dakota, South Dakota, Utah, and Wyoming from 1996 through 2002. Results indicated there were approximately 28,000 pits at that time (U.S. EPA, 2003b).

In the absence of an inventory of pits in Pennsylvania, the organization SkyTruth led an effort using volunteers to produce a map of pits believed to be associated with drilling and hydraulic fracturing the Marcellus Shale (Manthos, 2014). The identification of pits was based on USDA aerial imagery taken in 2005, 2008, 2010, and 2013. SkyTruth acknowledges the uncertainties associated with identifying pits based on aerial images and volunteer labor. They have described their methodology as including multiple reviewers and QA/QC procedures. The study cannot differentiate ponds for drilling fluids and fracturing fluids from those for wastewater. Their preliminary findings indicate that the estimated number of ponds rose from 11 in 2005 to 529 in 2013, with the structures themselves increasing in size from a median size of 3,713 ft^2 (345 m^2) in 2005 to 66,844 ft^2 (6,210 m^2 in 2013. SkyTruth also notes that impoundments are not permanent and that of 581 ponds delineated in 2010, only 116 of them were found in the images from 2013.

Evaporation ponds, referred to as Commercial Oil Field Waste Disposal Facilities (COWDFs), are a waste management strategy most commonly used in the western states such as Utah, Wyoming, and Colorado (USFWS, 2014). According to a 2016 list of approved COWDFs posted by the Utah Division of Oil, Gas, and Mining (Utah Division of Oil, 2016), 20 facilities in Utah are approved to accept produced water. All are in the eastern part of the state where the Uinta and Paradox basins are found (unconventional shale formations). The Wyoming Department of Environmental Quality website, accessed in 2016, lists 35 active COWDFs (WDEQ, 2016b). The increase in hydraulic fracturing activity in Wyoming has resulted in significant increase in wastewater disposed of in COWDFs (USFWS, 2014). Data from the Colorado Oil & Gas Conservation Commission includes eight active evaporation pits, five of which are unlined (COGCC, 2016). Ninety-five other active pits are listed in Colorado, with descriptors such as "production," "multi-well pit," "skim," or "produced water." Seventy-one of these are unlined, and 22 have synthetic liners. Eleven pits are located in

Garfield County, where there is hydraulic fracturing activity. The Colorado data do not distinguish pits at centralized commercial facilities from on-site pits.

8.4.5.2 Unlined Storage Pits and Percolation Pits

Whether an unlined pit is designed and intended to percolate wastewater into the ground for disposal or if it is built for storage, it provides a pathway for wastewater to infiltrate into the subsurface and potentially reach groundwater. Such pits have been used historically for conventional oil and gas wastewater. More recently, they have received wastewater in areas where hydraulic fracturing takes place. States such as Montana and Wyoming allow unlined pits to be used for storage if the quality of the waste fluid meets specified limits and the pit is not in close proximity to environmentally sensitive areas such as drinking water resources, wetlands, and floodplains (Kuwayama et al., 2015b; Richardson et al., 2013).

In the past, several states have allowed unlined pits designed to dispose of wastewater via percolation into the subsurface. For example, until July 2015, percolation pits were permitted for wastewaters from hydraulically fractured wells in the Central Valley Region in California (Grinberg, 2016). The California Department of Conservation's Division of Oil, Gas, and Geothermal Resources (DOGGR) listed "evaporation-percolation" as the management method for almost 60% (190 million gal) of the wastewater generated via well stimulation in Kern County between 2011 and 2014 (CCST, 2015a). However, according to DOGGR's 2015 report addressing well stimulation activities in Kern County from January 1, 2014 through September 30, 2015, evaporation/percolation was not employed as a disposal option during that period (98% of the produced water was disposed of via operator-owned Class II injection wells, 1.75% was disposed of via commercial Class II injection wells, and 0.16% was reused).

While the practice of disposal via percolation pits has been discontinued in most states, as of July 2016, Wyoming's regulations still allow the use of percolation for disposing produced water specific to CBM operations in the Powder River Basin. To be permitted, the operator must demonstrate that the disposed fluid will comply with water quality standards of the Department of Environmental Quality (WYOGCC, 2015).

8.4.5.3 Evaporation Ponds

Evaporation is a simple water management strategy involving transporting wastewater to a pond or pit with a large surface area and allowing passive evaporation of the water from the surface (NETL, 2014; Clark and Veil, 2009). As discussed above, this disposal option, often referred to as a COWDF, is practical for drier climates of the western United States. Evaporation ponds have been used for oil and gas wastewater disposal in Montana, Colorado, Utah, New Mexico, and Wyoming (Veil et al., 2004). However, New Mexico no longer allows the use of pits for disposal (NM EMNRD OCD, 2013), and in Montana, evaporation ponds are no longer allowed because they do not put extracted water to a beneficial use (NRC, 2010). Figure 8-8 shows an example of a lined evaporation pit in Montana (DOE, 2006).

Figure 8-8. Lined evaporation pit in the Battle Creek Field (Montana).
Source: DOE (2006). Reproduced with permission from ALL Consulting.

As the water component of the wastewater is subject to evaporation, the fluid remaining in the pond becomes concentrated, and a sludge layer is formed. Remaining residual brines in the pond can be collected and disposed of via an underground injection well, and the solids can be taken to a landfill (see Section 8.4.7 for more details). In cold, dry climates, a freeze-thaw evaporation method has also been used to purify water from oil and gas wastewater (Boysen et al., 1999).

Nowak and Bradish (2010) describe the design, construction, and operation of two large commercial evaporation facilities in Southern Cross, Wyoming and Danish Flats, Utah. Each facility includes 14,000 gal (53,000 L) three-stage concrete receiving tanks, a sludge pond, and a series of five-acre (20,234 m^2) evaporation ponds connected by gravity or force-main underground piping. The Wyoming facility, which opened in 2008, consists of two ponds with a total capacity of approximately 84 million gal (2 million bbls or 318 million L). The Utah facility, open since 2009, consists of 13 ponds with a total capacity of approximately 218.4 million gal (5.2 million bbls or 826.6 million L). Each facility receives 0.42 to 1.47 million gal (10,000 to 35,000 bbls; 1.59 million to 5.56 million L) of wastewater per day from oil and gas production companies in the area.

Evaporation ponds or pits are subject to state regulatory agency approval and must meet state standards for water quality and quantity (Boysen et al., 2002).

8.4.5.4 Impacts and Potential Impacts from Pits and Impoundments

Pits containing hydraulic fracturing wastewater have the potential to impact drinking water resources if spills and overflows cause runoff to surface water or if wastewater percolates through the soil and reaches groundwater. In addition to contaminants in the wastewater itself, wastewater that reaches groundwater may mobilize constituents in pit bottoms or soils, and it may also reach hydrologically connected surface water. These impacts are amplified with increasing pit/impoundment size (Quaranta et al., 2012). Percolation may be accidental (through tears or improper installation of liner) or by design in unlined pits (Sumi, 2004).

Compromised pit liners can result in leaks, and extreme weather events, such as floods, can cause pits to overflow. An analysis of three state databases (New Mexico, Oklahoma, and Colorado) where pits and tanks have been used for storage of hydraulic fracturing wastewater found that for pits, the most common causes of spills were from overflows and liner malfunctions (Kuwayama et al., 2015b). For instance, of the 106 pit-related spills reported in New Mexico between 2000 and 2014, 33% were due to overflows and 26% were caused by liner malfunctions. Of the 62 tank spills reported, 44% were due to leaks, and 27% were related to overfilling (Kuwayama et al., 2015b). The types of constituents in pits that may be of concern from such events include VOCs, metals, TDS, oil, and TENORM (Kuwayama et al., 2015b).

Operational factors also influence potential impacts from pits and impoundments. These can include water level management (influent, seepage, spillage), the length of time water is stored in the pit/impoundment, the composition of the water, the local climate (rainfall and/or evaporation), and the transmission method (piped or delivered in an open channel) (NRC, 2010).

Construction and Capacity Issues

Construction requirements typically include specifications for features that can reduce the potential for impacts on groundwater or surface water. These can include liner specifications, depth to groundwater, secondary containment, setback requirements, freeboard, leak detection, and water quality monitoring (Kuwayama et al., 2015b).[1,2] For example, in a 2012 review of 19 states with shale gas development or potential for shale gas development, many states had setback requirements for pits in sensitive areas including surface water, wetlands, and floodplains. As of December 2015, however, 12 of the 19 states surveyed did not include setback requirements in their regulations. Many states did address the vertical separation of pits from the water table (e.g., 20 in (0.5 m) to seasonal high water table in PA; 10 ft (3 m) in WY; 50 ft (15 m) in NM) (Kuwayama et al., 2015b).

Despite construction standards, impacts on groundwater or surface water due to overflows, liner breaches, and other construction issues have been documented. In 2007 in Knox County, Kentucky, retention pits holding hydraulic fracturing flowback fluids overflowed into Acorn Fork Creek during the development of four natural gas wells (CCST, 2015a; Papoulias and Velasco, 2013). The incident caused the pH of the creek to drop from 7.5 to 5.6 and the conductivity to increase from 200 to 35,000 μS/cm. In addition, organics and metals including iron and aluminum formed precipitates in the stream. Fish and aquatic invertebrates were killed or distressed in the area of the stream affected by the release (Papoulias and Velasco, 2013).

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[1] Setback is the distance between the pit and a stream, lake, building, or other feature or structure that needs protection.

[2] Freeboard is the vertical distance between the level of the water in an impoundment and the overflow elevation (an outfall or the lowest part of the berm).

Similarly, in 2009, Marcellus wastewater stored in an impoundment from a hydraulic fracturing operation in Washington County, Pennsylvania overflowed the bank of the impoundment and reached surface water (a tributary of Dunkle Run) (CCST, 2015a). NRC (2010) reported continuous overfilling of an impoundment in the Powder River Basin (Wyoming) with CBM produced water, resulting in significant erosion of a seasonal water channel. The CBM operator was required through litigation to manage flows to the impoundment to prevent overflows. The literature did not report specific impacts on groundwater or surface water from the Pennsylvania or Wyoming incidents.

In Pennsylvania in 2010, pit liner failure was reported to have impacted groundwater through leakage of Marcellus wastewater from six impoundments (Colaneri, 2014). Ziemkiewicz et al. (2014) note that a study of 15 pits and impoundments in West Virginia found that slope stability and liner deficiencies were common problems. Construction quality control and quality assurance were often inadequate; the authors found a lack of field compaction testing, use of improper soil types, excessive slope lengths, buried debris, and insufficient erosion control, although no breaches were reported. A statistical analysis of oil and gas violations in Pennsylvania found that structurally unsound impoundments or inadequate freeboard were the second most frequent type of violation, with 439 instances in the period from 2008 to 2010 (Olawoyin et al., 2013).

Unlined Pits

Impacts on groundwater from historic and current uses of unlined pits in the oil and gas industry have been documented. In a review of records spanning 25 years (1983 – 2007), 63 incidents of private water supply contamination from the infiltration of saline fluids from unlined or inadequately constructed reserve pits were identified in Ohio (Kell, 2011). The same study (Kell, 2011) identified 57 legacy (pre-1984) incidents in Texas involving groundwater contamination from unlined produced water disposal pits. Such pits were phased out in Texas by 1984, prompting a move towards disposal of oil and gas wastewater in disposal wells.

Kern County, California has experienced impacts on groundwater associated with unlined percolation pits. A 2014 study notes that there are hundreds of pits across Kern County and elsewhere in the state, stretching state resources for regulatory oversight (Grinberg, 2014). Past sampling of water in percolation pits has shown exceedances of California's Tulare Lake Basin Plan (Basin Plan), which specifies maximum levels permitted for discharges of oil field well wastewater to unlined ponds overlying groundwater (Grinberg, 2014).[1] For example, the McKittrick 1 and 1-3 pits are large percolation pits in Kern County near oil fields where most of the hydraulic fracturing in California takes place (Grinberg, 2014). The pits are situated close to a number of important resources. They are located within a few miles of the Kern River Flood Channel, the California State Water Project, farmland, and are in an area of high quality groundwater (Grinberg, 2014). Sampling of fluids in the pits dating back to 1997 showed consistent exceedances of Tulare Basin Plan standards for TDS, chlorides, and boron. Sampling also revealed the presence of BTEX, gasoline range organics (GRO), and diesel range organics (DRO) (MTA, 2014). Sampling of three monitoring

[1] The Basin Plan sets limits for salinity (1,000 μmhos/cm measured as electrical conductivity), chloride (175 mg/L), and boron (1 mg/L) (California Regional Water Quality Control Board Central Valley Region, 2015).

wells indicated that in 2004, a plume had migrated at least 4,000 ft (1,000 m) from the pits and was still detected in test wells in 2013. As of July 1, 2015, California's Code of Regulations includes a provision that no longer allows the use of pits, including percolation pits, for fluids produced from stimulated wells (Grinberg, 2016).

Unlined pits that were used from the 1960s until the mid-1990s for disposal of drilling muds and flowback and produced waters associated with hydraulic fracturing operations have been linked to groundwater contamination in Pavillion, Wyoming (Digiulio and Jackson, 2016; AME, 2015). A report by the Wyoming Oil and Gas Conservation Commission (WYOGCC) (WYOGCC, 2014a) summarizes site investigations and reclamation activities conducted by WOGCC, the Wyoming Department of Environmental Quality (WDEQ), and Encana Oil and Gas for pits in the Pavillion Well Field. The report includes information on samples collected between 2006 and 2013 from shallow groundwater in the vicinity of the pits. Some sites had detections for one or more of the following contaminants: GRO, DRO, BTEX, and/or naphthalene. Of the shallow groundwater sites with detections, some were associated with pits located within one-quarter mile of a domestic well. One of these sites exceeded clean-up levels established by the WDEQ Voluntary Remediation Program for DRO (13,000 µg/L) and benzene (110 µg/L).[1] The report noted that there was insufficient evidence to determine whether or not drinking water supply wells in the vicinity of the pits were contaminated by disposal of hydraulic fracturing wastewater in those pits (WYOGCC, 2014a).

Other examples in the literature include the detection of VOCs in groundwater downgradient of an unlined pit containing oil and gas wastewater near the Duncan Oil Field in New Mexico (Sumi, 2004) (Section 8.5). Groundwater impacts downgradient of an unlined pit in Oklahoma included high salinity (3500-25,600 mg/L) and the presence of VOCs (Kharaka et al., 2002). Neither New Mexico nor Oklahoma currently allows unlined pits for disposal or storage (OCC OGCD, 2015; NM EMNRD OCD, 2013).

Mobilization and Transport of Constituents

Groundwater impacts may result not just from constituents in the wastewater but also from mobilization of existing constituents in the soil or sediment. A CBM produced water impoundment in the Powder River Basin of Wyoming was studied for its impact on groundwater (Healy et al., 2011; Healy et al., 2008). Infiltration of water from the impoundment was found to create a perched water mound in the unsaturated zone above bedrock in a location with historically little recharge. Elevated concentrations of TDS, chloride, nitrate, and selenium were found at the site, with one lysimeter sample exceeding 100,000 mg/L of TDS (Healy et al., 2008). Most of the solutes found in the groundwater mound did not originate with the CBM produced water, but rather were the consequence of dissolution of previously existing salts and minerals (Healy et al., 2011).

Generally, the deeper that wastewater can move into an aquifer, as impacted by the volume and timing of the release, the longer the duration of contamination (Whittemore, 2007). Kharaka et al. (2007) reported on studies at a site in Oklahoma with one abandoned and two active unlined pits.

[1] WDEQ cleanup levels are derived from a combination of promulgated levels (MCL, state-assigned water quality standards) and risk-based cleanup level concentrations (WDEQ, 2016a).

Produced water from these pits penetrated 10 to 23 ft (3 to 7 m) thick shale and siltstone units, creating three plumes of high-salinity water (5,000 to 30,000 mg/L TDS). The impact of these plumes on the receiving water body (Skiatook Lake) was judged to be minimal, although the estimate was based on a number of notably uncertain transport quantities (Otton et al., 2007).

Vadose (unsaturated) zone transport was illustrated at a site in Oklahoma where two abandoned pits were major sources for releases of produced water and oil. Saline water from the pits flowed through thin soils and readily percolated into underlying permeable bedrock. Deeper, less-permeable bedrock was contaminated by salt water later in the history of the site, presumably due to fractures. The mechanisms proposed were vertical movement through permeable sand bodies, lateral movement along shale fractures, and possibly increased clay permeability due to the presence of highly saline water (Otton et al., 2007).

Summary

Collectively, the above examples show that regardless of the purpose of pits (storage or disposal), they present a potential pathway for wastewater constituents to impact groundwater or surface water. Good construction standards and practices, including liners, adequate freeboard, and setbacks, are important for minimizing potential impacts on both surface water and groundwater. Proper monitoring and maintenance (e.g., avoiding overfilling, maintaining the integrity of liners and berms) are also important for protecting surface water and groundwater. Unlined pits, in particular, can lead to groundwater contamination. This can be long-lasting, as evidenced by legacy impacts from older pits. Most states have phased out unlined disposal pits and unlined storage pits, but if such pits are still in use, they can provide ongoing potential sources of groundwater contamination (CCST, 2015a; Grinberg, 2014).

8.4.6 Other Management Practices and Issues

Additional strategies for wastewater management in some states include directly discharging to surface waters and land application. In particular, wastewater from CBM fracturing and production generally has lower TDS concentrations than wastewater from other types of unconventional formations and more readily lends itself to other uses.

8.4.6.1 Land Application and Road Spreading

Road spreading has been used as a disposal option for high-TDS wastewaters (brines) from conventional oil and gas production. Road spreading can be done for dust control and de-icing. Although recent data are not available, an American Petroleum Institute (API) survey estimated that approximately 75.6 million gal (1.8 million bbls or 286.2 million L) of wastewater was used for road spreading in 1995 (API, 2000). The API estimate does not specifically identify hydraulic fracturing wastewater. There is no current nationwide estimate of the extent of road spreading using hydraulic fracturing wastewater.

Road spreading with hydraulic fracturing wastewater is regulated primarily at the state level (Hammer and VanBriesen, 2012) and is prohibited in some states. For example, with annual approval of a plan to minimize the potential for pollution, PA DEP allows spreading of brines from conventional (as defined by PA DEP) wells for dust control and road stabilization. Hydraulic fracturing flowback, however, cannot be used for dust control and road stabilization (PA DEP, 2011b). In West Virginia, use of gas well brines for roadway de-icing is allowed per a 2011 memorandum of agreement between the West Virginia Division of Highways and the West Virginia Department of Environmental Protection, but the use of "hydraulic fracturing return fluids" is not permitted (Tiemann et al., 2014; West Virginia DEP, 2011).

Concerns about road application center on contaminants such as barium, strontium, and radium. A report from PA DEP analyzed several commercial rock salt samples and compared results with contaminants found in Marcellus Shale flowback samples. The results noted elevated barium, strontium, and radionuclide levels in Marcellus Shale brines compared with commercial rock salt (Titler and Curry, 2011). Another study found increases in metals (radium, strontium, calcium, and sodium) in soils ranging from 1.2 to 6.2 times the original concentrations (for radium and sodium, respectively), attributed to road spreading of wastewater from conventional oil and gas wells for de-icing (Skalak et al., 2014).

Potential impacts on drinking water resources from road spreading have been noted by Tiemann et al. (2014) and Hammer and VanBriesen (2012). These include potential effects of runoff on surface water and migration of brines to groundwater. Snowmelt can carry salts and other chemicals from the application site, and transport can increase if application rates are high or rain occurs soon after application (Hammer and VanBriesen, 2012). Research on the impacts of conventional road salt application has documented long-term salinization of both surface water and groundwater in the northern United States (Kelly, 2008; Kaushal et al., 2005). When conventional oil field brine was used in a controlled road spreading experiment, elevated chloride concentrations were detected in shallow groundwater (Bair and Digel, 1990). The amount of salt attributable to road application of hydraulic fracturing wastewaters has not been quantified.

To evaluate land application of solid wastes from oil and gas production, a laboratory study mimicking land spreading of conventional oilfield scales and sludges indicated that 20% of the radium in barite sulfate scales was released by microbial processes during incubation with soil (Matthews et al., 2006; Swann et al., 2004). Although the radium was then complexed with the soil, it would be more mobile and more bioavailable than when it was associated with the barite. Overall, potential effects on drinking water resources from land spreading are not well understood, including the amounts of hydraulic fracturing wastes that are managed by land spreading.

8.4.6.2 Management of Coalbed Methane Wastewater

Many, but not all, CBM wells are hydraulically fractured to enhance recovery, using fluids that range from water alone to more complex gel formulations with proppant (e.g., Engle et al., 2011; McCartney, 2011; NRC, 2010; Halliburton, 2008; U.S. EPA, 2004a). The literature indicates that hydraulic fracturing of CBM formations is being conducted in the San Juan, Raton, Piceance, and Uinta Basins, among others. Literature such as NRC (2010) notes that hydraulic fracturing may not be common in the Powder River Basin. Additionally, when CBM well stimulation does take place, it can be accomplished using very simple hydraulic fracturing fluid formulations (Chapter 3).

Wastewater from CBM wells can be managed like other hydraulic fracturing wastewater discussed above. However, the wastewater from CBM wells can also be of higher average quality (typically lower TDS content) than wastewater from other hydraulically fractured wells. The lower TDS content makes it more suitable for certain management practices and uses. A number of management strategies have been proposed or implemented, with varying degrees of treatment required depending on the quality of the wastewater and the intended use (Hulme, 2005; DOE, 2003, 2002). Although specific volumes managed through the practices discussed below are not well documented, qualitative information and considerations for feasibility are available and presented. The discussion below covers both dilute and higher-TDS wastewater from CBM formations.

The quality of CBM wastewater plays a large role in how the wastewater is managed. The TDS content can range from an average of nearly 1,000 mg/L in the Powder River Basin to an average of about 14,000 mg/L (and as high as approximately 62,000 mg/L) in the Black Warrior Basin (Appendix Table E-3). Data sources from about 2002 through 2008 indicate that operators in some basins such as the San Juan, Uinta, and Piceance, and Raton (in New Mexico), where TDS is typically higher compared to other basins (e.g., Powder River), manage most wastewater by injection into disposal wells (NRC, 2010; U.S. EPA, 2010a).

Discharge to rivers and streams, a management option governed by the CWA, may be permitted in cases where wastewater is of high quality.[1] To be discharged, the wastewater must meet technology-based effluent limitations established by the permitting authority on a case-by-case “best professional judgment” basis as well as any more stringent limitations necessary to meet applicable water quality standards. For example, as a means of protecting high-quality waters of the state, the Montana Supreme Court ruled in 2010 that treatment is required for all CBM produced water prior to discharge to surface water (NRC, 2010).

A 2008 EPA survey of CBM operators found that of the projects represented in the results, direct discharge to surface water was by far most prevalent in the Powder River Basin but was also reported as a management practice in the Green River, Raton, Black Warrior, Cahaba, Illinois, and Appalachian basins (U.S. EPA, 2013e, 2010a).[2] Discharges to surface water can provide habitat maintenance, restoration of wildlife-waterfowl fishery habitat, and flow augmentation to benefit downstream water users. However, hydrologic changes from such discharges could also have unanticipated effects on ecosystems previously adapted to intermittent streamflow.

Some CBM wastewater can be put to agricultural use, including livestock and wildlife watering, and crop irrigation. Livestock watering with CBM wastewater can be done using on-channel or off-channel impoundments, and irrigation is an area of active research (e.g., Engle et al., 2011; NRC, 2010). However, wastewater from some higher-salinity CBM basins (e.g., San Juan, Uinta, and Piceance) would need blending or treatment before such uses. Irrigation with treated CBM

[1] Although discharge to rivers and streams is generally prohibited under the EPA’s oil and gas ELGs, the ELGs do not apply to CBM.
[2] These reports did not describe certain non-discharging wastewaters management strategies in basins with few operators in order to preserve CBI. The reports also do not provide information on hydraulic fracturing activities in the basins. Not also that results are presented by numbers of projects, which may vary in the number of wells they contain.

wastewater would be most suitable on coarse-textured soils for cultivation of salt-tolerant crops (DOE, 2003). NRC (2010) remarks that "use of CBM produced water for irrigation appears practical and sustainable," provided that appropriate measures are taken such as selective application, dilution or blending, appropriate timing, and rehabilitation of soils.

Although CBM wastewater is generally lower in TDS than wastewater associated with shale gas development, it can still have higher TDS concentrations than stream water. This poses concerns regarding the sodium adsorption ratio (SAR) for agricultural soils. A USGS study performed trend analysis of water quality at sampling sites in the Tongue and Powder River watersheds (Powder River Basin) (Sando et al., 2014). One of the study objectives was to determine possible effects of CBM produced water particularly in areas where the water was discharged to impoundments or upper reaches of in-stream channels for infiltration. Trend analysis showed potential effects of CBM production on downstream water quality (increases in sodium, alkalinity, and SAR) in the main-stem Powder River but found mixed results at the Tongue River sites (some appeared to be impacted by CBM activities while others did not) (Sando et al., 2014).

Sando et al. (2014) found that CBM pumping rates (i.e., discharge of produced water) were high relative to streamflow in the Powder River Basin. For the three main-stem Powder River sites, the CBM pumping rates were 26-34% of the 2001-2010 median streamflows. For one site in the Little Powder River watershed, the CBM pumping rate was 360% of 2001-2010 median streamflow. This underscores that in arid climates in the western United States, permitted discharges from CBM activities (whether hydraulically fractured or not) at a particular site may be large relative to the size of the receiving water and may sometimes dominate flows.

As noted above, a degree of treatment is needed (or required) for some uses. Plumlee et al. (2014) examined the feasibility, treatment requirements, and potential costs of several hypothetical uses for CBM wastewater. In several cases, costs for these uses were projected to be comparable to or less than estimated disposal costs. In one case study, use of CBM wastewater for streamflow augmentation or crop irrigation could potentially cost between \$0.26 and \$0.27 per bbl. For comparison, reported disposal costs in 2000-2001 ranged from \$0.01 per bbl for a pipeline collection system with impoundment to \$2.00 per bbl for hauling to disposal or treatment. The 2010 NRC report (NRC, 2010) noted that 15 to 18% of CBM produced water in the Powder River Basin was being treated to reduce SAR in order to satisfy NPDES permits for discharge.[1] If wastewater is treated to address SAR, reported costs are approximately \$0.12 to \$0.60/bbl (NRC, 2010).

The applicability of particular uses may be limited by ecological and regulatory considerations as well as the irregular nature of CBM wastewater production (voluminous at first, and then declining and halting after a period of years). Legal issues, including overlapping jurisdictions at the state level and senior water rights claims in over-appropriated basins (in western states) can also determine the use of CBM wastewater (Wolfe and Graham, 2002).

[1] SAR is the relative proportion of sodium to other cations in water. It is also an indication of risk to soil from alkalinity. The higher the SAR, the less suitable the water is for irrigation, and long-term use can damage soil structure.

8.4.6.3 Other Documented Uses of Hydraulic Fracturing Wastewater

Uses of wastewater from shales or other hydraulically fractured formations face many of the same possibilities and limitations as those associated with wastewater from CBM operations. The biggest difference is in the quality of the water. Wastewaters vary widely in water quality, with TDS values from shale and tight sand formations ranging from less than 1,000 mg/L TDS to hundreds of thousands of mg/L TDS (DOE, 2006) (Chapter 7). Wastewaters on the lower end of the TDS spectrum could be reused in many of the same ways as CBM wastewater, depending on the concentrations of potentially harmful constituents and applicable federal, state, and local regulations. High TDS wastewaters have more limited uses, and pre-treatment may be necessary (Shaffer et al., 2013; Guerra et al., 2011; DOE, 2006). Agricultural and wildlife uses are subject to the produced water daily effluent discharge limit of 35 mg/l for oil and grease.[1]

Potential uses for wastewater in the western United States include livestock watering, irrigation, streamflow supplementation, fire protection, road spreading, and industrial uses, with each having their own water quality requirements and applicability (Guerra et al., 2011). Guerra et al. (2011) summarized the least conservative TDS standards for five possible uses in the western United States that include 500 mg/L for drinking water (the drinking water secondary maximum contaminant level (SMCL)), 625 mg/L for groundwater recharge, 1,000 mg/L for surface water discharge, 1,920 mg/L for irrigation, and 10,000 mg/L for livestock watering. The authors estimated that wastewater from 88% of unconventional wells in the western United States could be used for livestock watering without TDS removal based on a maximum TDS concentration of 10,000 mg/L. However, wastewater from only 10% of unconventional wells could be used for surface discharge without treatment for TDS based on the least conservative standard among the western states of 1,000 mg/L TDS (Guerra et al., 2011). Guerra et al. (2011) indicate that in several basins in the western United States (e.g., Wind River, Green River, and Powder River), wastewater from 50% or more of oil and gas wells is suitable for agricultural use. In other basins (e.g., San Juan, Piceance, and Permian) over 50% of oil and gas wastewater is unsuitable for use without treatment. A 2006 Department of Energy (DOE) study pointed out that the quality necessary for use in agriculture depends on the plant or animal species involved and that in the Bighorn Basin in Wyoming, low-salinity wastewater is used for agriculture and livestock watering after minimal treatment to remove oil and grease (DOE, 2006).

Although TDS is a common criterion for water quality, there are also recommended limits or considerations for some metals, alkalinity, and nitrate in water for use in livestock watering, and for metals, SAR, electrical conductivity (ECw), and pH for water for irrigation (Guerra et al., 2011). Also, using TDS/salinity as the primary criterion may not be appropriate if wells contributing to the produced water have undergone hydraulic fracturing or if maintenance chemicals are being used on the well.

The water quality standards and monitoring requirements for direct discharge for use in irrigation or livestock watering include few specifications. In California, the California Council on Science and Technology (CCST, 2015a) notes that the testing and treatment required by the regional water

[1] 40 CFR 435.52(b).

quality control boards prior to use of produced water for irrigation do not include assessment for chemicals associated with hydraulic fracturing and that there are no policies prohibiting the use of hydraulic fracturing wastewaters for irrigation.

In the Wind River Basin in Wyoming, three NPDES permits were appealed by environmental groups due to concerns that the permits failed to address maintenance and hydraulic fracturing chemicals (Natural Resources Defense Council, 2015; PEER, 2015). The environmental groups argued that the EPA's regulations do not allow for the discharge of produced water containing chemicals from well treatment, and that, moreover, the EPA lacked sufficient information regarding the well treatment chemicals to determine whether the discharge would be "good enough quality" for wildlife and agricultural use, as required under the ELG regulations. As an example, the environmental groups pointed to MSDS information provided upon request for six maintenance products, which included toxic chemicals such as ethylene glycol, benzyl chloride, isopropanol, naphthalene, benzene, and xylene, among others. This raised concerns that produced water permitted for direct discharge may contain toxic chemicals or their degradation products. Ultimately, pursuant to a settlement agreement with the environmental groups and permittees, the EPA issued modified permits that included additional conditions for handling of and reporting about well stimulation and well maintenance chemicals.

8.4.7 Management of Solid and Liquid Residuals

Solid and liquid residuals associated with hydraulic fracturing wastewater are formed from treatment processes at CWTs, buildup of sludges in tanks and pits, and scale formation on pipes and equipment. These residuals must be managed and disposed of properly to avoid impacts on ground and surface water resources. (Note that drill cuttings and drilling muds are outside the scope of this chapter.)

8.4.7.1 Solid Residuals

The solid residuals produced at a CWT depend on the constituents in the untreated water and the treatment processes used and are likely to contain TSS, TDS, metals, radionuclides, and organics. Solid residuals can consist of sludges (from precipitation, filtration, settling units, and biological processes), spent media (filter media, adsorption media, or ion exchange media), and other material such as spent filter socks used to remove gross particulates. In addition, solids that accumulate in storage tanks and pits and scale that deposits on equipment are part of the residual load from a site. These residuals can constitute a considerable fraction of solid waste in an oil or gas production area.

Handling and disposal of residual sludges from treatment processes can present some of the biggest challenges associated with these technologies (Igunnu and Chen, 2014). Additional treatment may be applied to solid residuals including thickening, stabilization (e.g., anaerobic digestion), and dewatering processes prior to disposal. The solid residuals are then typically sent to a landfill, land spread on-site, or incinerated (Morillon et al., 2002). Land spreading is a waste management method in which wastes are spread over the soil surface and tilled into the soil to allow the hydrocarbons in the wastes to biodegrade (Smith et al., 1998); note that inorganic constituents (e.g., salts, metals, metalloids, and radionuclides) will not degrade. In addition, pits or impoundments that have reached the end of their useful life have accumulated residuals. Practices used to decommission these pits include draining and leveling the pit in place or land farming the residual materials into the ground (Rich and Crosby, 2013), although more information is needed on the potential for these practices to affect water resources.

A particular concern for the management of residual wastes is TENORM that originates from the geologic formation and was present in the produced water (SAFER PA, 2015). Studies have found TENORM in solid residuals at oil and gas operations including the filter cake (PA DEP, 2015b), filter socks (Harto et al., 2014), and pit sludges (Rich and Crosby, 2013). Researchers have assessed Marcellus produced water samples, finding that many with low barium and high radium-226 levels would generate sludges that exceed the maximum acceptable radium-226 activity for nonhazardous landfill disposal in Pennsylvania (Silva et al., 2014b; Silva et al., 2014a). In scales that build up on hydraulic fracturing and treatment equipment and sludges that accumulate in tanks and pits, radium can coprecipitate with barium, strontium, or calcium sulfates (Smith et al., 1999). (See Section 8.5.2 for additional discussion of TENORM associated with residuals.)

The accumulation of TENORM in the solids generated can limit or preclude landfills as a disposal option. Walter et al. (2012) point out that wastes containing TENORM can be problematic due to the possibility of radon emissions from the landfill. Regulatory limits on permissible radionuclide levels accepted at non-hazardous landfills vary by state (Silva et al., 2014a).[1] Some states have volumetric limitations on TENORM in their landfill permits (e.g., Colorado). Also, some states write criteria, such as gamma exposure rates (radiation) and radioactivity concentration limits, into permits for many landfills that are permitted to accept TENORM. Silva et al. (2014a) note that there are 50 nonhazardous (RCRA-D) disposal facilities in Pennsylvania, but no TENORM disposal facilities. Texas and other states have disposal facilities for TENORM.

8.4.7.2 Liquid Residuals

Liquid residuals include concentrated brines (from membrane or evaporation processes) and regeneration or cleaning chemicals (from ion exchange, adsorption, and membrane processes) (Fakhru'l-Razi et al., 2009). Practices for managing liquid residual streams from treatment processes are generally the same as for untreated hydraulic fracturing wastewaters, although the treated volumes are smaller, resulting in lower costs (Hammer and VanBriesen, 2012). Concentrations of contaminants in liquid residuals, however, will be higher. The most common disposal method is injection into disposal wells.

If the liquid is not injected into a disposal well, treatment to remove salts would be required for surface water discharge to meet NPDES permit requirements and protect the water quality for downstream users such as drinking water utilities (Section 8.5). Because some constituents of concentrated liquid residual waste streams can pass through or impact municipal wastewater treatment processes (Linarić et al., 2013; Hammer and VanBriesen, 2012), these residuals would

[1] Examples of permissible radionuclide levels at non-hazardous landfills: Pennsylvania requires alarms to be set at all municipal landfills, with a trigger set at 10 μR/hr above background radiation. Texas sets a radioactivity limit, requiring that any waste disposed by burial contains less than 30 pCi/g radium or 150 pCi/g of other radionuclides.

not be appropriate for discharge to a POTW. Elevated salt concentrations, in particular, can have detrimental effects on microbiological treatment at municipal wastewater systems, such as activated sludge treatment (Linarić et al., 2013).

Liquid residuals can also be mixed with a solidifying agent such as Portland cement and then disposed of in landfills, or they can undergo advanced treatment processes to generate products such as road salt or industrial chemicals (SAFER PA, 2015).

8.4.7.3 Potential Impacts from Solid and Liquid Residuals

Residual wastes have the potential to impact the quality of drinking water resources if contaminants leach to groundwater or reach surface water. In a recent study by PA DEP, radium was detected in leachate from 34 of 51 landfills that accept waste from the oil and gas industry (Marcellus in particular). Radium-226 concentrations ranged from 54 to 416 pCi/L, and radium-228 ranged from 2.5 to 1,100 pCi/L (PA DEP, 2015b). (See also Section 8.5.2 and see Chapter 9 for health effects associated with radium). Countess et al. (2014) studied the potential for a wide array of elements to leach from sludges generated at a CWT handling hydraulic fracturing wastewater in Pennsylvania. Tests used strong acid solutions (to simulate the worst case scenario) and weak acid digestions (to simulate environmental conditions). The data illustrate the possibility of leaching of these constituents from landfills. The extent of leaching varied by constituent and by fluid type, but the authors concluded that boron, bromide, calcium, magnesium, manganese, silicon, sodium, and strontium had high potential to migrate from the residual solids, with bromide and sodium having the highest leaching potential (Countess et al., 2014). (See also Section 5.8 in Chapter 5 for discussion of the processes governing the movement of constituents in the subsurface.)

In another study assessing the leaching behavior of residuals from hydraulic fracturing operations, Sharma et al. (2015) found that alkali metals, alkaline earth metals, and bromide had the highest leaching potential of the constituents tested. The authors also found that disposing of hydraulic fracturing residuals along with other solids (e.g., at a municipal landfill) produces a greater leaching potential than if the residuals are disposed of by burying or land disposal designed for solely the hydraulic fracturing residuals. This is due to the more acidic leachate formed at the co-disposal locations (Sharma et al., 2015).

Sang et al. (2014) studied the potential for hydraulic fracturing fluid to mobilize colloidal particles in the soil. The study used microspheres and sand particles as surrogates for contaminant particles. The authors note that the chemistry of hydraulic fracturing fluid favors transport of colloids and mineral particles through rock cracks, and they found that infiltration of flowback fluid can transport existing pollutants such as heavy metals, radionuclides, and pathogens, in unsaturated soils (Sang et al., 2014). Heavy metals can also move through soil. Although not specific to hydraulic fracturing wastes, Camobreco et al. (1996) report high levels of heavy metal transport in soil columns, with 12% recovery for lead, 15% for copper, 23% for zinc, and 30% for cadmium (Camobreco et al., 1996).

Residuals, whether liquid or solid, are the most concentrated wastes and waste streams associated with hydraulic fracturing operations. Contaminants in the produced water will accumulate in the sludges in storage tanks/pits, in scale on the equipment, and in treatment facilities. Proper management and disposal of these highly concentrated wastes is critical to minimize the potential for impacts on water resources.

8.5 Potential Impacts of Hydraulic Fracturing Wastewater Constituents on Drinking Water Resources

The previous section discussed the potential impacts of specific wastewater management strategies on drinking water resources. The severity of impacts, however, depends largely on the constituents in the wastewater, the concentrations of those constituents, and their health and ecological effects. This section will discuss the potential impacts of several specific types of hydraulic fracturing wastewater constituents on drinking water resources.

The impacts or potential impacts discussed in the literature are heavily focused on discharges from CWTs, including treated wastewater that is discharged indirectly through POTWs. Available evidence suggests that the effects of hydraulic fracturing on surface water quality are related to discharges of partially treated wastewater (Kuwayama et al., 2015a). Other avenues of contamination for both surface water and groundwater include leaks from pits and impoundments, landfill leachate, and leaching from contaminated sediments and other improperly managed solid wastes.

As noted, an important consideration regarding the potential impacts of hydraulic fracturing wastewater on receiving water is whether there are constituents of concern known to have health effects or that can give rise to compounds with health effects. See Chapter 9 for discussion of the health effects of wastewater constituents. For some classes of constituents, such as DBP precursors, considerable research exists regarding concentrations in the waste stream and impacts on downstream drinking water treatment plants or the finished drinking water after treatment. For other constituents, information is limited, especially within the context of hydraulic fracturing activities. There may also be unknown constituents because some ingredients in the original hydraulic fracturing fluids are claimed to be CBI. The following subsections identify several classes of constituents known to occur in hydraulic fracturing wastewater, discuss whether potential impacts are likely, and detail information gaps.

8.5.1 Bromide, Iodide, and Chloride

Halides, including bromide, chloride, and iodide, are commonly found in high-TDS hydraulic fracturing wastewater. As noted in Section 8.3.1.1, chloride is a regulated contaminant with a secondary MCL of 250 mg/L. Bromide and iodide are not regulated, but are of concern due to their role in the formation of DBPs (Parker et al., 2014; Krasner, 2009). (See Appendix F for information on DBP formation.) High-TDS wastewaters from the Marcellus Shale have been the focus of concern due to the state's history of treating these wastewaters at POTWs (without pretreatment) and at CWTs without TDS removal capabilities (Text Box 8-1). Discontinuing the practice of sending shale gas wastewater to POTWs without pretreatment (States et al., 2013), and compliance with the new EPA pretreatment standards for discharges of unconventional oil and gas wastewaters helps mitigate this problem. This section describes the role of some constituents in high-TDS fluids in the formation of DBPs and provides more details on the effects on surface waters as observed in Pennsylvania. The lessons learned and steps taken in the Marcellus region can provide valuable knowledge for operators and state agencies in other parts of the country that treat and discharge high-bromide and high-iodide wastewaters.

8.5.1.1 Influence of Bromide and Iodide on Formation of Disinfection Byproducts

Disinfection byproducts (DBPs) are formed when organic material comes in contact with disinfectants (e.g., chlorine, chloramine, chlorine dioxide, or ozone). Of particular concern are DBPs formed in the presence of halides (e.g., bromide or iodide). The type of DBP formed depends on the organic precursors in the source water and the disinfectant used. Regulated DBPs include total trihalomethanes (TTHM), five haloacetic acids (HAA5), bromate, and chlorite. There are, however, many additional DBPs that are not regulated and may in fact be of greater concern than the regulated species. Brominated forms of DBPs, for example, are considered to be more toxic and carcinogenic than chlorinated species (McGuire et al., 2014; Parker et al., 2014; States et al., 2013; Krasner, 2009; Richardson et al., 2007). Another halide, iodide is also found in some hydraulic fracturing wastewater (Chapter 7), and although its effects have not been as well documented as those associated with bromide, iodide raises some of the same concerns regarding formation of toxic DBPs as bromide (Xu et al., 2008).

Studies have found that elevated bromide levels in water correlate with increased DBP formation in the drinking water that is delivered to customers (also called "finished drinking water") (Obolensky and Singer, 2008; Matamoros et al., 2007; Hua et al., 2006; Yang and Shang, 2004). Harkness et al. (2015) studied the chemical composition of flowback, produced waters, treated wastewaters, instream flows downstream from discharges, and accidental spill sites. The study found high concentrations of bromide and iodide in the flowback and produced waters, concluding that the elevated levels of these constituents could promote the formation of toxic brominated and iodinated DBPs in downstream drinking water systems (Harkness et al., 2015).

In terms of the resulting DBP formation, laboratory experiments using hydraulic fracturing wastewater from the Marcellus and Fayetteville shales and river water from the Allegheny and Ohio rivers suggest that a relatively small portion of hydraulic fracturing wastewater can notably affect DBP formation (Parker et al., 2014). In particular, trihalomethanes (THM; a category of DBPs) were shown to shift towards greater brominated and iodinated species with a little as 0.01% hydraulic fracturing wastewater in disinfected samples. Modeling work by Landis et al. (2016) evaluated the impact of CWT discharges on DBP formation at a drinking water system and suggested that although only a 3% increase in overall TTHM formation was predicted, the model predicted a decrease in chlorinated THM and a substantial shift toward a higher percentage of the more-toxic brominated THMs (Landis et al., 2016).

States et al. (2013) found a strong correlation between bromide concentrations in source water from the Allegheny River in Pennsylvania and the percentage of brominated THMs in finished water at a drinking water facility using Allegheny source water. Bromide concentrations in the river water measured during the study ranged from less than 25 μg/L to 299 μg/L. The authors noted that source water containing 50 μg/L of bromide resulted in treated drinking water with approximately

62% of total THMs consisting of brominated species. When the source water contained 150 µg/L bromide, the brominated THM percentage was 83% (States et al., 2013).

Pope et al. (2007) reported that increased bromide levels are the second best indicator of DBP formation, with pH being the first. Furthermore, bromine (which may be formed from bromide in the water during disinfection) reacts as much as ten times faster and more efficiently with DBP precursors than chlorine (Westerhoff et al., 2004). These studies show that increased bromide concentration in a drinking water resource shifts the DBP formation towards more-toxic brominated forms.

If disinfection is accomplished using ozonation instead of or in addition to chloramination or chlorination, bromide and iodide in the source water can form two additional constituents: bromate and iodate. Iodate, although formed during disinfection by ozonation, is not considered a DBP and is non-toxic (Allard et al., 2013). Bromate, however, is a DBP of concern and has an MCL of 0.010 mg/L (U.S. EPA, 1998).

Another category of DBP that is not regulated is the nitrogenous DBPs, including nitrosamines. Data are lacking on the formation of nitrogenous DBPs specifically linked to hydraulic fracturing wastewater, but their formation is possible. During chloramination, bromide can enhance the formation of the nitrosamine N-nitrosodimethylamine (NDMA) in waters containing the precursor dimethylamine (DMA) (Le Roux et al., 2012; Luh and Mariñas, 2012). As with some other non-regulated DBPs, nitrogenous DBPs may be more toxic than the regulated ones (Harkness et al., 2015; McGuire et al., 2014; Parker et al., 2014).

As discussed in Section 8.4 and Text Box 8-3, removal of dissolved solids, including chloride and bromide, requires advanced treatment processes such as reverse osmosis (RO), distillation, evaporation, or crystallization. Unless the treatment plant receiving the high-TDS wastewater employs processes specifically designed to remove these constituents, effluent discharge may contain high levels of bromide and chloride. Drinking water systems with intakes downstream of these discharges may receive water with correspondingly higher levels of bromide and chloride and may have difficulty complying with SDWA regulations related to DBPs.

8.5.1.2 Effects on Receiving Waters

Studies show that discharges from oil and gas wastewater treatment facilities can elevate TDS, bromide, and chloride levels in receiving waters, and potential impacts may be detectable far downstream (> 1km) of an outfall (States et al., 2013; Warner et al., 2013a; Wilson and Van Briesen, 2013). The work by Landis et al. (2016) in the Allegheny River mentioned above is consistent with these findings. The authors studied the impacts of a CWT accepting oil and gas wastewater on water quality at a downstream drinking water intake. They found that compared to data from upstream (background) locations, bromide concentrations at the intake were increased by 53% at low streamflow and 22% during high streamflow.[1]

[1] Background samples are those taken from locations upstream of, and therefore unaffected by, permitted facilities.

Elevated TDS, chloride, and bromide can serve as indicators of potential influence from hydraulic fracturing wastewater in surface water and can also raise concerns about DBP formation in downstream drinking water systems. Elevation of bromide has been shown to place a burden on downstream drinking water systems. The Pittsburgh Water and Sewer authority (PWSA) drinking water system concluded that elevated bromide in their source water led to elevated TTHMs in their finished drinking water, with a substantial increase in the percentage of brominated TTHMs (States et al., 2013). The utility modified their treatment process and proposed improvements to their storage facilities to address the elevated TTHM levels in the distribution system (Chester Engineers, 2012).

Conversely, changes in regional wastewater handling that reduce bromide discharges can be reflected in receiving waters. A three-year study at water intakes downstream of wastewater discharges on the Monongahela River in western Pennsylvania evaluated water chemistry in the context of flow measurements. The authors concluded that an overall decrease in bromide concentrations at drinking water intakes from 2010 to 2012 was likely associated with shale gas operators voluntarily ceasing the practice of sending high-bromide wastewaters to treatment facilities that discharge to surface waters without adequate TDS removal (Wilson and Van Briesen, 2013).

Elevated TDS and halides need to be interpreted in the context of other inputs into a watershed. An EPA source apportionment study of the Allegheny River in Pennsylvania (U.S. EPA, 2015o) found that CWTs accounted for almost 90% of the bromide at one drinking water treatment plant intake and 37% of the bromide at another intake. Other sources include coal-fired power plants and acid mine drainage. Furthermore, although effluent is diluted when discharged to a water body, this may not always be sufficient to avoid water quality problems if there are existing pollutant loads in the waterbody from other contributors (e.g., such as acid mine drainage or power plant effluent) (Ferrar et al., 2013). Warner et al. (2013a) evaluated effluent from the Josephine Brine Treatment Facility, which treated both conventional and unconventional (as defined by PA DEP) oil and gas wastewater at the time of the study. The authors concluded that even a 500 to 3,000-fold dilution of the wastewater would not reduce bromide levels to background. Modeling by Weaver et al. (2016) suggests that bromide levels in receiving streams can be improved by reducing concentrations in the effluent, discharging during periods of high streamflow, and discharging intermittently (pulsing). (See Appendix F for additional description of modeling studies.)

In addition to concerns about formation of DBPs within downstream drinking water systems, treatment at the upstream CWTs and POTWs themselves can also produce DBPs if the facilities disinfect prior to discharge. The DBPs may then be released into receiving waters and increase concerns about the total loads of brominated and iodinated DBPs at downstream drinking water systems (Hladik et al., 2014). A study by Hladik et al. (2014) documented brominated and iodinated DBPs at the outfalls of CWTs and POTWs treating both conventional and unconventional wastewater and noted that this DBP signature was different than for those plants that did not accept oil and gas wastewater.

8.5.1.3 Other Constituents That Can Affect Downstream DBP Formation

In addition to halogens, organic matter and ammonium can also be present in hydraulic fracturing wastewater (Chapter 7; Appendix E) and can have an influence on the formation of DBPs at downstream drinking water systems (Harkness et al., 2015). Experimental work by Parker et al. (2014) found that a mixture of river water with 1-2% flowback by volume could contribute to DBP formation due to the higher dissolved organic carbon content of the flowback. Harkness et al. (2015) studied the chemical composition of water associated with oil and gas production and found high concentrations of ammonium in the flowback and produced waters. Elevated levels of ammonium can convert chlorine to chloramines at downstream drinking water treatment plants. This could have an impact on the plant's disinfection practices because chloramines are a weaker disinfectant than chlorine (Harkness et al., 2015; Parker et al., 2014).

8.5.1.4 Mitigating Impacts from TDS and Halides on Drinking Water Utilities

High bromide concentrations and low flow conditions in waterways have been shown to increase DBP formation in downstream drinking water systems (States et al., 2013). Most drinking water treatment plants are not designed to address high concentrations of TDS (including bromide and iodide), limiting their options for restricting the formation of brominated and iodinated DBPs when these halides are present.

To mitigate these impacts, one strategy that was implemented in Pennsylvania was to disallow influent of high-TDS wastewaters to POTWs and CWTs that discharged to streams and were not designed to treat for TDS. Wilson and Van Briesen (2013) showed that this strategy was effective for reducing bromide concentrations at drinking water utilities downstream from POTWs and CWTs that had formerly accepted hydraulic fracturing wastewaters (States et al., 2013; Warner et al., 2013a; Wilson and Van Briesen, 2013). Alternatively, advanced treatment processes such as reverse osmosis, distillation, evaporation, and crystallization, can be employed to reduce constituents that contribute to high TDS (e.g., such as chloride, bromide, and iodide), reducing impacts on surface waters and, subsequently, downstream drinking water utilities. Strategies such as discharging during higher streamflow periods and using a pulsing or intermittent discharge could also reduce the frequency and severity of potential impacts on drinking water systems from elevated TDS.

8.5.2 Radionuclides

Potential impacts on drinking water resources from TENORM associated with hydraulic fracturing wastewater can arise through a number of pathways, including: treated wastewater in which radionuclides were not adequately removed; accumulation of radionuclides in surface water sediments downstream of wastewater treatment plant discharge points; migration or mobilization from soils that have accumulated radionuclides from previous activities such as pits or land application; and inadequate management of treatment plant solids (such as filter cake), landfill leachate, or sediments in pits or tanks that have accumulated radionuclides.

An additional concern is the potential for underestimation of radium concentrations in hydraulic fracturing wastewater due to the high TDS content. When using wet chemical techniques, high TDS concentrations can result in poor recovery of some chemical constituents. For radium, recovery may be as low as <1% in a high-salt matrix (Nelson et al., 2014). This may lead to the inability to identify an impact on drinking water resources or an underestimation of the severity of an impact. Research suggests that spectroscopic methods are more appropriate for analysis of radium in high-TDS wastewaters (Nelson et al., 2014).

A recent study by the PA DEP (PA DEP, 2015b) provides information that helps fill a general data gap regarding TENORM content in oil and gas wastes that are treated and discharged to surface waters. The study, although not exclusive to Marcellus wastes, was motivated by concerns over an increase in radionuclides in oil and gas wastes observed during the expansion of Marcellus Shale production. The study began in 2013 and quantified radionuclide (radium-226, radium-228, K-40, gross alpha, and gross beta) levels at CWTs, POTWs, well sites, and landfills and discussed human health and environmental implications. Other relevant studies addressing radionuclides focus on CWTs that have handled Marcellus wastewater, evaluation of solids in storage pits, and analysis of scale on pipes and tanks.

8.5.2.1 Effluent from POTWs

In Pennsylvania between 2007 and 2010, TENORM-bearing wastewaters were sent to POTWs, which are generally not required to monitor for radioactivity (Resnikoff et al., 2010). Although management of Marcellus wastewaters via POTWs has declined, there is still potential for input of radionuclides to surface waters via discharge of CWT effluent either directly to surface water or indirectly through discharge to POTWs. The potential for TENORM to pass through treatment at POTWs is one of the concerns addressed in the EPA's recently promulgated pretreatment standards for unconventional oil and gas wastewaters that discharge to POTWs.

Six of the POTWs in the PA DEP TENORM study received effluent from a CWT along with municipal wastewater. Note that the CWTs in the study are not described as receiving exclusively Marcellus wastewater. The POTWs that receive both CWT effluent and municipal waste had radium in their effluent (overall average effluent radium-226 concentration of 103 pCi/L, with a range of <35 to 340 pCi/L). Those POTWs receiving only municipal wastewater also contained radium, with an average effluent radium-226 concentration of 145 pCi/L.[1] These concentrations are many times higher than the MCL for radium (5 pCi/L) and are also orders of magnitude higher than typical background values; radium-226 in river water generally ranges from 0.014 pCi/L to 0.54 pCi/L (IAEA, 2014).[2]

[1] These values are for unfiltered samples. In filtered samples, the POTWs that receive both CWT effluent and municipal waste had higher average radium-226 values than those for POTWs only treating municipal waste (497 pCi/L vs. 85 piCi/L). Filtered samples are passed through a filter to remove fine particles; concentrations of constituents in filtered samples are often lower than in unfiltered samples. However, liquid samples in this study were filtered after preservation with acid. Therefore, the difference between unfiltered and filtered samples may not be reliable.

[2] A confounding issue for this study is that it was not clear why the radium-226 concentrations were comparable or higher for those POTWs not receiving oil and gas CWT effluent. However, sample sizes were small and possible alternative sources for the radium were not discussed. The report also did not describe how it was verified that the POTWs did not receive contributions from oil and gas wastewater.

8.5.2.2 Effluent from CWTs

Four of the ten CWTs sampled during the PA DEP TENORM study (PA DEP, 2015b) discharged to surface water under a National Pollution Discharge Elimination System (NDPES) permit, and the other six discharged to POTWs. The average radium-226 concentration in the effluent from the CWTs (1,840 pCi/L for unfiltered samples) was an order of magnitude higher than in effluent from the POTWs. Samples of treated wastewater from zero-discharge facilities contained higher concentrations, averaging 2,610 pCi/L radium-226 and 295 pCi/L radium-228 (PA DEP, 2015b). The treated wastewater from these zero-discharge facilities will likely be reused for subsequent hydraulic fracturing jobs, postponing the need for disposal, but reuse could result in overall increases in some constituents of concern due to repeated passage through the subsurface. In addition, there is also a potential for impacts on drinking water resources from spills and leaks associated with wastewater storage and handling at these facilities.

Sampling done at the Josephine Brine Treatment Plant in western PA from 2010 – 2012 (Warner et al., 2013a) detected radium in the effluent (mean values of 4 pCi/L of radium-226 and 2 pCi/L of radium-228). Treatment at the facility removes radium by coprecipitation with barium sulfate. The authors note that if the activities of radium-226 and radium-228 in Marcellus brine influent at the CWT are similar to those reported by other researchers (Rowan et al., 2011), then the CWT achieved a 1,000-fold reduction in radium content. (This facility also accepted conventional oil and gas wastewater.) The detection of radium in the effluent from this CWT suggests that if the influent concentration is extremely high, radium will still be found in the effluent of a treatment plant even if the treatment process removes a high percentage (see Section 8.4 and Appendix F for additional discussion on constituent removal efficiencies at CWTs).

8.5.2.3 Accumulation in Sediments

In addition to concerns about TENORM in discharges to surface waters, studies have shown the potential for a legacy of radionuclide accumulation in surface water sediments. The PA DEP TENORM study (PA DEP, 2015b) found radium in sediments near the outfalls for CWTs (averages of 84.2 pCi/g and 19.8 pCi/g for radium-226 and -228, respectively) and three POTWs receiving treated oil and gas wastewater from CWTs (radium-226 and radium-228 concentrations ranging from 1.8 to 18.2 pCi/g). Typical background soil levels of radium are approximately 1 to 2 pCi/g (PA DEP, 2015b).

Warner et al. (2013a) measured radium-226 levels in stream sediment samples at the point of discharge of a CWT that had treated both conventional oil and gas wastewater and unconventional Marcellus wastewater. They found concentrations approximately 200 times greater than upstream and background sediments. This indicates the potential for accumulation of contaminants in localized areas near wastewater discharge facilities. Although the CWT studied by Warner et al. (2013a) also accepted conventional oil and gas wastewater, the authors observed that the radium-228/radium-226 ratio in the river sediments near the discharge (0.22 – 0.27) is consistent with ratios in Marcellus wastewater. The authors indicate that the radium likely accumulated in the sediments, originating from the discharge of treated unconventional Marcellus oil and gas wastewater. Accumulation of TENORM can also occur in sediments receiving discharged effluent from landfills that accept oil and gas wastes. In the PA DEP TENORM study (PA DEP, 2015b), samples of impacted soils were collected at three landfill outfalls. Radium-226 and -228 were detected in all samples (2.82 to 4.46 pCi/g and 0.979 to 2.53 pCi/g, respectively).

A study by Skalak et al. (2014), on the other hand, did not find elevated levels of alkali earth metals (including radium) in sediments just downstream of the discharge points of five POTWs that had previously treated Marcellus wastewater. These inconsistencies among studies suggest that accumulation of contaminants in sediment may depend on treatment processes and their removal rates for each constituent as well as stream chemistry and hydrologic characteristics. Contamination with radium-226 would potentially be long lived because of the long half-life of radium.[1]

The association of radium with sediments near discharge points is attributed to adsorption of radium to the sediments, a process governed by factors such as the salinity of the water and sediment characteristics. Increased salinity promotes desorption of radium from sediments, while lower salinity promotes adsorption, with radium adsorbing particularly strongly to sediments high in iron and manganese (hydr)oxides (Porcelli et al., 2014; Gonneea et al., 2008). Warner et al. (2013a) speculate that when saline CWT effluent is discharged into stream water, the lower salinity of the stream environment facilitates sorption of radium onto streambed sediments. The long-term fate of radium sorbed to sediments depends upon changes in water salinity and the sediment properties, including any reduction/oxidation chemical reactions that affect iron and manganese minerals in the sediments. Additionally, the sediment may be physically transported downstream due to high flows or if sediment is disturbed and resuspended.

8.5.2.4 Pits and Tanks

Where pits or impoundments are used, radionuclides may accumulate in the bottom sludges and can also be found in soils once the pit is closed and leveled. A study of three centralized wastewater storage impoundments in southwestern Pennsylvania (Zhang et al., 2015a) showed that radium-226 accumulated in various components of the bottom solids, including through coprecipitation with barium sulfate. Sludge from one pit showed a substantial increase in radium-226 between sampling events 2.5 years apart (from 8.8 pCi/g to 872 pCi/g). The authors attributed the steep increase to enrichment in radium during cycles of wastewater reuse. In Texas, accumulation of radionuclides (potassium, thorium, bismuth, radium, and lead) was documented for two pits that stored fluids associated with hydraulic fracturing (Rich and Crosby, 2013). One pit was decommissioned and used as farmland, and the other was active at the time of sampling. Analyses of soil and sludge samples detected a number of radionuclides, including radium-226, radium-228, thorium-228, strontium-90, and potassium-40 (radium-226 was only found at the former pit). Rich and Crosby (2013) note a total beta radiation value of 1,329 pCi/L in one sample from the active pit. They note that this value exceeded regulatory guidelines even though the values for individual

[1] The half-life of radium-226 is approximately 1,600 years, while the half-life of radium-228 is 5.76 years. The half-life is the time it takes for half of the nuclei in a sample of a radioactive element to decay. After two half-lives, one fourth of the original sample will be left, and after three half-lives there will be one eighth of the original sample remaining, and so forth.

radionuclides did not exceed regulatory guidelines, suggesting that using a single radionuclide (i.e., radium) as an indication of exposure can underestimate total radioactivity.

Although the sample sizes were small for both the Zhang et al. (2015a) and the Rich and Crosby (2013) studies, the results suggest that radionuclides associated with sediments from some pits could have potential impacts on surface water or groundwater. These studies illustrate the need for appropriate management where wastes have high TENORM content. Rich and Crosby (2013) note that pits are often found in agricultural regions. If pit solids that are incorporated into soils (e.g., during draining and leveling or during land application) contain radionuclides, they may reach surface water in runoff or leach from the solids and migrate to groundwater. In active pits, Rich and Crosby (2013) note that TENORM in the contents may be deposited onto crops and soil through aerosolization or breaching. The Pennsylvania study (Zhang et al., 2015a) suggests that landfill leachate may be affected by receiving sludges from impoundments that store produced water and will need to be managed appropriately.

With radium-226 values of 121 pCi/g and 872 pCi/g, sludges from the pits studied by Zhang et al. (2015a) exceeded the limit for disposal as a nonhazardous solid in a municipal or industrial solid waste landfill but would meet the radium-226 limits for disposal in a hazardous waste landfill. There are currently no federal requirements to test solid residuals for radionuclides before disposal. At landfills studied in the PA DEP TENORM report (PA DEP, 2015b), seven samples of treated effluent from nine facilities that accept oil and gas waste had radium-226 values ranging from 105 pCi/L to 378 pCi/L and radium-228 values ranging from <6 pCi/L to 1,100 pCi/L. Untreated effluent from the nine landfills had radium-226 contents ranging from 70 to <139 pCi/L. The study authors conclude that there is "limited potential" for environmental impacts from spills or discharges of leachate from these facilities.

Where wastewater is stored in tanks, TENORM concentrations can increase through radioactive ingrowth.[1] Radium-226 and radium-228 are generally considered the radionuclides of greatest concern in wastewaters and are the most frequently measured. But recent research indicates that in closed environments such as tanks, where the radium decay product radon cannot escape, total radioactivity may increase due to ingrowth of other decay products of radium such as Pb-210, Po-210, and Th-228 (Nelson et al., 2015). Experimental work by Nelson et al. found that concentrations of these decay products in Marcellus produced water that was stored in a sealed drum started growing immediately. Concentrations started at zero and reached 10.49 pCi/L for Po-210 and 155 pCi/L for Th-228 over the first 50 and 66 days of storage, respectively. The authors note that these decay products are not soluble, would be associated primarily with particulates, and could be bioavailable. This study demonstrates that analyzing for radium will not provide a complete indication of sample radioactivity if the water is stored in a closed environment and that subsequent management decisions would need to take into account possible increases in radioactivity due to ingrowth.

[1] The ingrowth, or growth within a sample, of radioactive daughter products from radionuclides initially present in the sample can cause greater radioactivity than that resulting from the parent radionuclides alone.

8.5.2.5 Other Solids

Other solid wastes associated with unconventional oil and gas production that may contain radionuclides include solid residuals from POTWs and CWTs and scale in oil and gas equipment. Filter cake samples from POTWs were found by PA DEP (2015b) to have highly variable radium-226 concentrations, with an average of 16 pCi/g, while typical soil concentrations in Pennsylvania have been found to be less than 2.5 pCi/g (Greeman et al., 1999). Filter cake from CWTs had an average radium-226 concentration of 111 pCi/g. The authors conclude that there could be impacts on surface waters through spills or effects on groundwater from landfill leachate containing contaminants originating in residuals sent to landfills.

Accumulation of TENORM-bearing scale in CWTs or POTWs may continue to affect the treatment plant even after discontinuing treatment of wastewaters containing high radionuclide concentrations. Radium can adsorb onto scales in pipes and tanks and can also be removed from water by coprecipitation if sulfate or carbonate is added to hydraulic fracturing wastewater to precipitate calcium, barium, or strontium (Kappel et al., 2013; USGS, 2013a). Pipe scale in oil and gas production facilities has been found to have radium concentrations as high as 154,000 pCi/g, although concentrations of less than about 13,500 pCi/g are more common (Schubert et al., 2014). A similar issue, the potential for accumulation and possible release of radionuclides and other trace inorganic constituents in water distribution systems, has gained attention, with the potential for drinking water concentrations to exceed drinking water standards (Water Research Foundation, 2010). Scale eventually removed from pipes or other oil and gas equipment can end up in landfills and then leach into groundwater or run off to surface water (USGS, 2013a). Also, laboratory research suggests that radium in land-applied barium sulfate scales from conventional oil and gas operations may become mobilized by microbial processes, rendering the radium more mobile and bioavailable (Matthews et al., 2006; Swann et al., 2004); see discussion in Section 8.4.6.1. Monitoring would be needed in order to ascertain the potential for accumulation and release of radionuclides from systems that have treated or continue to treat hydraulic fracturing wastewaters with elevated TENORM concentrations.

8.5.2.6 Road Spreading

Salt and radionuclide accumulation can occur near road spreading sites; one study in Pennsylvania describes a roughly 20% increase in average radium-226 concentrations in soils near five roads where wastewaters from conventional operations had been spread for de-icing (Skalak et al., 2014). However, the standard deviation for the samples was large (24 pCi/g), and background concentrations were approximately 1 pCi/g. Should significant accumulation of radionuclides in soils near roads occur, it would present a vehicle for potential impacts on drinking water resources. The frequency with which hydraulic fracturing wastewater contributes to this type of impact depends on state-level regulations dictating whether the wastewater can be used for road spreading.

8.5.2.7 Potential for Monitoring

Effluent from treatment plants (e.g., CWTs, POTWs) and receiving waters can be monitored for radionuclides. Research suggests that radium-226 and radium-228 are the predominant radionuclides in Marcellus Shale wastewater, and they account for most of the gross alpha and gross beta activity in the waters studied (Rowan et al., 2011). Gross alpha and gross beta measurements may, therefore, serve as an effective screening mechanism for the presence of radionuclides in hydraulic fracturing wastewater. This in turn can help in evaluating management strategies. Portable gamma spectrometers allow rapid screening of wastewater effluent. Sediments can also be measured for radionuclide concentrations at discharge points. If an accurate assessment of total radioactivity is needed rather than a screening, measuring radium content may not be adequate depending upon how wastewater has been stored. Analyses of other radionuclides such as Pb-210, Po-210, and Th-228 may be warranted, especially if the wastewater has been stored in closed loop systems.

8.5.3 Metals

Given the presence in hydraulic fracturing wastewaters of some heavy metals, as well as barium and strontium concentrations that can reach hundreds or even thousands of milligrams per liter (Table 7-5), surface waters may be impacted if discharges from CTWs or POTWs indirectly receiving oil and gas wastewater via CWTs are not managed appropriately or if spills occur.

Common treatment processes, such as chemical precipitation, are effective at removing many metals (Section 8.4). Effluent sampling results collected between October 2011 and February 2013 from seven facilities in Pennsylvania treating oil and gas wastewaters were requested by the EPA. The results revealed low to modest concentrations of copper (0 – 50 μg/L), zinc (14 – 256 μg/L), and nickel (8 – 22 μg/L) (U.S. EPA, 2015f, g). However, metals such as barium and strontium were found to range from low to elevated concentrations in the effluent for some of the facilities. The data showed effluent barium concentrations ranging from 0.35 to 25 mg/L (median of 3.5 mg/L and average of 6.7 mg/L). For results that were greater than 2 mg/L, the drinking water MCL for barium was exceeded. Strontium concentrations ranged from 0.36 to 546 mg/L (median of 297 mg/L and mean of 236 mg/L) (U.S. EPA, 2015g). (See Chapter 9 for information on health effects for barium and strontium.)

Volz et al. (2011) discussed a December 2010 effluent sampling effort at a Pennsylvania CWT that had been treating both conventional and Marcellus wastewater; they measured average barium and strontium concentrations of 27 mg/L and nearly 3,000 mg/L, respectively (eight samples from the one plant) (Volz et al., 2011). NPDES compliance data submitted for 2011 shows that effluent from the same CWT had average barium effluent levels ranging from 26 to 98 mg/L in the months prior to PA DEP's April 2011 request to cease sending hydraulic fracturing wastewater to this and other facilities exempt from the 2010 TDS regulation (U.S. EPA, 2015f, g). After May, 2011, barium effluent concentrations dropped to average values of 9 to 22 mg/L. The facility is scheduled to upgrade its TDS removal capabilities, which should help decrease concentrations of metals in the effluent.

Limited data are available on metal concentrations in wastewater and treated effluent that are directly discharged; additional information would be needed to assess whether there could be downstream effects on drinking water utilities. NPDES discharge permits, which restrict TDS discharge concentrations, would likely reduce metal effluent concentrations due to the additional treatment necessary to minimize TDS.

8.5.4 Volatile Organic Compounds

Benzene is a common constituent in hydraulic fracturing wastewater, and it is of concern due to recognized human health effects. A wide range of concentrations of BTEX compounds occurs in wastewater from the Barnett and Marcellus shales. Natural gas formations generally produce more BTEX than oil formations (Veil et al., 2004), and lower concentrations of BTEX naturally occur in wastewater from CBM production (Appendix Table E-9). The organic chemistry of Marcellus wastewater has been found by Akob et al. (2016) to be more variable than that of inorganic constituents, indicating the need to consider the concentrations of organic compounds when planning wastewater management.

Processes such as air stripping or dissolved air flotation can remove VOCs during treatment, but if treatment is not adequate prior to discharge, the VOCs may reach water resources. For example, the average benzene concentration measured in the discharge from a Pennsylvania CWT in December 2010 was 12 μg/L (Volz et al., 2011) exceeding the MCL for benzene of 5 μg/L.[1] The facility was receiving wastewater from both conventional and unconventional operations at that time. Ferrar et al. (2013) measured for BTEX in effluent from the same facility, and mean concentrations among the four compounds ranged from approximately 2 to 46 μg/L. Concentrations were lower for samples taken after May 19, 2011 (when Marcellus operators voluntarily stopped sending wastewater to POTWs and CWTs exempt from the 2010 TDS regulation), and the difference between pre and post May 2011 sampling was considered statistically significant.

Spills and leakage from pits creates another potential route of entry to drinking water resources, as described in Section 8.4.5. Akob et al. (2016) documented the microbial degradation of organic compounds in Marcellus produced water and note that more research is needed to evaluate how this could mitigate the migration of organic constituents in the event of spills or leaks.

8.5.5 Semi-Volatile Organic Compounds

Little is known about the fate of the SVOC, 2-butoxyethanol (2-BE) (an antifoaming and anti-corrosion agent used in slick-water) (Volz et al., 2011) or its potential impact on surface waters, drinking water resources, or drinking water systems. This compound is very soluble in water and is subject to biodegradation, with an estimated half-life of approximately 1-4 weeks in the environment (Wess et al., 1998). It is classified by the EPA's Integrated Risk Information System (IRIS) as not likely to be carcinogenic to humans, and the International Agency for Research on Cancer (IARC) classifies it as having insufficient evidence to determine carcinogenicity (see Chapter 9 for more information). 2-BE was detected in the discharge of a Pennsylvania CWT at

[1] Among the BTEX compounds, the MCL for benzene is the lowest at 5 μg/L; the MCL for ethylbenzene is 700 μg/L, the MCL for toluene is 1,000 μg/L, and the MCL for xylenes is 10,000 μg/L.

concentrations of 59 mg/L (Volz et al., 2011). Ferrar et al. (2013) detected 2-BE in the effluent from a CWT in western Pennsylvania at average concentrations of 34 – 45 mg/L; the latter value was measured when the CWT was receiving only conventional oil and gas wastewater. Data are lacking on 2-BE concentrations in surface waters that receive treated effluent from hydraulic fracturing wastewater treatment systems.

Polyaromatic hydrocarbons (PAHs; a group of SVOCs) have been found in hydraulic fracturing wastewater (Section 7.3.4.7, Table 7-6). PAHs detected in an unlined pit containing oil and gas wastewater near the Duncan Oil Field in New Mexico were also detected in soils 82 ft (25 m) downgradient at concentrations ranging from 2,000 to 4,900 μg/kg and 164 ft (50 m) downgradient, with concentrations ranging from 22 to 370 μg/kg (Sumi, 2004; Eiceman, 1986).

8.5.6 Oil and Grease

Oil and grease in oil and gas wastewater can come from the formation or from oil-based drilling fluids. Typically, oil and grease are separated from the wastewater before discharge either by heat treatment or by gravity separation followed by skimming. If these processes are inefficient, oil and grease can be integrated with the discharge to surface waters. For example, in some cases, oil and grease are allowed to separate in pits, and water is then withdrawn from the lower part of the pit. If the oil layer is allowed to drop to the level of the standpipe or if the water is agitated, oil and grease may be discharged along with the water. Oil and grease are also often dispersed in wastewater in the form of small droplets that are 4 to 6 microns in diameter. These droplets can be difficult to remove using typical oil/water separators (Veil et al., 2004).

A study was conducted in Wyoming by the U.S. Fish and Wildlife Service from 1996 to 1999 of sixty five oil and gas sites that discharge to ephemeral streams and subsequently to wetlands. Fifteen percent of the wetlands receiving wastewater contained oil-stained vegetation and had a visible oil sheen on the sediments. In addition, ten of twelve sites that were randomly selected for water sample collection (from oil field separator or skim pit effluent) exceeded the discharge limit of 10 mg/L for oil and grease with one site as high as 54 mg/L (Ramirez, 2002).

8.6 Synthesis

A variety of strategies may be considered for the management of hydraulic fracturing wastewater. Important factors for planning management include cost, logistics, wastewater composition, wastewater volumes, and regulations. Available information suggests that Class IID wells regulated under the UIC Program are the most frequently used wastewater management practice, but reuse, sending to a CWT, and various other methods are also employed.

8.6.1 Summary of Findings

8.6.1.1 Wastewater Volumes

The most current national estimate of the total wastewater volume generated in the oil and gas industry (both onshore and offshore) was 889.59 billion gal (21.18 billion bbls or 3.37 trillion L) in 2012, although this estimate is subject to a number of uncertainties (Veil, 2015). The total amount of wastewater generated may increase if hydrocarbon production increases in a region, although Veil (2015) suggests that this trend may not hold true at the national level. Geographically, a large portion of onshore oil and gas wastewater in the United States is reported to be generated in the western part of the country, consistent with the areas where most oil and gas wells are located and most production takes place.

Obtaining reliable national estimates of the amount of wastewater attributable to hydraulic fracturing is a challenge. State data collection efforts vary, and in many states, production data do not identify which wells have been hydraulically fractured. However, annual estimates compiled from those states where hydraulic fracturing wastewater is identified range from hundreds of millions to billions of gallons of wastewater generated each year. Data from individual states indicate that along with an increase in the numbers of hydraulically fractured wells, associated wastewater volumes have generally increased over the last several years into 2014. However, while there is a general correlation between unconventional oil and gas production and wastewater volume, the relationship is complicated by several factors such as timing of drilling and production. More complete and comparable estimates of local, state, and regional wastewater volumes would facilitate wastewater management on the part of operators as well as planning on the part of agencies that oversee wastewater management.

8.6.1.2 Wastewater Management Practices

Hydraulic fracturing wastewater is managed in a variety of ways, including disposal via Class IID wells; minimal treatment and reuse (in subsequent fracturing operations); more complete treatment followed by reuse; sending to CWTs for treatment followed by direct discharge or transfer to POTWs; evaporation; and other uses such as agriculture and wildlife use (allowed only in the arid west when the wastewater is of good enough quality for such uses). All of these strategies have the potential to affect drinking water resources. Wastewater management practices continue to shift in response to evolving understanding of environmental concerns, emplacement of new regulatory controls, changes in costs, and changes in technology and operator practices. Unauthorized discharges of hydraulic fracturing wastewater have also been documented, and such discharges can potentially impact drinking water resources.

As of 2015, available information suggests that Class IID disposal wells are a primary wastewater management practice for operators in most of the major unconventional reservoirs in the United States, with the notable exception of the Marcellus Shale region in Pennsylvania. Class IID wells tend to be economically favorable, especially if they are located within a reasonable transportation distance from well sites (U.S. GAO, 2012). In particular, large numbers of active injection wells are found in Texas (7,876), Kansas (5,516), Oklahoma (3,837), Louisiana (2,448), and Illinois (1,054) (U.S. EPA, 2016d).

Pennsylvania is somewhat unique in having only nine Class IID wells (as of February 2015), along with having experienced significant growth of shale gas production in the Marcellus and corresponding production of large volumes of wastewater. Operators producing from unconventional formations (as defined by PA DEP) have managed their wastewater through the use of POTWs (a practice that is subject to recently promulgated regulations), CWTs, extensive reuse for hydraulic fracturing operations, and hauling to disposal wells (to a lesser degree). The wastewater management history in Pennsylvania provides an example of evolving strategies to manage the treatment, discharge, storage, and reuse of hydraulic fracturing wastewaters that are high in constituents of concern (e.g., bromide, TDS, and TENORM).

Reuse of hydraulic fracturing wastewater to formulate fluid for subsequent hydraulic fracturing jobs is most prevalent in Pennsylvania (as high as 90%), with much of the reuse happening on-site (PA DEP, 2015b). Reuse is practiced in other regions as well (e.g., Haynesville Shale, the Fayetteville Shale, the Barnett Shale, and the Eagle Ford Shale), but at much lower rates (about 5 – 20%). Reliable estimates are not available for all areas of the United States because waste management practices are not consistently reported across all states. If hydraulic fracturing activity slows, demand for wastewater for reuse will also likely decrease, and other forms of wastewater management will be needed. Potential impacts associated with reuse center on concerns over the storage of untreated or minimally treated wastewater on-site or transport to CWTs.

Treatment of hydraulic fracturing wastewater may be done at CWTs or using mobile or semi-mobile systems designed for on-site use. Treatment at a CWT may be followed by direct discharge by the CWT to surface water, indirect discharge to a POTW in accordance with recently promulgated regulations, or reuse. Most CWTs treating hydraulic fracturing wastewater are located in Pennsylvania (about 40 facilities), with a limited number in other states. CWTs vary widely in treatment capabilities, ranging from producing high-quality effluent to minimal treatment for reuse.

Other wastewater management practices, such as evaporation and agricultural uses, represent a smaller fraction of wastewater management nationally. These practices can, however, be locally significant. Although specific instances of contamination were not identified for this assessment, these practices could lead to impacts on drinking water resources if facilities are not properly constructed and maintained or if water quality is not adequately characterized to ensure that management is appropriate.

8.6.1.3 Treatment and Discharge

Wastewater that is treated and subsequently discharged by CWTs can result in impacts due to inadequate treatment. A frequently cited concern is the high TDS content in wastewaters from unconventional formations, which poses challenges for treatment, discharge, and reuse. Treatment processes such as sedimentation, filtration, flotation, and chemical precipitation are capable of removing constituents such as oil and grease, major cations, metals, and TSS. They do not, however, adequately reduce TDS in high-salinity wastewaters. More advanced processes such as reverse osmosis (RO) or distillation are needed if TDS removal is required (Shaffer et al., 2013; Younos and Tulou, 2005). Most available information on treatment of hydraulic fracturing wastewater is based on practices used in Pennsylvania because that is where most data have been collected.

Hydraulic fracturing wastewater discharged from treatment facilities without advanced TDS removal processes has resulted in elevated TDS concentrations (including bromide, iodide, and chloride levels) in receiving waters. Impacts from these discharges is due largely to the role of bromide and iodide in DBP formation at downstream drinking water systems, potentially causing higher levels of harmful DBPs in finished drinking water.[1] Modeling suggests that very small percentages of hydraulic fracturing wastewater added to a river used as a source for drinking water systems could cause a notable increase in DBP formation.

Radionuclides (i.e., TENORM), which are present in some hydraulic fracturing wastewaters, can cause impacts if the wastewater is discharged without adequate treatment. TENORMs have been measured in effluent from wastewater treatment facilities receiving Marcellus wastewater (which includes effluent sent for reuse and not discharged to surface water). Radium-226, radium-228, gross alpha, and gross beta are most cited as the radioactive constituents of concern, likely due to the availability of test methods for these constituents in wastewater. Radium concentrations can range up to thousands or tens of thousands of pCi/L. Fewer data are available on concentrations of uranium and other radionuclides in hydraulic fracturing wastewaters. Also, fewer data exist on radionuclide concentrations in wastewaters from unconventional formations other than the Marcellus, limiting our ability to assess potential impacts from TENORM on a nationwide basis.

Other constituents posing health or environmental concerns that can be discharged in inadequately treated hydraulic fracturing wastewater include organic compounds, barium, strontium, and other metals. Chemicals used in the fracturing fluid or their degradation products could also be present. A variety of treatment processes can be used for removal of these contaminants, from commonly used methods such as chemical precipitation and filtration to more advanced and more costly techniques, such as reverse osmosis, distillation, and mechanical vapor recompression.

8.6.1.4 Storage and Disposal Pits and Impoundments

Regardless of the wastewater management practices used, some type of temporary storage of fluids is generally required. Storage can be in the form of tanks as well as pits and/or impoundments. Pits encompass a variety of structures, from on-site pits for storage at the well site to larger, centralized facilities (typically referred to as "impoundments" or "ponds"). Some states allow evaporation pit facilities or percolation pits as a means of wastewater disposal. The locations and number of pits are not well documented in most states, nor are pits associated with hydraulic fracturing operations necessarily identified, despite efforts by the U.S. EPA (U.S. EPA, 2003b) and environmental groups such as SkyTruth to identify pits in use. Information that is typically available on state websites includes permitted centralized commercial evaporation facilities (COWDFs) most commonly used in the western United States.

Impacts on both groundwater and surface water resources due to inadequate pit capacities, overfilling, and leaks have been documented. In extreme precipitation events, pits can be overtopped. Leaks can occur if liners are compromised or were not used. With an increased emphasis on reuse in some regions, the need for temporary storage of high-TDS wastewater increases the potential for leaks and spills from pits and during fluid handling.

[1] Some types of DBPs are regulated under SDWA's Stage 1 and Stage 2 DBP Rules, but a subset of DBPs, including a number of chlorinated, brominated, nitrogenous, and iodinated DBPs, are not regulated. Brominated and iodinated DBPs are more toxic than other species of DBPs.

Unlined pits, in particular, provide a pathway for contaminants to reach groundwater, and impacts on groundwater from historic and current uses of unlined pits in the oil and gas industry have been documented. The resulting contamination can be long-lasting. States have taken measures to phase out the use of unlined disposal and storage pits, but unlined pits that are still in use can provide an ongoing potential source of contamination for groundwater (Grinberg, 2014).

8.6.1.5 Residuals

Solid and liquid residuals associated with hydraulic fracturing wastewater (treatment residuals from CWTs, sludges from tanks and pits, and pipe scale) could have impacts on drinking water resources if not managed and disposed of properly. Liquid residuals are inappropriate for surface water discharge or discharge to a POTW due to high concentrations of salts and other contaminants; they are commonly disposed of in an injection well. Solid residuals may leach a number of constituents, such as alkali metals, alkaline earth metals, and bromide. They can also contain TENORM if radionuclides are present in the wastewater being treated. Given that residuals are commonly disposed of in landfills, TENORM can be problematic due to the possibility of radon emissions and radioactivity in the landfill leachate. Solids from pits or tanks can also contain TENORM if the wastewater contains radionuclides, and one study has shown the potential for radioactivity to increase in the closed environment of tanks.

8.6.2 Factors Affecting the Frequency or Severity of Impacts

The frequency and severity of impacts on drinking water resources from hydraulic fracturing wastewater will depend upon the wastewater composition and volumes, and the mix of wastewater management strategies used.[1] The types of potential impacts (along with frequency and severity) may shift in time as management practices change in response to evolving environmental, regulatory, economic, or logistical drivers. The frequency and severity of impacts can also depend on the size and initial quality of the drinking water resource and its proximity to wastewater management operations.

8.6.2.1 Role of Changing Wastewater Management Practices

The most common disposal option for hydraulic fracturing wastewater is injection into Class II disposal wells. If this option becomes restricted in a given location, the wastewater management options could shift, at least locally, towards other options such as sending wastewater to CWTs for treatment and either discharge or reuse. Although reuse avoids the immediate need to discharge wastewater by directing it to ongoing hydraulic fracturing activities, the practice could concentrate radionuclides or other constituents as fluid moves through cycles of reuse. Whether such concentrations would be significant depends on the ratio of recycled to "fresh" water when the wastewater gets reused. Alternatively, wastewater might need to be transported to more distant

[1] Both national and state regulations affect the wastewater management practices used. At a national level, although the EPA's oil and gas ELG regulations generally prohibit the direct discharge of oil and gas wastewater to waters of the U.S., treatment and discharge of hydraulic fracturing wastewater can occur under certain limited circumstances, such as under an exemption authorizing discharge for agricultural and wildlife use in the arid west, or by Centralized Waste Treatment facilities. For additional information on national regulations relevant to hydraulic fracturing wastewater management, see Text Box 8-2.

Class IID wells. This option, while attractive from the perspective of limited disposal impacts, could increase the frequency of impacts from spills and leaks during transportation (see Chapter 7 for discussion of roadway transport of produced water).

8.6.2.2 Treatment and Discharge

Both the frequency and severity of potential impacts on drinking water resources from treated hydraulic fracturing wastewater depend on the influent concentrations of the constituents in the wastewater and the type and adequacy of the treatment processes employed. If treatment and/or blending is inadequate, the resulting quality in a receiving water could, for example, influence formation of DBPs during subsequent drinking water treatment, impair biological treatment processes, and release TENORMs into receiving waters.

The volume of treated effluent discharged relative to the size of the receiving water body is an important local factor affecting the frequency and severity of potential impacts. Because of dilution effects, drinking water systems drawing from smaller rivers will likely face greater challenges in dealing with contaminants in their source water than systems drawing from larger rivers receiving the same volume of effluent. Seasonal changes in streamflow will also affect frequency and severity by affecting the degree of dilution. Existing loadings of pollutants from other sources in a watershed can increase the frequency and severity of potential impacts if the additional contributions from hydraulic fracturing wastewater cause concentrations to exceed thresholds.

Direct discharges of wastewaters with lower TDS concentrations to ephemeral streams are allowed in parts of the country where the wastewater is considered to be "of good enough quality" for livestock watering and wildlife use, and the discharges may constitute a large portion of streamflow. Permits authorizing such discharges may only require monitoring for a limited set of constituents. In particular, they may not necessarily require monitoring for specific constituents associated with hydraulic fracturing. The potential for water quality impacts from such discharges depends upon whether chemicals used for fracturing fluid or maintenance (or their degradation products) are present and at what concentrations. Long-term discharges to these ephemeral streams could result in ongoing impacts if there are unrecognized or unaddressed water quality issues.

Concerns about radionuclides in hydraulic fracturing wastewater have received considerable public attention, especially in the Marcellus region. The severity and frequency of impacts on receiving waters and sediments from TENORM depends upon the TENORM content in the wastewater (highest in regions with NORM-rich formations), temporal variability in the wastewater composition, and the treatment processes used. There are insufficient data to indicate whether radionuclides from these wastewaters have reached drinking water intakes. However, data do suggest that radionuclides can accumulate in sediments at or near discharge points from facilities that treat and discharge oil and gas wastewater. A recent PA DEP study (PA DEP, 2015b) reported radium in the effluent of both CWTs handling oil and gas wastewater and POTWs receiving effluent from such facilities.

Analysis of TENORM concentrations in hydraulic fracturing wastewaters prior to treatment, selection of appropriate treatment processes that adequately address the TENORM levels, and monitoring of TENORM in the treated effluent and receiving waters could help address the frequency and severity of potential impacts on drinking water resources in these areas. However, a confounding issue is underestimation of radium concentrations when using traditional wet chemical methods with high-TDS waters. This could consequently cause underestimation of frequency or severity of impacts. Newer studies have begun to use gamma spectroscopy for better recovery, which could help with more accurate assessment of frequency and severity of impacts (Nelson et al., 2014).

Accumulation of other contaminants such as organic compounds or metals in sediments at or near discharge points is also possible. If the sediments are disturbed or entrained due to dredging or flood events, contaminated sediments could be transported downstream closer to drinking water systems. The fate of such sediments and likelihood of mobilization of contaminants originating from hydraulic fracturing wastewaters have not been explored. The frequency and severity of impacts related to contaminated sediments would depend on a number of site-specific factors such as concentrations in the sediments, effluent quality, volume from the discharging facility, stream water quality, and stream hydrodynamics.

8.6.2.3 Storage and Disposal Pits and Impoundments

Tanks, pits, and impoundments, ever-present at oil and gas operations and CWTs, provide an opportunity for impacts on drinking water resources. Tanks are generally regarded as being safer than pits in terms of containment, although recent research has shown the potential for an increase in radioactivity in tank sediment if the wastewater contains TENORM. For pits and impoundments, the likelihood and severity of impacts due to spills and leaks depends in part on state construction and maintenance requirements for pits and how well these are observed. Frequency and severity of impacts will be lessened by attention to design standards, competent construction, and operational practices.

Liners, in particular, are an important measure to protect groundwater resources and are a common aspect of pit construction requirements. Liner specifications address materials, thickness, and leak detection. If a liner is compromised or nonexistent, the severity of impacts on groundwater will be affected by the volume leaked, the composition of the water in the pit, the depth to the water table, soil permeability, and the capacity of the soil to retain certain pollutants as the water percolates through. If substantial sediment has built up in the bottom of the pit, then in the event of a liner breach, contaminants may leach if the sediments permit water to pass through and into the soil. The fate and transport of wastewater contaminants in the subsurface is governed by a complex set of physical, chemical, and biological processes that dictate interactions between wastewater constituents and soil minerals, degradation or transformation of wastewater constituents, and possible mobilization of constituents in the soil under a pit (see Section 5.8 in Chapter 5 for a thorough discussion of processes affecting movement of constituents in the subsurface). Duration of use is also a consideration; the longer a pit with a faulty or nonexistent liner receives wastewater, the more severe the ultimate impact could be on underlying sediment and groundwater.

In the event of overtopping of a pit due to overfilling or extreme weather, the severity of impacts on surface water or groundwater will depend on the volume that overflows, wastewater composition, distance to surface water (if wastewater flows over land), depth to the water table, and soil properties (if the overflow infiltrates into the soil). If the overflow reaches a stream or river, the size of the spill relative to stream size and flow rate could also affect the severity of the impact. The combined factors that can contribute to overflows include capacity of the pit, the volume of fluid stored in the pit (i.e., freeboard) at the start of the precipitation event, and failure to monitor/ reduce pit fluid levels if needed.

As with concerns over discharges, the potential for impacts will be tied to other, existing stresses within a watershed. If the surface water is already receiving pollutant loadings from other sources, then an additional contribution from a pit-related leak or spill may not be as readily accommodated without causing water quality impairment.

8.6.2.4 Other Management Practices and Management of Residuals

Other management strategies such as irrigation, road spreading, and evaporation are less frequently employed for hydraulic fracturing wastewaters. The severity of impacts on surface waters from irrigation and road spreading will depend on the constituents in the wastewater (e.g., salts, radionuclides, and chemicals used in hydraulic fracturing), the distance to a receiving water, and whether stormwater management measures exist to mitigate runoff. The factors influencing whether constituents will migrate to shallow groundwater include depth to the water table, precipitation, soil permeability, and the soil's ability to retain pollutants that can adsorb to particles. If irrigation and road spreading are long-term management practices, the frequency of impacts will likely be proportional to the frequency with which the practices are employed.

Liquid and solid residuals generated from the treatment, storage, and handling of hydraulic fracturing wastewater have highly concentrated waste constituents. This could increase the potential severity of impacts due to spills that reach surface water resources or leach to groundwater. Potential impacts from management of residuals can be lessened in frequency and severity through careful handling, adequate characterization (including TENORM content), and selecting an appropriate disposal method, including use of a landfill that can accept TENORM waste if needed.

8.6.3 Uncertainties

A full understanding of hydraulic fracturing wastewater management is limited by a lack of available data in several areas. First, it is difficult to assemble a complete national- or regional-level picture of wastewater volumes and the management practices used because the tracking and availability of data vary from state to state. Although some states provide well-organized and relatively thorough data, not all states collect or make such information available. It can be difficult to identify wastewater volumes specifically associated with hydraulic fracturing (as compared to all oil and gas production activities). Such data would be needed to place hydraulic fracturing wastewater in the broader context of all oil and gas wastewaters. It is also generally difficult to determine whether hydraulic fracturing wastewater is being injected under a given disposal well permit because the permit rarely identifies which production wells are contributing to the wastewater stream. Data are also generally difficult to locate for wastewater production volumes, the chemical composition and concentrations in wastewater, and the management and disposal strategies for residuals.

Up-to-date information on the volume of hydraulic fracturing wastewater disposed of via underground injection by state is not uniformly available. Without this information, it is difficult to assess whether disposal well capacity will become an issue in areas where hydraulic fracturing activity is expected to increase or where use of disposal wells may become restricted locally or regionally.

For CWTs permitted to discharge to surface water, the ability to assess the potential effects of these discharges on drinking water resources is limited by the lack of effluent water quality data. Some monitoring data are required by the permit, but the list of monitored constituents may be limited. Selection of the appropriate water quality parameters to be monitored is critical to ensure that potentially problematic constituents are identified (e.g., chemicals associated with hydraulic fracturing fluids, maintenance chemicals, and degradation products of those chemicals). Some chemicals used in fracturing fluids are not disclosed, and analytical methods are lacking for some chemicals of concern and their degradation products.

Pollutant removal capabilities of the treatment facilities would also be valuable information to have, but this would require well-coordinated collection of both influent and effluent samples; this type of data is even less commonly available. In addition, the use of inappropriate analytical methods for the high TDS wastewater associated with hydraulic fracturing operations can complicate the use of available data. Methods used should be suitable for the highly complex matrix of contaminants encountered with oil and gas wastewater to have confidence in the results of chemical analyses.

Monitoring of surface waters downgradient of discharges, such as screening with a TDS proxy (i.e., conductivity), would also help assess the frequency of impacts on receiving waters by hydraulic fracturing activities (including spills and discharges of wastewater). Such data can also give an estimation of the severity of those impacts. Other than a few studies in the Marcellus Shale region, these types of water quality data are lacking. Existing data are also limited regarding legacy effects, such as accumulation of contaminants in sediments at discharge points, soil accumulation due to application of de-icing brines or salts from wastewater treatment, and handling of wastewater treatment residuals.

Assessing longer-term impacts on surface water quality from hydraulic fracturing activities in general is severely hampered by inadequate data. Bowen et al. (2015) state that available national-level databases are inadequate for addressing the question of whether there is evidence of national-level trends in surface water quality (as measured by specific conductivity and chloride) in areas where unconventional oil and gas production is taking place. Work by the Northeast-Midwest Institute and the USGS (Betanzo et al., 2016) was undertaken to explore the types and amounts of data needed to assess whether shale gas development activities contaminate surface water or groundwater in the Susquehanna River Basin. The focus was on longer-term cumulative impacts because detection of such impacts requires water quality monitoring. Detection of impacts (in either surface water or groundwater) requires a systematic monitoring approach that includes sampling at appropriately selected locations at an adequate frequency and duration and for a suite of water quality parameters to detect changes over time. Comparison sites without hydraulic fracturing activity are needed as well. The authors concluded that the data necessary to detect changes in surface water or groundwater due to hydraulic fracturing activities do not currently exist for the Susquehanna River Basin.

8.6.4 Conclusions

Oil and gas operations in the United States generated an estimated 2.43 billion gal of wastewater per day (about 60 million bbls/day) in 2012 (Veil, 2015). This includes wastewater associated with hydraulic fracturing activities, although what portion of this oil and gas wastewater is attributable to hydraulic fracturing operations is difficult to estimate. Available information indicates that the majority of hydraulic fracturing wastewater is injected into Class IID wells regulated under the UIC Program. In the Marcellus Shale region in Pennsylvania, this option is limited, and the majority of wastewater is reused (either with or without treatment) for new hydraulic fracturing jobs. Hydraulic fracturing wastewater may also be treated at a CWT and discharged by the CWT to surface water or to a POTW. In the western United States, wastewater is used in other ways (e.g., livestock watering) if water quality allows. Wastewater is also sent to evaporation ponds for disposal or stored on-site or in centralized pits or impoundments prior to final disposal or reuse.

Impacts on drinking water resources have resulted from discharges of inadequately treated wastewater and from leaks, spills, and percolation associated with pits. Other mechanisms for impacts include improper handling of treatment residuals or pit and tank sludges as well as leaching and runoff associated with other wastewater management practices. The impacts related to pits and residuals/sludges affect both surface water and groundwater; unlined pits or those with compromised liners present a particular concern (see Chapter 7 for additional discussion of spills). The constituents that have received the greatest attention in the literature include TDS, DBP precursors (especially bromide), and radium, although hydraulic fracturing wastewater can contain elevated concentrations of a number of organic and inorganic constituents of concern. Regardless of the management option utilized, if the wastewater is not thoroughly characterized or sampling is not conducted for analytes of concern, the severity and frequency of the impacts will be unknown or unquantified. The nature and volume of wastewater generated through hydraulic fracturing activities necessitate careful consideration of handling, treatment, and ultimate reuse or disposal to ensure that water resources are not adversely impacted. There is also a need for reliable and consistent waste generation data collection and reporting, improved efforts to characterize wastewater quality (both treated and untreated), and systematic monitoring efforts to be able to detect impacts on drinking water resources.

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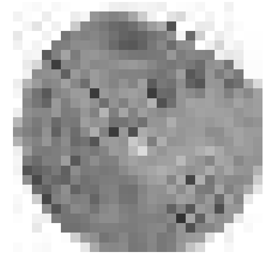

Chapter 9. Identification and Hazard Evaluation of Chemicals across the Hydraulic Fracturing Water Cycle

Abstract

This chapter identifies chemicals associated with hydraulic fracturing and provides an overview of the potential human health effects associated with these chemicals, as well as variables that could affect chemical occurrence in drinking water. The EPA has identified 1,606 chemicals associated with hydraulic fracturing, including 1,084 chemicals that are used in hydraulic fracturing fluid and 599 chemicals that have been detected in produced water. There is some uncertainty surrounding this chemical list, as it does not include a subset of chemicals that are classified as confidential business information, and because understanding of produced water composition is constrained by limitations of analytical chemistry as well as site-specific variations in the geochemistry of hydraulically fractured rock formations.

The EPA used selected federal, state, and international sources of toxicological data to identify toxicity values that can be used to support risk assessment for these chemicals, including chronic oral reference values (RfVs) for noncancer effects and oral slope factors (OSFs) for cancer. Chronic oral RfVs or OSFs were available for 173 (11%) of the total 1,606 chemicals. Health effects associated with chronic oral exposure to these chemicals include carcinogenicity, neurotoxicity, immune system effects, changes in body weight, changes in blood chemistry, liver and kidney toxicity, and reproductive and developmental toxicity.

For the majority of chemicals that lack chronic oral RfVs or OSFs, risk assessors will have to turn towards other sources of toxicological information that may have greater uncertainty than RfVs and OSFs, including quantitative structure-activity relationship (QSAR) models or additional data from the EPA's Aggregated Computational Toxicology Resource (ACToR) database. To understand whether specific chemicals can affect human health through their presence in drinking water, data on chemical concentrations in drinking water are needed. In the absence of these data, a preliminary analysis of relative hazard potential for drinking water resources can be conducted using the multi-criteria decision analysis (MCDA) approach outlined in this chapter. The MCDA combines data on toxicity, occurrence, and physicochemical properties for selected subsets of chemicals and was used in this chapter to highlight several chemicals that may be more likely than others to reach drinking water resources and present a health hazard.

Overall, while evidence suggests that hydraulic fracturing has the potential to impact human health, the actual human health implications are not well understood or well documented. Given that chemicals in hydraulic fracturing fluids and produced water are likely to vary on a regional basis and even between individual wells, the materials presented in this chapter are best applied for risk assessment and risk management decision-making at the local level.

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9. Identification and Hazard Evaluation of Chemicals across the Hydraulic Fracturing Water Cycle

9.1 Introduction

In this chapter, we present and integrate what is known about chemicals in the hydraulic fracturing water cycle, and provide an initial assessment of the potential for these chemicals to impact human health. The discussion is focused on the availability of toxicity values and qualitative assessments that can be used to inform the risk assessment of these chemicals for oral exposure via drinking water—in particular, the available noncancer oral reference values (RfVs), cancer oral slope factors (OSFs), and qualitative cancer classifications.[1,2,3] Public health impacts will depend upon both the inherent toxicity of these chemicals and the potential for human exposure. We highlight several field studies that have detected hydraulic fracturing-related chemicals in drinking water resources, and discuss properties of chemicals related to environmental fate and transport that could affect their potential impact on drinking water resources. To the extent information was available to do so, knowledge of toxicological and chemical properties was combined to illustrate a preliminary analysis of the relative hazard that these chemicals could pose to drinking water resources. The data are presented in this chapter as follows:

Section 9.2 provides a brief background on public health concerns surrounding hydraulic fracturing and unconventional oil and gas extraction, which have been highlighted in several recent studies.

Section 9.3 discusses how information sources were used to create a list of chemicals used in or detected in various stages of the hydraulic fracturing water cycle. The consolidated chemical list includes chemicals reportedly added to hydraulic fracturing fluids in the chemical mixing stage, as well as fracturing fluid chemicals, formation chemicals, or their reaction products that may be carried in produced water.

Section 9.4 provides an overview of the methods that were used for gathering information on toxicity and physicochemical properties for all chemicals identified in Section 9.3, and outlines the number of chemicals that had available data on these properties. For toxicological data, the primary focus is on chronic oral RfVs, OSFs, and qualitative cancer classifications from selected data sources that met the EPA's criteria for inclusion in this assessment. This section also discusses other

[1] A reference value (RfV) is an estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfV is a generic term not specific to a given route of exposure (U.S. EPA, 2011f). In the context of this report, the term RfV refers to reference values for non-cancer effects occurring via the oral route of exposure and for chronic durations, except where noted.

[2] An oral slope factor (OSF) is an upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100 (U.S. EPA, 2011f).

[3] Qualitative cancer classifications are a system used for the hazard identification of probable carcinogens, in which human data, animal data, and other supporting evidence are combined to characterize the weight of evidence (WOE) regarding the potential of an agent to cause cancer in humans.

potential sources of toxicity information: the use of quantitative structure-activity relationship (QSAR) modeling to estimate chemical toxicity, as well as the availability of toxicological information on the EPA's Aggregated Computational Toxicology Resource (ACToR) database. A brief description of other potential tools and approaches that may be used by stakeholders for site-specific evaluation of chemical hazards, but are not used in this report, is provided in Appendix G.

Section 9.5 describes the potential hazards of subsets of chemicals identified as being of interest in previous chapters of this report. This includes chemicals in hydraulic fracturing fluid (Chapter 5); organic chemicals, inorganic chemicals, and pesticides detected in produced water (Chapter 7); stray gas, such as methane (Chapter 6); and disinfection byproducts (DBPs) formed from constituents of hydraulic fracturing fluid wastewaters (Chapter 8). We discuss instances in which these chemicals have been detected in drinking water resources in areas of hydraulic fracturing activity, and provide an overview of the available toxicological information for these chemicals.

Section 9.6 uses a multi-criteria decision analysis (MCDA) framework to provide a preliminary analysis of the potential hazards of chemicals used in hydraulic fracturing fluids or detected in produced water. The MCDA framework is used to integrate data on chemical toxicity, occurrence, and physicochemical properties. In this context, occurrence and physicochemical properties are used as metrics to estimate the likelihood that a chemical will reach and impact drinking water resources. Chemicals considered in these hazard evaluations include a subset of chemicals identified in the EPA FracFocus 1.0 project database, as well as a subset of organic chemicals that have been detected in produced water.

This chapter is not a human health risk assessment. As shown in Text Box 9-1, risk assessment consists of four basic steps: hazard identification, dose-response assessment, exposure assessment, and risk characterization. This chapter provides an overview of hazard identification and dose-response assessment for these chemicals, but lacks information to fully characterize exposure and risk. In Section 9.5, we highlight instances in which these chemicals have been detected in drinking water resources, but these data are only available for a small number of chemicals. The MCDA approach in Section 9.6 provides a method for integrating data on toxicity and exposure potential, but should be considered only as a preliminary analysis, and should not be used in place of local data on chemical exposure.

This chapter is focused on potential human health hazards of chemicals for the oral route of exposure (drinking water); therefore, the toxicological properties and physicochemical ranking metrics described herein do not necessarily apply to other routes of exposure that may occur with these chemicals, such as inhalation or dermal exposure. We additionally note that this analysis is focused on individual chemicals, rather than mixtures of chemicals used as additives.

In general, characterizing chemicals and their properties on a national scale is challenging and the use and occurrence of chemicals is likely to differ between geological basins and possibly on a well-to-well basis (Chapters 5 and 7). Therefore, for the protection of human health at the local level, chemical hazard evaluations are best conducted on a regional or site-specific scale. This level of analysis is outside the scope of this report; however, the methods of hazard evaluation presented here can also be applied on a regional or site-specific scale in order to identify chemicals that may present a potential human health hazard.

Text Box 9-1. Applying Toxicological Data for Human Health Risk Assessment.

Understanding potential human health impacts requires knowledge not only of the inherent toxicity of the chemicals found in contaminated environmental media, but also of the potential for exposure to these chemicals. The process of evaluating the nature and probability of such impacts is known as human health risk assessment. Overall, human health risk assessment includes four basic steps (U.S. EPA, 2016a):

1. Hazard identification: Examining whether a chemical has the potential to cause harm to humans and/or ecological systems, and if so, under what circumstances.

2. Dose-response assessment: Examining the numerical relationship between exposure and effects.

3. Exposure assessment: Examining what is known about the frequency, timing, and levels of contact with a chemical.

4. Risk characterization: Examining how well the data support conclusions about the nature and extent of risk from exposure to a chemical. Information from the hazard identification, dose-response assessment, and exposure assessment are summarized and integrated into quantitative and qualitative expressions of risk.

The RfVs and OSFs compiled by the EPA in this study pertain to the first two steps of human health risk assessment: identifying chemicals that have the potential to affect human health (hazard identification), and characterizing the exposure levels at which those effects occur (dose-response assessment). These toxicity values may be used in combination with site-specific chemical exposure information (exposure assessment) in order to evaluate potential human health risks (risk characterization). Qualitative cancer classifications characterize the weight of evidence regarding the potential for a chemical to cause cancer, and therefore provide additional information that can be used for hazard identification.

Toxicity information spans a wide range with respect to extent, quality and reliability. The RfVs, OSFs, and qualitative cancer classifications compiled in this study are those identified by the EPA as being of the highest quality and reliability, per the criteria discussed in this chapter. The QSAR-based toxicity estimates discussed in this chapter are considered to be lower on the continuum of quality and reliability, but may provide useful information pertaining to hazard identification and dose-response assessment when a chemical does not have an RfV or OSF available. The EPA's ACToR database provides an aggregation of a wide range of toxicological data that may also be useful for supporting the risk assessment of these chemicals. This chapter provides information on whether a chemical has data available from ACToR; however, it is beyond the scope of this report to evaluate the quality and reliability of data for these chemicals within ACToR, or to provide guidance on how the data within ACToR should be used to support human health risk assessment.

9.2 Overview: Hydraulic Fracturing and Potential Impacts on Human Health

As discussed in the previous chapters of this assessment, a variety of chemicals are associated with the hydraulic fracturing water cycle. Chemicals are added to hydraulic fracturing fluids at the chemical mixing stage (Chapter 5), and then injected into the well (Chapter 6). These chemical additives may return to the surface in produced water, along with chemicals from the formation (Chapter 7). The chemicals in produced water may persist in wastewater effluents, with some constituents contributing to the formation of disinfection byproducts in treated wastewater (Chapter 8). Through events such as large volume spills (Figure 9-1), mechanical integrity failures, hydraulic fracturing directly into groundwater resources, or discharge of inadequately treated hydraulic fracturing wastewater, there are specific instances in which these chemicals have been demonstrated to enter drinking water resources. Thus, there is potential for human exposure to these chemicals, and the potential for adverse human health effects resulting from exposure.

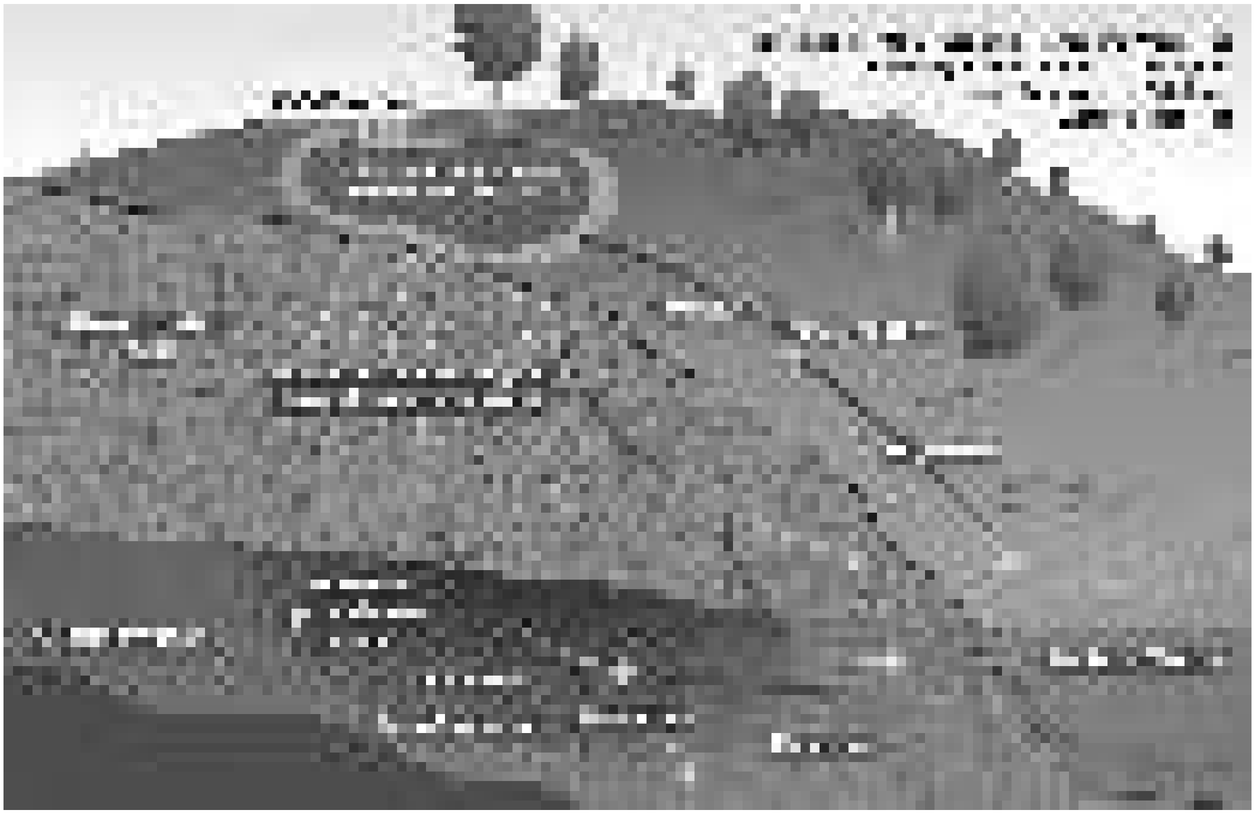

Figure 9-1. Fate and transport schematic for a hydraulic fracturing-related spill or release.

Multiple authors have noted with the recent increase in hydraulic fracturing operations there may be an increasing potential for significant public health and environmental impacts (Goldstein et al., 2014; Finkel et al., 2013; Korfmacher et al., 2013; Weinhold, 2012). These concerns have been highlighted in several recent studies. An epidemiological study in Colorado demonstrated residential proximity of pregnant mothers to natural gas wells is associated with an increased incidence of congenital heart defects, and, to a lesser extent, neural tube malformations (Mckenzie et al., 2014). A similar study in Pennsylvania found pregnant mothers living closer to unconventional natural gas wells were more likely to have infants that were small for gestational age, with lower birth weights compared to infants from mothers living farther from wells (Stacy et al., 2015). Residential proximity to natural gas wells in the Marcellus Shale is associated with an increase the number of self-reported health symptoms, particularly upper respiratory and dermal symptoms (Rabinowitz et al., 2015), chronic rhinosinusitis, migraine headache, and fatigue symptoms (Tustin et al., 2016). Laboratory studies have found that endocrine disrupting activity measured using in vitro bioassays may be elevated in surface and groundwater at known hydraulic fracturing spill sites (Kassotis et al., 2014) and in surface water downstream from a hydraulic fracturing wastewater injection facility (Kassotis et al., 2016). Although none of these studies demonstrate a direct effect of hydraulic fracturing activity on human health, and none of the epidemiological studies provided measures of individual or population level exposures or differentiated between drinking water contamination and other potential routes of exposure (e.g., air pollution), all are suggestive of a relationship between unconventional oil and gas development and adverse health outcomes.

Previous chapters of this report have identified cases in which contamination of drinking water resources could clearly be linked to hydraulic fracturing activity. For example, equipment failure and human error have led to spills of hydraulic fracturing fluids across the country and have affected the quality of drinking water resources (U.S. EPA, 2015m; Brantley et al., 2014; COGCC, 2014; Gradient, 2013). Other studies highlighted in previous chapters provide indirect evidence hydraulic fracturing activity has contaminated surface water or groundwater. For example, two recent studies in the Marcellus Shale detected known hydraulic fracturing-related chemicals in nearby groundwater wells, and used multiple lines of evidence to link the origin of these chemicals to hydraulic fracturing activity (Drollette et al., 2015; Llewellyn et al., 2015).

There have also been documented impacts on ecological receptors. In Knox County, Kentucky, retention pits holding hydraulic fracturing flowback fluids overflowed into Acorn Fork Creek during the development of four natural gas wells, causing a decrease in pH and increase in conductivity.[1] Organics and metals including iron and aluminum formed precipitates in the stream, and fish and aquatic invertebrates were killed or displaced in a 2.7 km length of the stream affected by the release (Papoulias and Velasco, 2013). A field report from the Pennsylvania Department of Environmental Protection (PADEP) described a leak in an overland pipe carrying a mixture of flowback and freshwater between two impoundments that impacted a 0.6 km length of a stream, in which 168 fish and 6 salamanders were killed (PA DEP, 2009b).

In some instances, chemical concentrations in drinking water resources impacted or potentially impacted by hydraulic fracturing activity exceeded their respective primary or secondary maximum contaminant level (MCL), or health advisory levels provided by the EPA's National Primary Drinking Water Regulations (NPDWRs) and Drinking Water Standards and Health Advisories (DWSHA) tables (U.S. EPA, 2012b), indicating that these chemicals are present at levels that may impact human health.[2] Examples will be discussed in Section 9.5. These studies generally did not indicate the contaminated water was used directly for human consumption, so it is not clear that people are being exposed to these chemicals at these levels. Nevertheless, these studies indicate that hydraulic fracturing activity may contribute to the entry of chemicals into drinking water resources at potentially harmful levels.

[1] "Flowback" refers to fluids containing predominantly hydraulic fracturing fluid that return from a well to the surface. Flowback is a type of produced water. See Chapter 7 for more details.

[2] Maximum contaminant level (MCL): The highest level of a contaminant that is allowed in drinking water. MCLs are set as close to the maximum contaminant level goal (MCLG) as feasible using the best available analytical and treatment technologies and taking cost into consideration. MCLs are enforceable standards. The MCLG is a non-enforceable health benchmark goal which is set at a level at which no known or anticipated adverse effect on the health of persons is expected to occur and which allows an adequate margin of safety (U.S. EPA, 2012b).

Risk assessment and risk management decisions will be informed by scientific information on the toxicity of chemicals in hydraulic fracturing fluid and wastewater. The U.S. House of Representatives' Committee on Energy and Commerce Minority Staff released a report in 2011 noting that more than 650 products (i.e., chemical mixtures) used in hydraulic fracturing contain 29 chemicals that are either known or possible human carcinogens or are currently regulated under the Safe Drinking Water Act (House of Representatives, 2011). More recently, several studies have performed a reconnaissance of toxicity and/or physicochemical property data for specific subsets of chemicals used in hydraulic fracturing fluids (Elliott et al., 2016; Wattenberg et al., 2015; Stringfellow et al., 2014; Colborn et al., 2011), and have provided discussion on the hazards inherent to these chemicals. In all cases, authors reported toxicity data was not available for many of the chemicals assessed in these studies, with some studies indicating significant data gaps. For instance, Wattenberg et al. (2015) evaluated 168 chemicals commonly used in hydraulic fracturing fluids in North Dakota, and reported that 59% did not have chronic toxicity data available, and 35% did not have acute toxicity data available. Elliott et al. (2016) performed a systematic evaluation of reproductive and developmental toxicity for 1021 chemicals used in hydraulic fracturing fluids or detected in wastewater, and found this toxicity information was lacking for 76% of these chemicals.

Overall, while combined evidence suggests hydraulic fracturing has the potential to impact human health via contamination of drinking water resources, the actual public health impacts are not well understood and not well documented. Available information indicates there are many chemicals within the hydraulic fracturing water cycle that are known to be hazardous to human health, as well as hundreds of chemicals for which toxicological data is limited or unavailable.

In this chapter, our primary goal is to evaluate the availability of toxicity data for a list of chemicals used in hydraulic fracturing fluids or present in produced water, focusing primarily on toxicity values from sources that meet the criteria for inclusion in this assessment, and to highlight chemicals that may pose human health hazards.

9.3 Identification of Chemicals Associated with the Hydraulic Fracturing Water Cycle

As the initial step towards evaluating the hazards of chemicals in the hydraulic fracturing water cycle, the EPA compiled a list of chemicals used in or released by hydraulic fracturing operations across the country.[1] This section describes the compilation of that list. This consolidated list includes a total of 1,606 chemicals, and can be broken down into two sublists: (1) a list of chemicals used in hydraulic fracturing fluids, and (2) a list of chemicals detected in produced water from hydraulically fractured wells (Text Box 9-2).

This list demonstrates the range and variety of chemicals that are associated with the hydraulic fracturing industry. These chemicals should not be considered unique to the hydraulic fracturing

[1] We use the word "chemical" to refer to any individual chemical or chemical substance that has been assigned a CASRN (Chemical Abstracts Service Registry Number). A CASRN is a unique identifier for a chemical substance, which can be a single chemical (e.g., hydrochloric acid, CASRN 7647-01-0) or a mixture of chemicals (e.g., hydrotreated light petroleum distillates (CASRN 64742-47-8), a complex mixtures of C9 to C16 hydrocarbons). For simplicity, we refer to both pure chemicals and chemical substances that are mixtures, which have a single CASRN, as "chemicals."

industry; many of the chemicals used in hydraulic fracturing fluids are widely used industrial chemicals, and many of the chemicals in produced water are naturally occurring. Although this list represents the best information available to the EPA at the time of the assessment, it should not be considered comprehensive. It is likely that, as industry practices change, chemicals may be used or detected that are not included on these lists. Some additional limitations to this chemical list are described in the subsections below.

Text Box 9-2. The EPA's List of Chemicals Identified in Hydraulic Fracturing Fluids and/or Produced Water.

This chemical list progressed through multiple iterations as the EPA's hydraulic fracturing study was developed, culminating in the list of 1,606 chemicals presented in this report.

The first iteration of this chemical list was published in the interim progress report (U.S. EPA, 2012h), and included 1,026 chemicals that were identified from ten sources of information. Seven of these information sources were documents from federal and state governmental units—including the EPA (U.S. EPA, 2011a, e, 2004a; Material Safety Data Sheets), the U.S. House of Representatives (House of Representatives, 2011), the New York State Department of Environmental Conservation (NYSDEC, 2011), and the Pennsylvania Department of Environmental Protection (PA DEP, 2010a)—which obtained data directly from industry. This includes a list of chemicals provided directly to the EPA by nine well operating companies, representing chemicals used in hydraulic fracturing fluids between 2005 and 2009, and a list of chemicals detected by these companies in produced water from 81 wells. The remaining three sources are as follows: a technical report prepared by the Gas Technology Institute for the Marcellus Shale Coalition, which is a drilling industry trade group (Hayes, 2009); a peer-reviewed journal article by Colborn et al. (2011); and the FracFocus Chemical Disclosure Registry, which is a national hydraulic fracturing chemical registry developed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission (GWPC, 2012).

In the external review draft of the EPA's hydraulic fracturing study report (U.S. EPA, 2015d), this chemical list was updated to 1,173 chemicals. The updated chemical list includes the 1,026 chemicals published in the progress report, along with additional chemicals that were identified in the EPA FracFocus 1.0 report (U.S. EPA, 2015a).

For the final version of this assessment, the list has again been updated to include additional chemicals in produced water, which were identified from 18 additional literature sources. The final list includes a total of 1,606 chemicals that have been reported as used in hydraulic fracturing fluids or detected in produced water. The complete list of sources used to compile the final chemical list is provided in Appendix Table H-1. To the extent possible, after chemicals were identified from the sources in Table H-1, the EPA verified the identity of the chemicals used in hydraulic fracturing fluids and detected in produced water of hydraulically fractured wells as described in Appendix Section H.1.

9.3.1 Chemicals Used in Hydraulic Fracturing Fluids

Of the 1,606 total chemicals, the EPA identified 1,084 chemicals as being used in hydraulic fracturing fluids. This list was originally introduced in Chapter 5 of this assessment (Section 5.4), which describes some of the chemical classes and their purpose, and identifies the most frequently used chemicals. This list of 1,084 chemicals is shown in Appendix Table H-2.

Although a total of 8 sources were used to identify the list of chemicals used in hydraulic fracturing fluids, only one source—the EPA analyses based on disclosures submitted to FracFocus—had sufficient information for estimating the frequency with which these chemicals were used (Section 5.4, Text Box 5-1).[1] Of the 1,084 chemicals, 688 were identified in the EPA FracFocus 1.0 report (U.S. EPA, 2015a).[2] Frequency of use for individual chemicals ranged from low (480 chemicals on the list were reported in less than 1% of disclosures nationally) to very high (methanol was reported in 73% of disclosures nationally).

As discussed in Chapter 5, this list provides valuable information on the chemicals used in hydraulic fracturing fluids, but should not be considered complete. For example, in the analysis of the disclosures submitted to the FracFocus 1.0 registry, the EPA was only able to assign standardized chemical names to 65% of ingredient records. The remaining 35% of ingredient records did not have valid CASRNs and were excluded from the analysis because they could not be assigned a standardized chemical name (U.S. EPA, 2015a). In a more recent analysis of data reported to the FracFocus registry through April 2015, Konschnik and Dayalu (2016) found that 80% of chemicals had valid CASRN. That analysis identified an additional 263 CASRNs that are not on the EPA's list of chemicals used in hydraulic fracturing fluids (Dayalu and Konschnik, 2016).

Industry use of CBI is another factor that likely limits the completeness of this chemical list and introduces uncertainty. For example, companies submitting to FracFocus 1.0 were not required to disclose chemicals claimed as CBI. EPA determined that approximately 70% of the disclosures submitted to FracFocus 1.0 contain at least one CBI chemical, and for those disclosures, the average number of CBI chemicals per disclosure was five. Overall, 11% of ingredients were reported to FracFocus 1.0 as CBI (U.S. EPA, 2015a). Konschnik and Dayalu (2016) report a 5.6% increase in the number of CBI ingredients, as well as an increase in the number of disclosures reporting the use of at least one CBI ingredient (Section 5.4; Text Box 5-2).

Although FracFocus disclosures do not provide the name or CASRN of CBI chemicals, the chemical family is sometimes provided. The EPA determined that 79% of CBI ingredient records submitted to FracFocus 1.0 had enough information to partially define the chemical and assign it to a chemical family (U.S. EPA, 2015a). This resulted in the designation of 448 standardized chemical families to which these chemicals could be assigned. The most common standardized chemical families for CBI ingredients were oxyalkylated alcohol (4.7% of CBI ingredient records), petroleum distillates (4.0% of CBI ingredient records), and quaternary ammonium compounds (3.6% of CBI ingredient records) (U.S. EPA, 2015a) (Appendix Table B-1). These standardized chemical family designations are not discussed further in this chapter, but may be useful for site-specific risk assessment, as they

[1] The FracFocus frequency of use data presented in this chapter is based on 35,957 FracFocus disclosures that were deduplicated, within the study time period (January 1, 2011 to February 28, 2013), and with ingredients that have a valid CASRN. In the interest of including as many chemicals as possible, this analysis includes chemicals that do not have valid concentration data. The 692 chemicals includes 16 chemicals that are listed as being used as proppants.

[2] EPA analyses based on disclosures submitted to FracFocus identified 692 unique CASRN. Of these 692, we determined that 4 chemicals are listed under two different CASRN (indicated in the footnote of Appendix Table H-2). Frequency of use data is therefore available for 688 chemicals that were included on EPA's list of chemicals in hydraulic fracturing fluids.

may provide insight into potential physicochemical properties and toxicity of CBI chemicals used at a particular site.

9.3.2 Chemicals Detected in Produced Water

Of the 1,606 total chemicals, the EPA identified 599 as having been detected in produced water. Included among these chemicals are naturally occurring organic compounds, metals, radionuclides, industrial chemicals, and pesticides. These chemicals were originally introduced in Chapter 7 of this assessment, and were compiled from a total of 21 sources. Seventy-seven of the total 599 chemicals in produced water were also identified by at least one of the sources in Appendix H as being used in hydraulic fracturing fluid. However, the EPA used different sets of sources to identify chemicals used in hydraulic fracturing fluids versus those detected in produced water, and there is not a matched comparison between the chemicals used in hydraulic fracturing fluids and returned in produced water at each particular well. Therefore, it is difficult to draw direct comparisons between these two chemical lists, or to use these lists to draw conclusions on the persistence of chemicals in produced water from hydraulically fractured wells. The list of 599 chemicals identified in produced water is shown in Appendix Table H-4.

Although this list provides useful information on the chemical composition of produced water, it is not likely that the data sources were able to capture all of the chemicals present. Chemicals and their metabolites may go undetected in produced water because they were not targeted in the analytical protocols, they were below the limit of detection, or because no standard analytical method exists. Additionally, as discussed in Chapter 7, the composition and concentration of chemicals in produced water will differ depending upon factors like the geology of the formation, the chemicals used for hydraulic fracturing, and the amount of time that has elapsed since hydraulic fracturing. There is therefore expected to be a high degree of local and temporal variation in these chemicals, and there was not sufficient information to determine the frequency with which these chemicals were detected on a national basis.

Concentration data in produced water are available for 175 of these 599 chemicals (Appendix E), including inorganic contributors to salinity (Appendix Tables E-4 and E-5), metals (Appendix Tables E-6 and E-7), radioactive constituents (Appendix Table E-8), and organic constituents (Appendix Tables E-9, E-11, E-12, and E-13). The remaining chemicals were detected in produced water, but concentration was not reported. For these chemicals with concentration data, the measured concentrations spanned several orders of magnitude. For instance, for organic chemicals in produced water from the Marcellus shale formation (Appendix Table E-11), average or median measured concentrations ranged from 2.7 μg/L for N-nitrosodiphenylamine to 400 μg/L for carbon disulfide.

9.4 Toxicological and Physicochemical Properties of Hydraulic Fracturing Chemicals

As the next step towards evaluating the hazards of chemicals in the hydraulic fracturing water cycle, toxicological and physicochemical data were collected as available for each of the chemicals identified in Appendix H. This section describes the compilation of these data, and discusses the extent to which toxicological and physicochemical property data are available for this list of chemicals.

The primary focus of the toxicological analysis in this chapter is on the availability of chronic oral RfVs and OSFs from sources that met the EPA's criteria for inclusion in this study. Qualitative cancer classifications were also identified from these sources when available. This is not intended to be an exhaustive compilation of toxicity values for this chemical list. Rather, it is intended to be a reconnaissance of high-quality toxicological information that met the EPA's criteria for inclusion in this study. If a source of RfVs, OSFs, or qualitative cancer classifications was not included here, that only means that it did not meet the criteria for the purposes of the EPA's study, which are described in this chapter in Section 9.4.1.

Section 9.4.1 describes the criteria used to identify and select RfVs, OSFs, and qualitative cancer classifications, and describes the availability of these toxicological data for the chemicals on the EPA's list of hydraulic fracturing-related chemicals. The next two sections describe additional sources of toxicological information, which may be useful for hazard evaluation when chronic oral RfVs and OSFs are not available: Section 9.4.2 describes the use of a QSAR model to estimate chronic oral toxicity, and Section 9.4.3 describes the availability of additional toxicological information on the EPA's ACToR database. Section 9.4.4 describes other available software tools and approaches that may be used by stakeholders for site-specific risk assessment, but are not utilized in this report. Section 9.4.5 discusses the methods used in this report to generate physicochemical property data, and presents the availability of physicochemical property data for the chemicals on the EPA's list. A brief overview of the toxicity values discussed in the chapter is presented in Text Box 9-3.

As a resource that can be used to support risk assessment at hydraulic fracturing sites, all of the selected RfVs, OSFs, qualitative cancer classifications, QSAR-based toxicity estimates, and physicochemical property data described in this chapter will be compiled into an electronic database that will be publicly accessible via the EPA's website. Additionally, the EPA's compilation of toxicity data for this chemical list has been discussed in two recent manuscripts, both of which focused on the list of 1,173 chemicals that was presented in the external review draft of the EPA's hydraulic fracturing study report (U.S. EPA, 2015d). Yost et al. (2016b) describes the compilation of RfVs and OSFs for the list of 1,173 chemicals. Yost et al. (2016a) describes the use of a QSAR model to estimate toxicity for the list of 1,173 chemicals.

Text Box 9-3. Toxicity Values for Hydraulic Fracturing-Related Chemicals.

Here we provide a brief description of the toxicity values that are presented in this chapter, and how they should be interpreted and used to evaluate chemical hazards. Formal definitions of these terms are footnoted in the chapter and can also be found in the glossary (Appendix J).

Reference value (RfV): RfVs are health-protective values, which describe the dose of a chemical that is likely to be without an appreciable risk of adverse health effects. In general, lower RfVs indicate greater toxicity; however, comparison of RfVs among a set of chemicals requires careful consideration. RfVs are developed by considering the full database of epidemiological and experimental studies available for a particular chemical.

(Text Box 9-3 is continued on the following page.)

Text Box 9-3 (continued). Toxicity Values for Hydraulic Fracturing-Related Chemicals.

These data are used to identify the critical effect, which is the first adverse effect, or its known precursor, that occurs as the dose rate increases (U.S. EPA, 2011f). An RfV is then derived by starting with a quantitative point of departure (POD), which is the toxicological dose-response point that marks the beginning of a low-dose extrapolation for the critical effect, and applying uncertainty factors (UFs) to derive a value for the protection of human health. UFs are applied to account for 5 areas of uncertainty: (1) intraspecies variability; (2) interspecies uncertainty; (3) extrapolation from a subchronic study; (4) extrapolating from a no-observed-adverse-effect level (NOAEL); and (5) deficiencies in the database. A UF of 1, 3, or 10 can be applied for any of these areas of uncertainty depending upon the amount and/or type data available, up to a maximum total UF of 3,000 (U.S. EPA, 2002). Thus, a chemical with a low RfV may reflect high uncertainty in the value, and not necessarily the toxicity of the chemical. Chemicals with a lower total UF generally have more reliable and robust health effect information.

Oral slope factor (OSF): An OSF is a measure of the increased cancer risk from a lifetime oral exposure to an agent. Higher OSFs indicate greater carcinogenic potency. As with RfVs, OSFs are developed by considering the full database of epidemiological and experimental studies for a particular chemical, and evaluating the increase in cancer incidence as dose rate increases. OSFs should be considered in conjunction with qualitative cancer classifications, which characterize the weight of evidence regarding the agent's potential to cause cancer in humans.

No-observed-adverse-effect level (NOAEL): NOAEL is defined as the highest exposure level at which there are no biologically significant increases in the frequency or severity of adverse effect between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered adverse or precursors of adverse effects (U.S. EPA, 2011f).

Lowest-observed-adverse-effect level (LOAEL): LOAEL is defined as the lowest exposure level at which there are biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control group (U.S. EPA, 2011f). Lower LOAELs indicate greater toxicity.

Maximum contaminant level (MCL): MCLs are the highest level of a contaminant that is allowed in drinking water. MCLs are set as close to the maximum contaminant level goal (MCLG) as feasible using the best available analytical and treatment technologies and taking cost into consideration. MCLs are enforceable standards. The MCLG is a non-enforceable health benchmark goal which is set at a level at which no known or anticipated adverse effect on the health of persons is expected to occur and which allows an adequate margin of safety (U.S. EPA, 2012b). Whereas RfVs, LOAELs, and NOAELs are expressed in terms of dose (mg/kg-day), MCLs are expressed in terms of the concentration of an agent in water (μg/L).

9.4.1 Reference Values (RfVs), Oral Slope Factors (OSFs), and Qualitative Cancer Classifications

For the purpose of this study, the EPA's primary goal was to identify high quality toxicity values that met the criteria for inclusion in this study, and that could be used by stakeholders to support the risk assessment of hydraulic fracturing chemicals (Text Box 9-1). Briefly, the sources of RfVs, OSFs, and qualitative cancer classifications selected by the EPA for the purposes of this chapter met the following key criteria:

1. The body or organization generating or producing the peer-reviewed RfVs, peer-reviewed OSFs, or peer-reviewed qualitative assessment must be a governmental or intergovernmental body.
2. The data source must include peer-reviewed RfVs, peer-reviewed OSFs, or peer reviewed qualitative assessments.
3. The RfVs, OSFs, or qualitative assessments must be based on peer-reviewed scientific data.
4. The RfVs, OSFs, or qualitative assessments must be focused on protection of the general public.
5. The body generating the RfVs, OSFs, or qualitative assessments must be free of conflicts of interest with respect to the chemicals for which it derives reference values or qualitative assessments.

These five criteria were developed by the EPA specifically for the purpose of this assessment, and are similar to the EPA Office of Solid Waste and Emergency Response (OSWER) recommendations for selecting toxicity values in conducting site-specific risk assessments (Regional Tier 3 Toxicity Value Workgroup, 2013; U.S. EPA, 2003a, 1989).[1] The OSWER directives provide recommendations on the appropriate sources of toxicity values and toxicological information that should be considered in risk assessments, and were designed to recognize toxicity values that were developed using the best available scientific information. In addition, these directives outline references to various resources that provide guidance on the approaches and issues considered in deriving toxicity values. This type of information can be especially important in cases in which multiple sources of toxicity values need to be considered or evaluated, or in which a value needs to be developed. More detail on these criteria for selection and inclusion of data sources, as well as the full list of data sources that were considered for this study, are available in Appendix G.

Table 9-1 shows the data sources that met these five criteria for the selection of toxicological information. The federal databases of RfVs or OSFs that met these criteria are the EPA's Integrated Risk Information System (IRIS) database, the EPA's Provisional Peer-Reviewed Toxicity Value (PPRTV) database, the EPA's Human Health Benchmarks for Pesticides (HHBP) database, and the Agency for Toxic Substances and Disease Registry (ATSDR) database. IRIS and PPRTV also provide qualitative cancer classifications. One state source of RfVs and OSFs, the California Environmental Protection Agency (CalEPA) Toxicity Criteria Database, met the criteria for inclusion.[2] One intergovernmental source of RfVs, the World Health Organization (WHO) International Programme on Chemical Safety (IPCS) Concise International Chemical Assessment Documents (CICAD), met the criteria for inclusion. The International Agency for Research on Cancer (IARC) and U.S. National Toxicology Program (NTP) Report on Carcinogens (RoC) also met the criteria and were used as additional sources for qualitative cancer classifications.

[1] OSWER changed its name to the Office of Land and Emergency Management (OLEM), effective December 15, 2015.

[2] State RfVs and OSFs are also publicly available from Alabama, Texas, Hawaii, and Florida, but they did not meet the criteria for consideration as sources for RfVs and OSFs in this report. See Appendix G for details.

Table 9-1. Sources of selected RfVs, OSFs, and qualitative cancer classifications.

Type of toxicological Information	Data source	Website
RfVs, OSFs, and qualitative cancer classifications	EPA Integrated Risk Information System (IRIS) database	http://cfpub.epa.gov/ncea/iris/index.cfm?fuseaction=iris.showSubstanceList
RfVs, OSFs, and qualitative cancer classifications	EPA Provisional Peer-Reviewed Toxicity Value (PPRTV) database	http://hhpprtv.ornl.gov/index.html
RfVs, OSFs	EPA Human Health Benchmarks for Pesticides (HHBP) database	http://iaspub.epa.gov/apex/pesticides/f?p=HHBP:home
RfVs	Agency for Toxic Substances and Disease Registry (ATSDR) Minimum Risk Levels	http://www.atsdr.cdc.gov/toxprofiles/index.asp#bookmark05
RfVs, OSFs	California Environmental Protection Agency (CalEPA) Toxicity Criteria Database	http://oehha.ca.gov/tcdb/index.asp
RfVs	World Health Organization (WHO) International Programme on Chemical Safety (IPCS) Concise International Chemical Assessment Documents (CICAD)	http://www.who.int/ipcs/publications/cicad/en/
Qualitative cancer classifications	National Toxicology Program (NTP) 13th Report on Carcinogens (RoC)	https://ntp.niehs.nih.gov/pubhealth/roc/
Qualitative cancer classifications	International Agency for Research on Cancer (IARC) Monographs	http://monographs.iarc.fr/

In addition to the sources in Table 9-1, we also consulted the NPDWRs and DWSHA tables (U.S. EPA, 2014a) to determine whether the chemicals on this list are regulated as drinking water contaminants. NPDWRs provide a list of MCLs, which are legally enforceable standards on the concentration of a substance that is allowed in drinking water under the Safe Drinking Water Act. In this chapter, MCL values are referenced as a means of comparison with reported concentration data where appropriate, and are reported in Appendix G and are compiled on the EPA's electronic database for the hydraulic fracturing study.

As noted above, this chapter focuses on the presentation and use of chronic RfVs. Chronic RfVs account for the potential that chemical exposure may be continuous, in low concentration, and over a longer duration. In the absence of reliable information on the potential duration of chemical exposure, this is a conservative assumption for the protection of human health. Chronic RfVs are also lower than less-than-chronic RfVs (e.g., acute, intermediate, or subchronic toxicity values), and are therefore more health protective. For these reasons, chronic RfVs are generally preferred as the default by risk assessors when conducting site-specific risk assessments (U.S. EPA, 1989) and when developing regional screening levels (U.S. EPA, 2016b). In contrast, acute RfVs are more applicable for single exposures and/or exposures of limited frequency to high concentration and shorter durations (e.g., emergencies). However, the availability of less-than chronic RfVs are also presented for the sake of completeness.

Some chemicals had chronic oral RfVs or OSFs available from more than one of the sources in Table 9-1. For these chemicals, we selected a single value for use in this chapter by applying a modification of the EPA OSWER Directives 9285.7-53 and 9285.7-86 tiered hierarchy of toxicity values (U.S. EPA, 2003a). A single RfV and/or OSF was selected from the sources in this order: HHBP (pesticides only), IRIS, PPRTV, ATSDR, and then other available values. The RfVs considered from these sources included chronic oral reference doses (RfDs) from the IRIS, PPRTV, and HHBP programs; chronic oral minimal risk levels (MRLs) from ATSDR; oral maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intakes (TDIs) from CICAD.[1,2,3,4,5]

Of the 1,606 chemicals identified by the EPA, 173 (11%) have federal, state, or international chronic oral RfVs and/or OSFs from sources listed in Table 9-1. Chronic oral RfVs and/or OSFs from the selected sources are lacking for the remaining 1,433 (89%) chemicals that the EPA has identified as associated with hydraulic fracturing. All available chronic oral RfVs and OSFs from the sources listed in Table 9-1 are tabulated in Appendix G. Chronic oral RfVs and OSFs for chemicals used in hydraulic fracturing fluids are listed in Appendix Tables G-1a through G-1c, and chronic oral RfVs and OSFs for chemicals reported in hydraulic fracturing flowback or produced water are listed in Appendix Tables G-2a through G-2c. The EPA's IRIS database was the most abundant source of these toxicity values.

Overall, when chemicals in hydraulic fracturing fluid and chemicals in produced water are considered separately, the availability of chronic RfVs and OSFs can be summarized as follows:

- For the 1,084 chemicals used in hydraulic fracturing fluid, chronic oral RfVs or OSFs from at least one of the selected federal, state, and international sources were available for 98 chemicals (9%). From the US federal sources alone, chronic oral RfVs were available for 81 chemicals (7%), and OSFs were available for 15 chemicals (1%).

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[1] The OSWER hierarchy indicates that sources should be used in this order: IRIS, PPRTV, and then other values. In this report, this hierarchy was followed, but HHBP values were used in lieu of an IRIS value for a few chemicals that are pesticides.

[2] An RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It can be derived from a NOAEL, LOAEL, or benchmark dose, with uncertainty factors generally applied to reflect limitations of the data used. Generally used in the EPA's non-cancer health assessments (U.S. EPA, 2011f). This estimate is expressed in terms of mg/kg-day.

[3] An MRL is an estimate of daily human exposure to a hazardous substance at or below which the substance is unlikely to pose a measurable risk of harmful (adverse), non-cancerous effects. MRLs are calculated for a route of exposure (inhalation or oral) over a specified time period (acute, intermediate, or chronic). MRLs should not be used as predictors of harmful (adverse) health effects (ATSDR, 2016). Chronic MRL: Duration of exposure is 365 days or longer. This estimate is expressed in terms of mg/kg-day.

[4] An MADL is the maximum allowable daily level of a reproductive toxicant at which the chemical would have no observable adverse reproductive effect, assuming exposure at 1,000 times that level (OEHHA, 2012). This estimate is expressed in terms of μg/day.

[5] A TDI is an estimate of the intake of a substance, expressed on a body mass basis, to which an individual in a (sub) population may be exposed daily over its lifetime without appreciable health risk (WHO, 2015). This estimate is expressed in terms of mg/kg-day.

- For the 599 chemicals reported in produced water, chronic oral RfVs or OSFs from at least one of the selected federal, state, and international sources were available for 120 chemicals (20%). From the US federal sources alone, chronic oral RfVs were available for 97 chemicals (16%), and OSFs were available for 30 chemicals (5%).

In addition to these chronic values, some of the chemicals also have less-than-chronic oral RfVs available from the sources listed in Table 9-1. Subchronic, acute, or intermediate oral RfVs were identified for 103 chemicals on the consolidated list, including 60 chemicals used in hydraulic fracturing fluid (Appendix Table G-1d), and 73 chemicals reported in produced water (Appendix Table G-2d). The majority of these chemicals also had chronic oral RfVs available, although there were 10 chemicals that had less-than-chronic oral RfVs but lacked a chronic oral RfV. All of these less-than-chronic RfVs were found on the PPRTV, ATSDR, or HHBP databases. As stated above, chronic values more protective of human health than less-than-chronic values, and are generally preferred for risk assessment. These less-than-chronic values are therefore not discussed further in this report, but are provided in Appendix G as supporting information.

Of the 1,606 chemicals identified by EPA, 207 (13%) had a qualitative cancer classification available from at least one of the sources listed in Table 9-1, which include IRIS, PPRTV, IARC, and RoC. These classifications are based on the weight-of-evidence (WOE) that a chemical causes cancer in humans. Of these 207 chemicals:

- 21 were reported by at least one source to be a known carcinogen in humans.
- 66 were reported by at least one source to be a probable or possible carcinogen in humans. These chemicals have been demonstrated to be carcinogenic in animal models, but have limited or insufficient data to adequately assess carcinogenicity in humans.
- 117 were reported to be not classifiable as to carcinogenicity in humans. These chemicals have been evaluated by at least one of these sources for their potential to cause cancer, but had inadequate evidence from human exposure and animal studies to assess carcinogenic potential.
- 3 were reported as not likely to be a human carcinogen.

The complete list of chemicals with qualitative cancer classifications are shown in Appendix Table G-1e (chemicals in hydraulic fracturing fluids) and Appendix Table G-2e (chemicals in produced water).

9.4.2 Estimating Toxicity Using Quantitative Structure Activity Relationship (QSAR) Modeling

Because the majority of chemicals identified in this report do not have RfVs and/or OSFs from the selected sources, it is likely that risk assessors at the local and regional level may turn to alternative sources of toxicological information. One potential resource is QSAR modeling software, which is able to provide estimates or predictions of toxicity based on chemical structure. A key advantage to QSAR models is that they are able to rapidly and inexpensively estimate toxicity values for chemicals. A disadvantage is that QSAR estimates may be of higher uncertainty and less reliable than values generated using traditional toxicological methods. However, because they increase the available pool of toxicity information, QSAR estimates may be a useful resource for risk assessors that are faced with evaluating potential exposures to data-poor chemicals.

A recent study by Yost et al. (2016a) used TOPKAT (Toxicity Prediction by Komputer Assisted Technology) QSAR software to estimate toxicity for the EPA's list of chemicals used in hydraulic fracturing fluids or detected in produced water, and evaluated how effectively these toxicity estimates could be used to rank chemicals based on toxicity. The chemical list examined in this study is the list of 1,173 chemicals published in the external review draft of the EPA's hydraulic fracturing study report (U.S. EPA, 2015d) (Text Box 9-2), so the full list of 1,606 chemicals was not assessed using the QSAR model. TOPKAT is commercially available QSAR software that is able to estimate the rat chronic oral lowest-observed-adverse-effect level (LOAEL), which is the LOAEL measured in a rat model following chronic oral exposure to a chemical.[1]

The authors of this study used TOPKAT to generate rat chronic oral LOAEL estimates for EPA's list of chemicals, and assigned qualitative confidence scores (high, medium, or low) to each estimate based on parameters reported by the model. The authors then examined a list of 48 chemicals that had both a high-confidence TOPKAT LOAEL estimate and a chronic oral reference dose (RfD) from EPA's IRIS database. The authors ranked these 48 chemicals from most toxic to least toxic based on either TOPKAT LOAEL estimate or on IRIS chronic oral RfD, and then used Spearman rank correlation to examine the similarity between these chemical rankings.

Of the 1,173 hydraulic fracturing chemicals, TOPKAT was able to generate toxicity estimates for 515 (44%) of the chemicals, including 453 chemicals that are used in hydraulic fracturing fluids, and 86 chemicals that have been detected in produced water. The authors found a strong and statistically significant correlation between chemical rankings based on high-confidence TOPKAT LOAEL estimates and on IRIS chronic oral RfDs, indicating that high-confidence TOPKAT LOAEL estimates can effectively be used to rank chemicals based on toxicity when experimentally derived toxicity values are not available. Overall, TOPKAT LOAEL estimates were available for 417 chemicals in this study that lack chronic oral RfVs or OSFs from the sources identified by EPA. Of these, 389 were found to be high-confidence estimates.

When available, the high-confidence TOPKAT LOAEL estimates from Yost et al. (2016b) are discussed in this chapter as an additional resource that can be used to rank chemicals based on toxicity. Low- or medium-confidence TOPKAT LOAEL estimates are not shown in this chapter, as the use of these values for chemical ranking has not been validated.

[1] LOAEL is defined as the lowest exposure level at which there are biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control group following chronic (lifetime) exposure.

9.4.3 Chemical Data Available from EPA's Aggregated Computations Toxicology Resource (ACToR) Database

An additional tool for obtaining information focused on toxicology and risk assessment is the EPA's ACToR database.[1] ACToR is a large data warehouse developed by the EPA to consolidate large and disparate amounts of public data on chemicals, including data on chemical identity, structure, physicochemical properties, in vitro assay results, and in vitro toxicology data. The primary goals of ACToR are to make information on chemical health effects and exposure potential readily accessible, to characterize chemical toxicological data gaps, and to provide a resource for model building to address data gaps in environmental risk information (Judson et al., 2012).

ACToR contains data on over 500,000 chemicals from over 2,500 data sources, covering many domains including hazard, exposure, risk assessment, risk management, and use. Data sources and collections in ACToR include the US EPA, National Institutes of Health (NIH), the Centers for Disease Control and Prevention (CDC), US Food and Drug Administration (FDA), State Agencies, the European Chemicals Agency (ECHA), corresponding government agencies in Canada (e.g., Health Canada), Europe and Japan, the World Health Organization (WHO), and non-governmental organizations (NGOs). Data within ACToR ranges from the federal RfVs and OSFs discussed in Section 9.3.1, which have undergone extensive peer review, to other toxicity values and study and test results that have undergone little to no peer review.

ACToR organizes these data into several levels of "assays" and "assay categories," which serve to classify data sets according to the nature of the data. For instance, the "Hazard" assay category includes all data that are associated directly or indirectly with toxicology experiments. The "Risk Management" assay category includes regulatory and non-regulatory risk management benchmarks. Considering the diversity and overlapping nature of the data resources within ACToR, a single data set may fall into multiple assay categories (Judson et al., 2012).

We searched the ACToR database for information related to the list of 1,606 hydraulic fracturing-related chemicals. Specifically, we searched within the "Hazard" and "Risk Management" assay categories of ACToR. Results of the query were then filtered to include the assays that are most relevant to chemical exposure via drinking water. These assays were assigned into the following nine data classes: carcinogenicity, dose response values, drinking water criteria, genotoxicity or mutagenicity, hazard identification, LOAEL/NOAEL, RfV, OSF, and water quality criteria.[2]

Of the 1,606 chemicals, it was found that 735 (46%) have some data available within these data classes on ACToR, with the total number of data points found for individual chemicals ranging from 1 to 243. Figure 9-2 shows the percentage of the total 1,606 chemicals that had data available in each of the nine ACToR data classes, and indicates the fraction of those chemicals that also had a chronic oral RfV or OSF available from at least one of the selected sources in Table 9-1. As can be seen in Figure 9-2, 37% of the chemicals had some information on hazard identification, 25% had

[1] The ACToR database, including the full list of data collections and assays, is available at: http://actor.epa.gov.

[2] NOAEL is defined as the highest exposure level at which there are no biologically significant increases in the frequency or severity of adverse effect between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered adverse or precursors of adverse effects. Source: U.S. EPA (2011f).

information on carcinogenicity, and 24% had a LOAEL or NOAEL identified. A LOAEL and/or NOAEL identified from a well conducted dose-response study are often considered the minimum data needed for RfV derivation (U.S. EPA, 2002).

Focusing on the 1,433 chemicals that lacked a chronic RfV and/or OSF from the selected sources described in Section 9.3.1, 567 (40%) had available data within at least one of these data classes on ACToR. Thus, ACToR has a significant amount of potentially useful data on chemical hazards, including for some data-poor chemicals, and might help to fill data gaps in the ongoing effort to understand potential hazards of hydraulic fracturing chemicals.

It is outside the scope of this assessment to evaluate the quality and reliability of data within ACToR that has not already undergone peer review. Therefore, with the exception of data from the sources listed in Table 9-1, data from ACToR was not considered in the hazard evaluation presented in this chapter. However, as a potential resource for risk assessors, the tables in this chapter indicate whether a chemical had data available on ACToR.

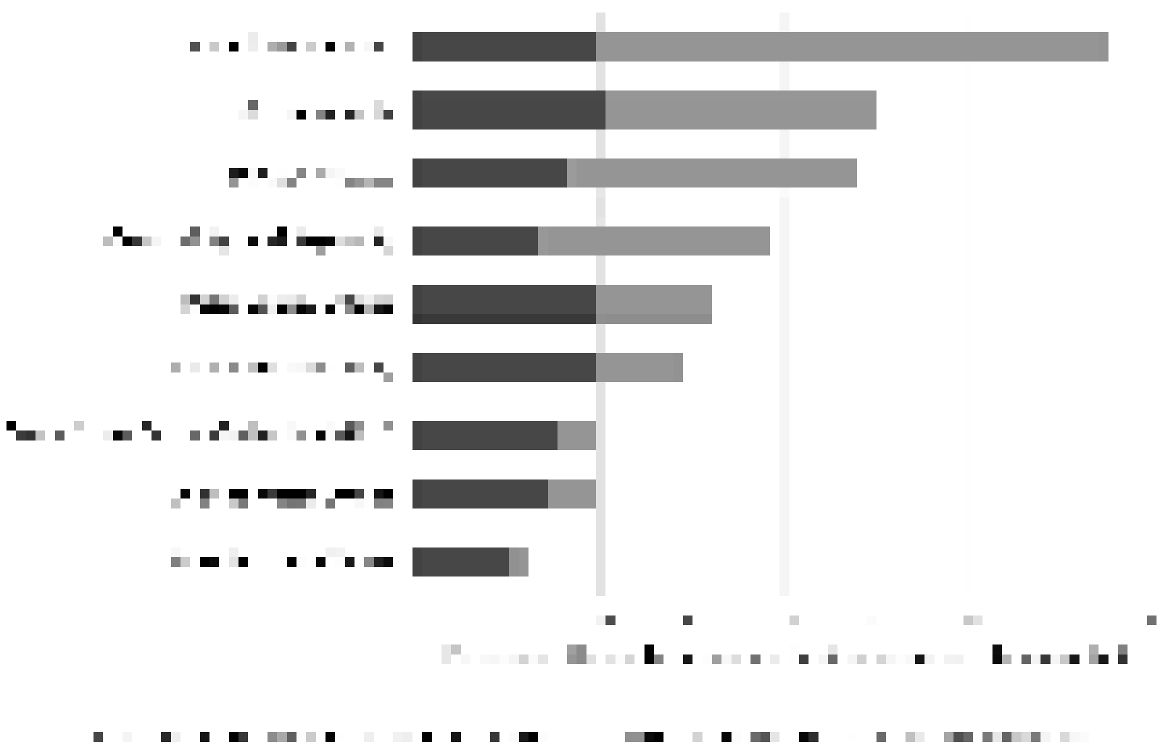

Figure 9-2. Percentage of hydraulic fracturing-related chemicals (out of 1,606 total) with at least one data point in each ACToR data class.

9.4.4 Additional Tools for Hazard Evaluation

In addition to the methods and approaches utilized in this chapter, there are other potential tools and approaches that could be used by stakeholders to prioritize and estimate toxicity of chemicals that have a limited toxicity database. We briefly describe three such approaches in Appendix G (Section G.4): the Threshold of Toxicological Concern (TTC) approach, the Organisation for

Economic Co-operation and Development (OECD) QSAR Toolbox, and the application of data from high throughput screening (HTS) assays. Toxicity predictions from these additional data sources can be either quantitative or qualitative, and may be used to fill and address gaps related to risk assessment.

Although these additional tools may be potentially useful for the evaluation of chemical hazards, they currently have limited utility in this chapter, and are not discussed further. The TTC approach requires an estimate of human intake, which is challenging for hydraulic fracturing-related chemicals, since the potential for human exposure is generally not clear. The OECD QSAR Toolbox is potentially useful for qualitative assessment, and may be useful for quantitative toxicity assessment as its human health hazard and repeated dose toxicity databases expand. HTS assays are an emerging technology, and the potential application of these data for human health risk assessment is not well understood. These tools would be more appropriately applied by stakeholders on a site-specific basis, as preliminary steps to identify potential chemicals of concern.

9.4.5 Physicochemical Properties

As presented in Chapter 5, EPI Suite™ software was used to generate data on the physicochemical properties of the hydraulic fracturing-related chemicals identified by EPA. EPI Suite provides an estimation of physicochemical properties based upon chemical structure, and will additionally provide experimentally measured values for these properties when they are available for a given chemical. For more details on this software and on the use of physicochemical properties for fate and transport estimation, see Chapter 5.

From the total list of 1,606 chemicals associated with hydraulic fracturing, EPI Suite was able to generate data on physicochemical properties for 917 (57%) of the chemicals (Appendix H). This includes 455 chemicals that are reported in hydraulic fracturing fluids, 521 chemicals that have been reported in produced water, and 59 chemicals that were both used in hydraulic fracturing fluids and reported in produced water. The remaining 689 chemicals on EPA's total list lacked the structural information necessary to generate estimates.

In addition to EPI Suite, two other software programs were consulted to generate physicochemical property data for EPA's list of hydraulic fracturing-related chemicals. QikProp (Schrodinger, 2012) and LeadScope (Leadscope Inc., 2012) are commercial products designed primarily as drug development and screening tools, which are able to estimate properties related to chemical fate and transport as well as pharmacokinetics. Properties generated by QikProp and LeadScope are generally more relevant to drug development than to environmental assessment. The properties generated by QikProp and LeadScope were not used in the analysis presented in this report, but will be compiled on the electronic database for EPA's hydraulic fracturing study.

9.4.6 Summary of Available Toxicological and Physicochemical Information for Hydraulic Fracturing Chemicals

Figure 9-3 summarizes the toxicological and physicochemical information that is available for the list of hydraulic fracturing chemicals identified by EPA in this study. This figure also summarizes the availability of data on the occurrence of these chemicals in hydraulic fracturing fluids (frequency of use) or in produced water (measured concentrations).

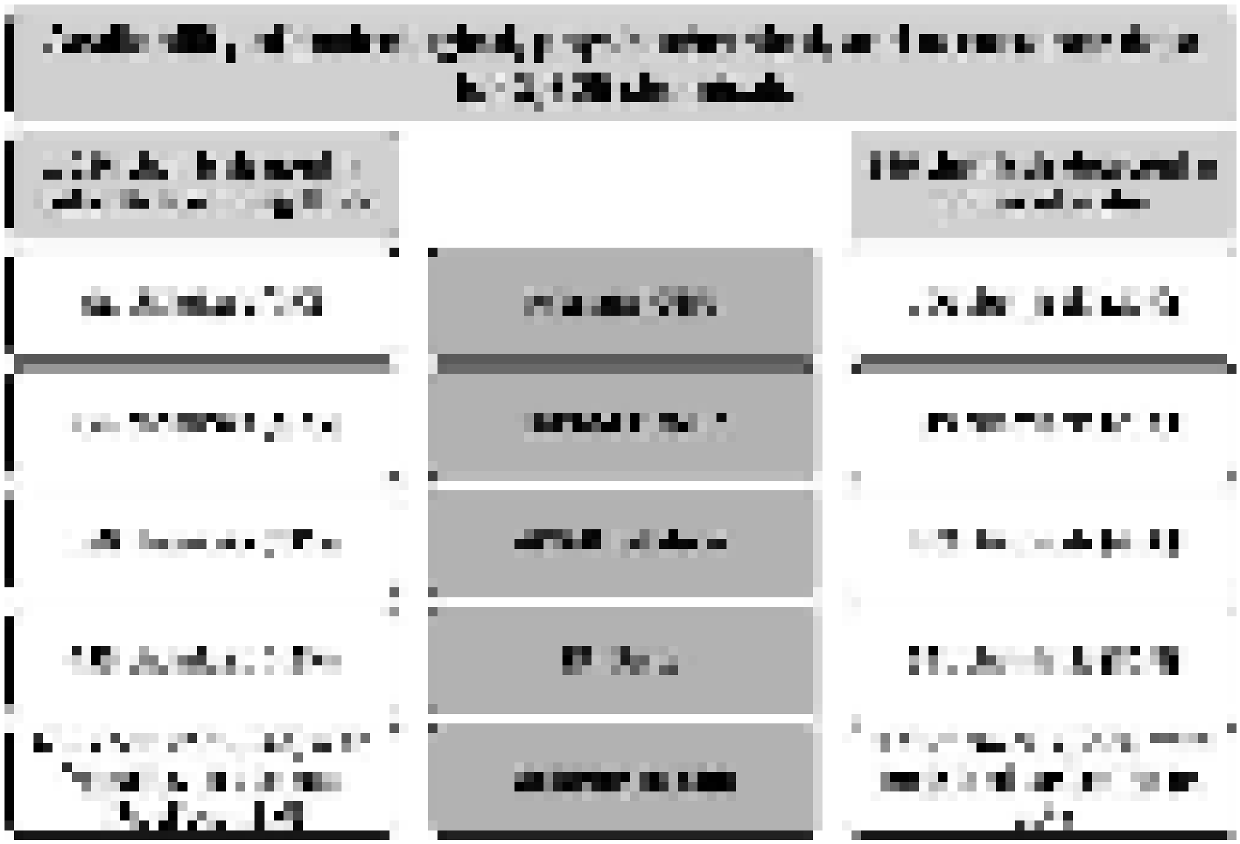

Figure 9-3. Overall representation of the selected toxicological, physicochemical, and occurrence data available for the 1,606 hydraulic fracturing-related chemicals identified by the EPA.

Overall, there is a clear paucity of chronic oral RfVs and OSFs for this list of chemicals, indicating that the majority of chemicals associated with hydraulic fracturing activity have not undergone significant toxicological assessment. QSAR-based toxicity estimates (TOPKAT LOAELs) were available for a larger number of these chemicals, and were often available for chemicals that lack chronic oral RfVs and OSFs. EPA's ACToR database offers additional toxicological data that may be useful for the hazard evaluation of these chemicals, although the quality and reliability of the data for these chemicals within ACToR was not evaluated here.

9.5 Hazard Identification of Hydraulic Fracturing Chemicals

This section focuses on the hazard identification of subsets of chemicals that were identified as being of particular interest in previous chapters of this report, or which otherwise may be of particular interest to risk assessors. For these chemicals, we summarize what is known about events that may lead to the entry of these chemicals into drinking water resources. We provide examples of recent studies that have reported these chemicals in drinking water resources, including examples in which these chemicals have been reported at concentrations exceeding MCLs. We then summarize the available toxicological data for these chemicals, including chronic oral RfVs, OSFs, cancer classifications, QSAR-based toxicity estimates (TOPKAT LOAELs), and the availability of relevant toxicological information from EPA's ACToR database—and indicate which chemicals are regulated by EPA as drinking water contaminates.

We focused on the following subsets of chemicals:

1. Chemicals used in hydraulic fracturing fluids (Chapter 5)
2. Organic chemicals that may be returned to the surface in produced water, including naturally occurring hydrocarbons such as BTEX (Chapter 7)
3. Inorganic chemicals that may be returned to the surface in produced water, including metals, inorganic ions, and technologically enhanced naturally occurring radioactive material (TENORM) (Chapter 7)
4. Methane in stray gas, which has been reported in drinking water resources in areas of hydraulic fracturing activity (Chapter 6)
5. Disinfection byproducts (DBPs) that may be formed from wastewater constituents (Chapter 8)
6. Banned chemicals reported in produced water, specifically organochlorine pesticides and polychlorinated biphenyls (PCBs).
7. Chemicals on EPA's consolidated list that were reported in both hydraulic fracturing fluids and produced water

The hazard identification for these subsets of chemicals is presented below.

9.5.1 Chemicals Used in Hydraulic Fracturing Fluids

Chapter 5 provided an overview of chemicals that are used in hydraulic fracturing fluids. These chemicals have the potential to enter drinking water resources through events such as spills of hydraulic fracturing fluids, injection of hydraulic fracturing fluids directly into groundwater, and leakoff of fluids into the formation. These chemicals may also persist in produced water, and may enter drinking water resources through spills or releases of produced water or inadequately treated wastewater.

Several recent field studies have detected chemicals that are commonly used in hydraulic fracturing fluids in groundwater near hydraulically fractured wells. In some cases, the origin of the chemicals could be clearly linked to hydraulic fracturing activity. For example, in Killdeer, North Dakota (Section 6.2.2.1), evidence strongly suggests a well blowout during hydraulic fracturing led to the contamination of a drinking water aquifer with tert-butyl alcohol, a degradation product of tert-butyl hydroperoxide used in hydraulic fracturing fluids at that site (U.S. EPA, 2015i). In groundwater monitoring wells in the Pavillion Field in Wyoming, Digiulio and Jackson (2016) reported detections of organic chemicals used in hydraulic fracturing fluids at that site, including 2-

butoxyethanol, naphthalene, 1,2,4-trimethylbenzene, diethylene glycol, methanol, ethanol, and isopropanol, likely as a result of shallow hydraulic fracturing in that region.

Other studies provide indirect evidence that chemical contaminants originated from hydraulic fracturing activity. For example, in the Marcellus Shale in Pennsylvania, Llewellyn et al. (2015) detected trace levels of 2-butoxyethanol in water wells near several hydraulically fractured wells, with multiple lines of evidence suggesting that the chemical originated from a surface spill or leak related to hydraulic fracturing activity. In northeastern Pennsylvania, Drollette et al. (2015) found trace concentrations of known constituents of hydraulic fracturing fluid in private residential groundwater wells, including di(2-ethylhexyl) phthalate, with evidence suggesting that the chemicals originated from known surface spills of hydraulic fracturing fluids. In the Barnett Shale, Texas, a survey of water quality in public and residential wells reported chemicals that are known to be used in hydraulic fracturing fluids, including methanol, ethanol, isopropanol, and propargyl alcohol, but it was not clear whether these chemicals originated from hydraulic fracturing activity or from other potential sources (Hildenbrand et al., 2015).

Table 9-2 shows the list of chemicals that were reported in at least 10% of disclosures nationally in the EPA FracFocus 1.0 project database (excluding water, quartz, and sodium chloride), and shows the noncancer toxicity data (chronic oral RfVs and TOPKAT LOAEL estimates) and ACToR data available for these chemicals.[1] Cancer information is provided in Table 9-3. Nine (26%) of these 34 chemicals have a chronic oral RfV available from at least one of the sources in Table 9-1. Chronic oral RfVs ranged from 0.002 mg/kg-day (propargyl alcohol) to 2 mg/kg-day (methanol and ethylene glycol). Critical effects for these chemicals include kidney/renal toxicity, hepatotoxicity, developmental toxicity (extra cervical ribs), reproductive toxicity, neurotoxicity, and decreased terminal body weight. Only one of these chemicals, sodium chlorite, is regulated in drinking water under the NPDWRs.

Of the 25 chemicals that lack chronic oral RfVs, 11 have high-confidence TOPKAT LOAEL estimates available. Of these, methenamine (~14% of disclosures) had the lowest TOPKAT LOAEL estimate, and choline chloride (~15% of disclosures) had the second lowest. All but five of these chemicals had at least some relevant toxicological data available on EPA's ACToR database.

[1] The analysis of the FracFocus 1.0 project database presented in this chapter did not exclude chemicals that lacked valid concentration data, in order to present a more inclusive analysis of the potential toxicity of chemicals used in hydraulic fracturing fluids. The chemical list and percent disclosures listed for each chemical is therefore slightly different that those shown in Chapter 5 (Table 5-3), which excluded chemicals lacking valid concentration data.

Table 9-2. Chemicals reported to FracFocus 1.0 from January 1, 2011 to February 28, 2013 in 10% or more disclosures, with the percent of disclosures for which each chemical is reported. Chronic oral RfVs, TOPKAT LOAEL estimates, and availability of ACToR data are shown when available.

Chemicals are ordered in the table, from high to low, based on their number of disclosures in the EPA FracFocus 1.0 project database. Water, quartz, and sodium chloride are excluded from this analysis. Asterisk (*) indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

			Chronic oral RfV[b]			QSAR	ACToR
Chemical Name	**CASRN**	**% of Disclo-sures[a]**	**RfV (mg/kg-day)**	**Source of RfV**	**Critical effect[c]**	**TOPKAT LOAEL[d] (mg/kg)**	**# of data points[e]**
Methanol	67-56-1	73%	2	IRIS	Extra cervical ribs		122
Distillates, petroleum, hydrotreated light	64742-47-8	67%					4
Hydrochloric acid	7647-01-0	66%					50
Ethylene glycol	107-21-1	47%	2	IRIS	Kidney toxicity	130	102
Isopropanol	67-63-0	46%				81.4	26
Diammonium peroxydisulfate	7727-54-0	44%					11
Guar gum	9000-30-0	39%					2
Sodium hydroxide	1310-73-2	39%					26
Propargyl alcohol	107-19-7	33%	0.002	IRIS	Renal and hepatotoxicity		42
Glutaraldehyde	111-30-8	33%				398	13
Ethanol	64-17-5	31%				59.2	182
Potassium hydroxide	1310-58-3	31%					21
Acetic acid	64-19-7	25%				183	35
Citric acid	77-92-9	24%				55.8	25
2-Butoxyethanol	111-76-2	23%	0.1	IRIS	Hemosiderin deposition in the liver	707	44
Solvent naphtha, petroleum, heavy arom.	64742-94-5	21%					5
Naphthalene	91-20-3	19%	0.02	IRIS	Decreased terminal body weight	67.5	157

Chemical Name	CASRN	% of Disclo-sures[a]	Chronic oral RfV[b]			QSAR	ACToR
			RfV (mg/kg -day)	Source of RfV	Critical effect[c]	TOPKAT LOAEL[d] (mg/kg)	# of data points[e]
2,2-Dibromo-3-nitrilopropionamide	10222-01-2	16%				52.4	
Choline chloride	67-48-1	15%				20.8	24
Phenol-formaldehyde resin	9003-35-4	14%					
Carbonic acid, dipotassium salt	584-08-7	14%				137	3
Methenamine	100-97-0	14%				12.3	15
Thiourea, polymer with formaldehyde and 1-phenylethanone	68527-49-1	13%					
1,2,4-Trimethylbenzene	95-63-6	13%	0.01	IRIS	Decreased pain sensitivity	91.5	71
Polyethylene glycol	25322-68-3	13%					5
Polyethylene glycol nonylphenyl ether	9016-45-9	13%					4
Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides	68424-85-1	12%	0.44	HHBP	Decreased body weight and weight gain		3
Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy branched	127087-87-0	12%					
Ammonium chloride	12125-02-9	12%					18
Formic acid	64-18-6	11%	0.9	PPRTV	Reproductive toxicity		72
Tetrakis(hydroxy-methyl) phosphonium sulfate	55566-30-8	11%				148	3
Sodium chlorite*	7758-19-2	11%	0.03	IRIS	Neuro-developmental effects		66

Chemical Name	CASRN	% of Disclo-sures[a]	Chronic oral RfV[b]			QSAR	ACToR
			RfV (mg/kg -day)	Source of RfV	Critical effect[c]	TOPKAT LOAEL[d] (mg/kg)	# of data points[e]
Alcohols, C12-14, ethoxylated propoxylated	68439-51-0	11%				1450	
Sodium persulfate	7775-27-1	10%					6

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Value; HHBP = Human Health Benchmarks for Pesticides; QSAR = Quantitative structure-activity relationship; TOPKAT = Toxicity Prediction by Komputer Assisted Technology; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] The FracFocus frequency of use data presented in this chapter is based on 35,957 FracFocus disclosures that were deduplicated, within the study time period (January 1, 2011 to February 28, 2013), and with ingredients that have a valid CASRN.

[b] Reference value (RfV): An estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[c] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[d] TOPKAT LOAEL: The LOAEL is the lowest exposure level at which there are biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group. TOPKAT LOAELs were predicted using a QSAR-based software model, as described in Section 9.4.2. Values are rounded to 3 significant figures.

[e] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

Table 9-3 shows the chemicals reported in at least 10% of disclosures nationally in the EPA FracFocus 1.0 project database that are considered by at least one of the sources in Table 9-1 to be known, probable, or possible human carcinogens. Ethanol is classified as a "carcinogenic to humans" (Group 1) by IARC. Naphthalene is classified by IARC as "possibly carcinogenic to humans" (Group 2B), and is classified by RoC as "reasonably anticipated to be a human carcinogen," while IRIS classifies naphthalene as having inadequate data to assess carcinogenic potential. Neither chemical has an available OSF.

Table 9-3. List of OSFs and qualitative cancer classifications available for all carcinogenic chemicals reported to FracFocus 1.0 from January 1, 2011 to February 28, 2013 in 10% or more disclosures.

Includes all chemicals from Table 9-2 that are classified as known, probable, or possible human carcinogens by at least one of the sources in Table 9-1.

		OSF[a]		Qualitative cancer classification			
Chemical Name	**CASRN**	**OSF (per mg/kg-day)**	**Source of OSF**	**IRIS[b]**	**PPRTV[c]**	**IARC[d]**	**RoC[e]**
Ethanol	64-17-5					1	
Naphthalene	91-20-3			"Data are inadequate to assess human carcinogenic potential"		2B	RAHC

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): An upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

In addition to evaluating chemicals that are frequently used in hydraulic fracturing fluids, we also evaluated the availability of toxicological data for subsets of chemicals that are used less frequently on a national basis (Figure 9-4). For this analysis, we binned the chemicals according to frequency of use as identified from the EPA FracFocus 1.0 project database (>10% of disclosures, 5-10% of disclosures, 1-5% of disclosures, <1% of disclosures, or unknown frequency of use), and evaluated the percentage of chemicals within each bin that have available chronic oral RfVs or OSFs, TOPKAT LOAEL estimates, and relevant data on ACToR. This analysis demonstrates that the availability of chronic oral RfVs and OSFs is low across all of these subsets of chemicals. Proportionately, the availability of chronic oral RfVs, OSFs, and data on ACToR is slightly higher for chemicals that are used in >10% of disclosures, compared to chemicals that are used less frequently.

Of the chemicals on the EPA's list that had frequency of use data available from the EPA FracFocus 1.0 project database, the majority were used in <1% of disclosures (n=480), suggesting that potential exposure to these chemicals is more likely to be a local issue rather than a national issue. Given that the analysis of the EPA FracFocus 1.0 project database presented in this chapter was based on 35,957 disclosures, a chemical used in <1% of wells nationally could still be used in several hundred wells. Chemicals used infrequently on a national basis could still be used more frequently within certain areas or counties, increasing the potential for local exposure to that chemical.

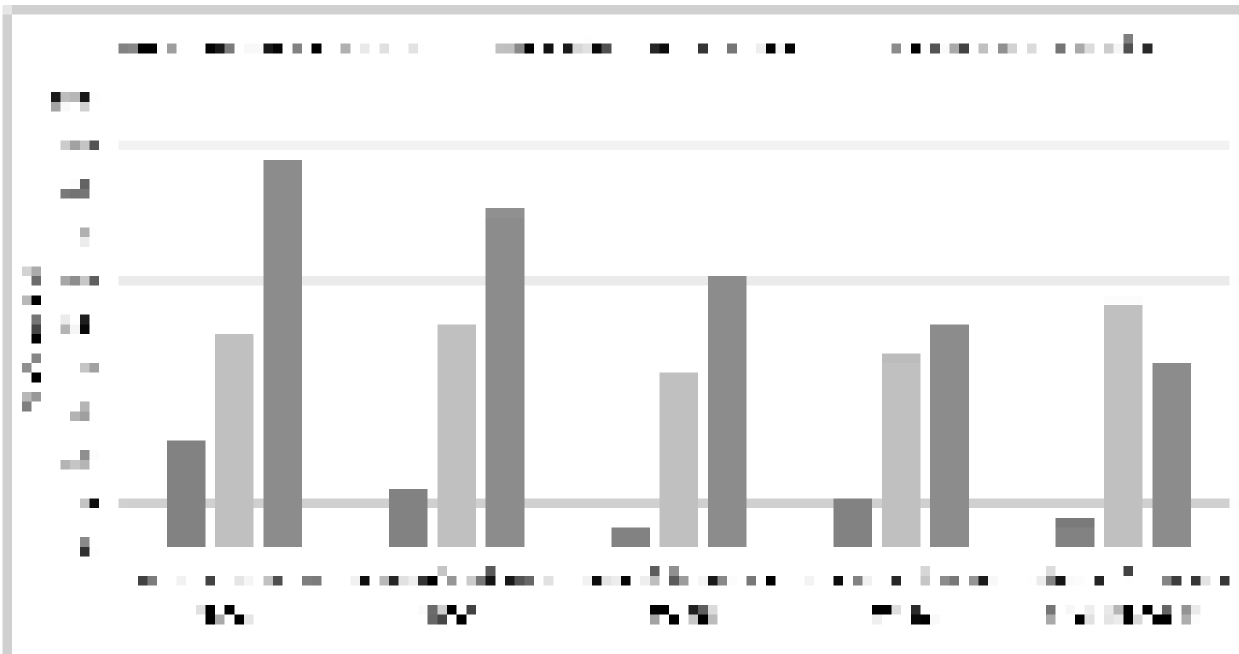

Figure 9-4. Availability of toxicity data (chronic oral RfVs/OSFs, TOPKAT LOAEL estimates, and relevant data on ACToR) for subsets of chemicals used at various frequencies in hydraulic fracturing fluids, as determined based on the number of disclosures in the EPA FracFocus 1.0 project database.

As described in Chapter 5, many of the chemicals used in hydraulic fracturing fluids can be classified as chemical mixtures. Among the most common chemical mixtures on EPA's list of chemicals are petroleum distillates (i.e., hydrocarbon solvents), which are complex mixtures of petroleum hydrocarbons.[1] Two of the most frequently used chemicals in Table 9-2 are petroleum distillates. (Petroleum) hydrotreated light distillates is a mixture of hydrocarbons having carbon numbers predominantly in the range of C9 through C16, and was reported as used in 67% of disclosures in the EPA FracFocus 1.0 project database. Heavy aromatic (petroleum) solvent naphtha is a mixture consisting predominantly of aromatic hydrocarbons in carbon fraction range of C9 through C16, and was reported as used in 21% of disclosures in the EPA FracFocus 1.0 project database. These petroleum distillates lack chronic oral RfVs or OSFs, and have little information available in ACToR. However, a methodology that describes the toxicity and derivation of surrogate

[1] Total petroleum hydrocarbons (TPH) is a term used to describe a large family of several hundred chemical compounds that originally come from crude oil. TPH is a mixture of chemicals, but they are all made mainly from hydrogen and carbon, called hydrocarbons. TPH are divided into groups of petroleum hydrocarbons that act alike in soil or water. These groups are called petroleum hydrocarbon fractions. Each hydrocarbon fraction contains many individual chemicals. Some chemicals that may be found in TPH are hexane, jet fuels, mineral oils, benzene, toluene, xylenes, naphthalene, and fluorene, as well as other petroleum products and gasoline components. Source: ATSDR (2011).

toxicity values for such mixtures was developed by the Total Petroleum Hydrocarbon Criteria Working Group (TPHCWG) (Edwards et al., 1997). This indicator/surrogate approach uses a combination of toxicity data and existing RfVs on individual compounds and fraction-specific mixtures. Examples of compounds present in each fraction include: toluene, ethylbenzene, and styrene (C5-C8) and isopropylbenzene (cumene), naphthalene, fluorene, pyrene, and methylnaphthalene (C9-C16). No data was available for consideration for C>16. Applying their methodology, the TPHCWG derived surrogate aliphatic and aromatic oral toxicity values for fractions in the C5-C8, C9-C16, and C17-C35 ranges. For aromatics, the toxicity ranking was C9-C16 and C17-C35 > C5-C8; and for aliphatics, the toxicity ranking was C9-C16 > C17-C35 > C5-C8. As reviewed by the TPHCWG, compounds above C20 are likely not volatile or soluble in groundwater and will remain at the release site and compounds above C35 are typically not likely to be bioavailable by the oral route of exposure. These surrogate toxicity values are not included in EPA's analysis in this report, but this methodology might be useful for risk assessors at sites where these petroleum distillates are used.

We additionally note that several of the frequently used chemicals in Table 9-2 are designated as being "generally recognized as safe" (GRAS) for use in food additives or food contact substances by the U.S. Food and Drug Administration (FDA). This includes hydrochloric acid, guar gum, sodium hydroxide, sodium chloride, potassium hydroxide, acetic acid, citric acid, choline chloride, carbonic acid dipotassium salt, ammonium chloride, and formic acid. Overall, 103 chemicals on EPA's list of chemicals used in hydraulic fracturing fluids have GRAS designations by the FDA. GRAS chemicals may be used by hydraulic fracturing industry operators in an effort to avoid more hazardous chemicals and minimizes concern in the public perception (Loveless et al., 2011). However, GRAS determinations are often specific to certain conditions as expressed in the FDA GRAS Notification Database and therefore do not indicate that the same chemical is safe for use in hydraulic fracturing fluids. For instance, formic acid is considered GRAS for specific use in paper food packaging materials (U.S. FDA, 2016), but has a chronic oral RfD of 0.9 mg/kg-day based on reproductive effects (U.S. EPA, 2010b). For human health risk assessment in areas of hydraulic fracturing activity, hazard and dose-response relations for these chemicals need to be assessed in the context of the use and levels that are likely to be encountered in an appropriate exposure scenario.

9.5.2 Organic Chemicals in Produced Water

Chapter 7 discussed the detection of volatile and semi-volatile organic chemicals in produced water. Many of these chemicals, including the BTEX chemicals and related hydrocarbons, occur naturally in hydrocarbon formations and are characteristic of produced water from oil and gas production wells in both conventional and unconventional reservoirs. Some of these chemicals have anthropogenic origins, such as di(2-ethylhexyl) phthalate, which does not occur naturally but has known use in hydraulic fracturing fluids. Naphthalene is an example of a chemical that may occur naturally in hydrocarbon formations but is also used frequently in hydraulic fracturing fluids (19% of disclosures in the EPA FracFocus 1.0 project database; Table 9-2). These chemicals have the potential to enter drinking water resources through events such as spills of produced water, mechanical integrity failures, infiltration into groundwater from produced water storage pits, and persistence in inadequately treated wastewater.

Several recent field studies have reported these organic constituents in surface water and groundwater in areas of hydraulic fracturing activity. For example, the BTEX chemicals, diesel-range organics, gasoline-range organics, and naphthalene were detected in groundwater monitoring wells in Pavillion Field, Wyoming, likely as a result of legacy contamination from leaking unlined production fluid storage pits (Digiulio and Jackson, 2016). BTEX chemicals were also found to be elevated above their respective MCLs following spills by the oil and gas industry in Colorado, and were reduced to lower concentrations following remediation (Gross et al., 2013). Ferrar et al. (2013) reported mean concentrations of the BTEX chemicals in effluent from a centralized waste treatment (CWT) facility in Pennsylvania ranged from about 2 to 46 μg/L, with significantly lower concentrations observed after oil and gas well operators were asked to stop discharging waste at this facility (Text Box 8-1). In a survey of 500 private and public water supply wells overlying and adjacent to the Barnett Shale in Texas, Hildenbrand et al. (2015) reported that benzene concentrations exceeded their MCL in all 34 wells where benzene was detected, while toluene, ethylbenzene, and xylenes were prevalent at trace levels; the authors noted that BTEX detections occurred at a high rate in an area that houses a large number of underground injection wells for drilling waste disposal, but it was not clear that these chemicals originated from hydraulic fracturing activity or from another potential source.

As there were a large number of organic chemicals identified on EPA's list, this section focuses on the toxicological evaluation of those organic chemicals that had measured concentration data available in Appendix E and had at least some toxicity data available from the sources in Table 9-1, TOPKAT, or ACToR (69 chemicals total).[1] There were an additional 46 organic chemicals that had measured concentration data in Chapter 7 or Appendix E that did not have any toxicity data available. Organic chemicals that lacked concentration data and are not discussed here.

For this subset of 69 organic chemicals, noncancer toxicity values (chronic oral RfVs and high confidence TOPKAT LOAEL estimates) and ACToR data availability are shown in Table 9-4, and cancer information (OSFs and qualitative cancer classifications) are shown in Table 9-5. Chronic oral RfVs were available for 31 of these chemicals, and ranged from 0.001 mg/kg-day (pyridine) to 0.9 mg/kg-day (acetone). Critical effects for these chemicals include kidney/renal toxicity, hepatotoxicity, neurotoxicity, reproductive toxicity (decreased maternal weight gain), developmental toxicity (decreased offspring body weight, fetal toxicity), and decreased terminal body weight. Six of the chemicals in Table 9-4 are regulated as drinking water contaminants under the NPDWRs: the BTEX chemicals (benzene, ethylbenzene, toluene, xylenes), benzo(a)pyrene, and di(2-ethylhexyl) phthalate.

Of the 38 chemicals in Table 9-4 that lack chronic oral RfVs, 10 have high-confidence TOPKAT LOAEL estimates available. Several of these had similarly low LOAEL estimates: benzo(g,h,i)perylene, indeno(1,2,3-cd)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, dibenz(a,h)anthracene, and N-nitrosodiphenylamine. Notably, 33 of the chemicals

[1] Note that chemical names presented in this chapter and in Appendix H sometimes differ from the chemical names presented with the concentration data in Appendix E. This is because Appendix E uses the chemical names provided by the original sources of chemical data, while this chapter and Appendix H use chemical names that were verified by EPA during the curation of the chemical list. See Appendix H for details on the curation of the chemical list.

in Table 9-4 were added to EPA's chemical list after the release of the external review draft (Text Box 9-2), and therefore were not included in the QSAR analysis (Section 9.4.2).

Table 9-4. List of a subset of organic chemicals that have been detected in produced water, with respective chronic oral RfVs, TOPKAT LOAEL estimates, and availability of ACToR data shown when available.

Includes organic chemicals that were identified on the EPA's list of chemicals in produced water (Appendix H) that have measured concentration data available in Appendix E and have at least some toxicity data available from the sources consulted by the EPA. Chemicals are ordered in the table from most toxic to least toxic based on chronic oral RfV. Chemicals without RfVs were ordered based on TOPKAT LOAEL, and then by number of data points on ACToR. *Indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

		Chronic oral RfV[a]			QSAR estimate	ACToR
Chemical Name	**CASRN**	**RfV (mg/kg-day)**	**Source of RfV**	**Critical Effect[b]**	**TOPKAT LOAEL[c] (mg/day)**	**# of data points[d]**
Pyridine	110-86-1	0.001	IRIS	Increased liver weight	69.5	114
Benzidine	92-87-5	0.003	IRIS	Brain cell vacuolization; liver cell alterations in females		127
2,4-Dichlorophenol	120-83-2	0.003	IRIS	Decreased delayed hypersensitivity response		122
Benzene*	71-43-2	0.004	IRIS	Decreased lymphocyte count	77.6	238
2-Methylnaphthalene	91-57-6	0.004	IRIS	Pulmonary alveolar proteinosis	103	52
1,3,5-Trimethylbenzene	108-67-8	0.01	IRIS	Decreased pain sensitivity	63	76
1,2,4-Trimethylbenzene	95-63-6	0.01	IRIS	Decreased pain sensitivity	91.5	71
Chloroform	67-66-3	0.01	IRIS	Moderate/marked fatty cyst formation in the liver and elevated serum glutamic pyruvic transaminase (SGPT)	47.1	221
Naphthalene	91-20-3	0.02	IRIS	Decreased mean terminal body weight in males	67.5	157
2,4-Dimethylphenol	105-67-9	0.02	IRIS	Clinical signs (lethargy, prostration, and ataxia) and hematological changes	112	88
Di(2-ethylhexyl) phthalate*	117-81-7	0.02	IRIS	Increased relative liver weight	4040	229

		Chronic oral RfV[a]			QSAR estimate	ACToR
Chemical Name	**CASRN**	**RfV (mg/kg-day)**	**Source of RfV**	**Critical Effect[b]**	**TOPKAT LOAEL[c] (mg/day)**	**# of data points[d]**
Pyrene	129-00-0	0.03	IRIS	Kidney effects (renal tubular pathology, decreased kidney weights)	36.1	129
1,4-Dioxane	123-91-1	0.03	IRIS	Liver and kidney toxicity	207	148
Fluoranthene	206-44-0	0.04	IRIS	Nephropathy, increased liver weights, hematological alterations, and clinical effects	44.6	103
Fluorene	86-73-7	0.04	IRIS	Decreased RBC, packed cell volume and hemoglobin	95.1	120
m-Cresol	108-39-4	0.05	IRIS	Decreased body weights and neurotoxicity	123	103
o-Cresol	95-48-7	0.05	IRIS	Decreased body weights and neurotoxicity	229	94
Toluene*	108-88-3	0.08	IRIS	Increased kidney weight	163	188
Diphenylamine	122-39-4	0.1	HHBP	Alterations in clinical chemistry; increased kidney. liver, and spleen weights	30.8	86
Carbon disulfide	75-15-0	0.1	IRIS	Fetal toxicity/ malformations	126	89
Benzyl alcohol	100-51-6	0.1	PPRTV	Effects on survival, growth, and tissue histopathology	210	45
Ethylbenzene*	100-41-4	0.1	IRIS	Liver and kidney toxicity	226	207
Cumene	98-82-8	0.1	IRIS	Increased average kidney weight in female rats	246	101
Acetophenone	98-86-2	0.1	IRIS	General toxicity	274	58
Dibutyl phthalate	84-74-2	0.1	IRIS	Increased mortality	2090	143
Xylenes*	1330-20-7	0.2	IRIS	Decreased body weight, increased mortality	110	174
Benzyl butyl phthalate	85-68-7	0.2	IRIS	Significantly increased liver-to-body weight and liver-to-brain weight ratios		194
Phenol	108-95-2	0.3	IRIS	Decreased maternal weight gain	134	170

Chemical Name	CASRN	Chronic oral RfV[a]			QSAR estimate	ACToR
		RfV (mg/kg-day)	Source of RfV	Critical Effect[b]	TOPKAT LOAEL[c] (mg/day)	# of data points[d]
Biphenyl	92-52-4	0.5	IRIS	Renal papillary mineralization in male F344 rats		103
Caprolactam	105-60-2	0.5	IRIS	Reduced offspring body weight		39
Acetone	67-64-1	0.9	IRIS	Nephropathy	119	79
Benzo(g,h,i)perylene	191-24-2				29.1	68
Indeno(1,2,3-cd)pyrene	193-39-5				38.6	111
Dibenz(a,h)anthracene	53-70-3				38.9	96
Benzo(b)fluoranthene	205-99-2				39	121
Benzo(k)fluoranthene	207-08-9				39	118
N-Nitrosodiphenylamine	86-30-6				39.4	99
Benzo(a)pyrene*	50-32-8				43	184
Phenanthrene	85-01-8				61.3	69
p-Cresol	106-44-5				95.5	98
Dioctyl phthalate	117-84-0				4740	61
Caffeine	58-08-2					134
Benz(a)anthracene	56-55-3					122
Chrysene	218-01-9					114
2-Mercaptobenzothiazole	149-30-4					95
1,2-Diphenylhydrazine	122-66-7					83
Dimethyl phthalate	131-11-3					79
N-Nitroso-N-methylethylamine	10595-95-6					42
4-(1,1,3,3-Tetramethylbutyl)phenol	140-66-9					30
p-Tert-butylphenol	98-54-4					27
2,6-Di-tert-butylphenol	128-39-2					22
Dimethylphenol	1300-71-6					17
2-Ethylhexyl diphenyl phosphate (Octicizer)	1241-94-7					14
2,5-Cyclohexadiene-1,4-dione	106-51-4					12
Cholesterol	57-88-5					11

		Chronic oral RfV[a]			QSAR estimate	ACToR
Chemical Name	**CASRN**	**RfV (mg/kg-day)**	**Source of RfV**	**Critical Effect[b]**	**TOPKAT LOAEL[c] (mg/day)**	**# of data points[d]**
Benzothiazole	95-16-9					10
Octadecanoic acid	57-11-4					9
Butanoic acid, butyl ester	109-21-7					9
Tetradecanoic acid	544-63-8					7
Triphenyl phosphate	115-86-6					7
Dodecanoic acid	143-07-7					6
Drometrizole	2440-22-4					6
3-(4-Methoxyphenyl)-2-ethylhexylester-2-propenoic acid	5466-77-3					6
2,6-Bis(dimethylethyl)-2,5-cyclohexadiene-1,4-dione	719-22-2					3
Diphenylmethane	101-81-5					3
Isopropyl myristate	110-27-0					2
2-[2-[4-(1,1,3,3-tetramethylbutyl)phen-oxy]ethoxy]-ethanol	2315-61-9					2
Sterane	50-24-8					1
3-(4-Methoxyphenyl)-2-propenoic acid	830-09-1					1

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Value; HHBP = Human Health Benchmarks for Pesticides; QSAR = Quantitative structure-activity relationship; TOPKAT = Toxicity Prediction by Komputer Assisted Technology; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] Reference value (RfV): An estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[b] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[c] TOPKAT LOAEL: The LOAEL is the lowest exposure level at which there are biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group. TOPKAT LOAELs were predicted using a QSAR-based software model, as described in Section 9.4.2. Values are rounded to 3 significant figures.

[d] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

Of the organic chemicals in produced water listed in Table 9-4, 17 have available OSFs and 23 are classified as known, probable, or possible carcinogens (Table 9-5). Benzidine and benzene were both classified as human carcinogens by IRIS, IARC, and RoC, with benzidine being the most potent carcinogen listed in Table 9-5 (OSF of 230 per mg/kg-day). Benzo(a)pyrine is classified as a human carcinogen by IARC, and as a probable human carcinogen by IRIS. The remaining chemicals were classified as probable or possible human carcinogens.

Table 9-5. List of OSFs and qualitative cancer classifications available for a subset of organic chemicals that have been reported in produced water.

Includes organic chemicals that were identified on EPA's list of chemicals in produced water (Appendix H) that have measured concentration data available in Chapter 7 or Appendix E (Table 9-4) and are classified as known, probable, or possible carcinogens. Chemicals that had OSFs available are ordered in this table from most potent (highest OSF) to least potent (lowest OSF).

		OSFs[a]		Qualitative Cancer Classifications			
Chemical Name	**CASRN**	**OSF (per mg/kg-day)**	**Source of OSF**	**IRIS [b]**	**PPRTV [c]**	**IARC [d]**	**RoC [e]**
Benzidine	92-87-5	230	IRIS	A (Human carcinogen)		1	Known
N-Nitroso-N-methylethylamine	10595-95-6	22	IRIS	B2 (Probable human carcinogen)		2B	
Benzo(a)pyrene	50-32-8	7.3	IRIS	B2 (Probable human carcinogen)		1	RAHC
Dibenz(a,h)anthra-cene	53-70-3	4.1	CalEPA			2A	RAHC
Indeno(1,2,3-cd)pyrene	193-39-5	1.2	CalEPA			2B	RAHC
Benzo(b)fluoran-thene	205-99-2	1.2	CalEPA			2B	RAHC
Benzo(k)fluoranthene	207-08-9	1.2	CalEPA			2B	RAHC
1,2-Diphenylhydrazine	122-66-7	0.8	IRIS	B2 (Probable human carcinogen)			RAHC
Benz(a)anthracene	56-55-3	0.7	PPRTV	B2 (Probable human carcinogen)		2B	RAHC
Chrysene	218-01-9	0.12	CalEPA	B2 (Probable human carcinogen)		2B	
1,4-Dioxane	123-91-1	0.1	IRIS	"Likely to be carcinogenic to humans"		2B	RAHC

		OSFs[a]		Qualitative Cancer Classifications			
Chemical Name	**CASRN**	**OSF (per mg/kg-day)**	**Source of OSF**	**IRIS [b]**	**PPRTV [c]**	**IARC [d]**	**RoC [e]**
Benzene	71-43-2	0.015-0.055	IRIS	A (Human carcinogen)		1	Known
Chloroform	67-66-3	0.019	CalEPA	B2 (Probable human carcinogen)		2B	RAHC
Di(2-ethylhexyl) phthalate	117-81-7	0.014	IRIS	B2 (Probable human carcinogen)		2B	RAHC
Ethylbenzene	100-41-4	0.011	CalEPA	D (Not classifiable as to human carcinogenicity)		2B	
Biphenyl	92-52-4	0.008	IRIS	"Suggestive evidence of carcinogenic potential"			
N-Nitrosodiphenyl-amine	86-30-6	0.0049	IRIS	B2 (Probable human carcinogen)		3	
Naphthalene	91-20-3			“Data are inadequate to assess human carcinogenic potential”		2B	RAHC
Cumene	98-82-8			D (Not classifiable as to human carcinogenicity)		2B	RAHC
2-Mercaptobenzo-thiazole	149-30-4					2A	
m-Cresol	108-39-4			C (Possible human carcinogen)			
o-Cresol	95-48-7			C (Possible human carcinogen)	“Data are inadequate for the assessment of human carcinogenic potential”		

Chemical Name	CASRN	OSFs[a]		Qualitative Cancer Classifications			
		OSF (per mg/kg-day)	Source of OSF	IRIS [b]	PPRTV [c]	IARC [d]	RoC [e]
Benzyl butyl phthalate	85-68-7			C (Possible human carcinogen)		3	

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; HHBP = Human Health Benchmarks for Pesticides; CalEPA = California Environmental Protection Agency; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): An upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

9.5.3 Inorganic Chemicals and TENORM in Produced Water

Chapter 7 discussed the detection of inorganic constituents such as metals, inorganic ions, and TENORM in produced water. Examples include barium, cadmium, chromium, copper, lead, manganese, nickel, zinc, and radium. In general, these chemicals are naturally occurring, and are characteristic of produced water from both conventional and unconventional reservoirs. These chemicals have the potential to enter drinking water resources through events such as spills of produced water, mechanical integrity failures, infiltration into groundwater from produced water storage pits, and persistence in inadequately treated wastewater.

The entry of inorganic constituents of produced water into drinking water resources has been documented in numerous studies. In Pennsylvania, elevated levels of barium and strontium have been observed in CWT effluent (PA DEP, 2015a), with effluent concentrations dropping after oil and gas well operators were asked to stop discharging waste at this facility (see Text Box 8-1 for details on temporal trends in wastewater management in Pennsylvania). Likewise, effluent concentrations at two publicly owned treatment words (POTWs) that had accepted Marcellus wastewater were found to have lower concentrations of bromide, chloride, barium, strontium, and sulfate after oil and gas well operators were asked to stop discharging waste at this facility in May 2011 (Ferrar et al., 2013). Effluents from POTWs and CWTs that handle Marcellus Shale wastewater have been found to have levels of radium-226 and radium-228 that exceed the MCL for radium and are significantly higher than typical background levels of radium in river water (PA DEP, 2015b). Radium-226 and radium-228 have been demonstrated to accumulate in sediments near the outfalls of CWTs and of POTWs that handle oil and gas wastewater from CWTs (PA DEP, 2015b; Warner et al., 2013a), and in sediments receiving effluent from landfills that accept oil and gas wastes (PA DEP, 2015b). In West Virginia, water samples collected downstream of a hydraulic fracturing wastewater injection facility had elevated specific conductance and total dissolved solids, elevated bromide, chloride, sodium, barium, strontium, and lithium concentrations, and different strontium isotope ratios compared to those found in upstream, background waters (Akob et al., 2016). In a survey of 500 groundwater wells overlying and adjacent to the Barnett Shale in Texas, Hildenbrand et al. (2015) reported a variety of metals and anions that are known produced water constituents at concentrations that sometimes exceeded primary or secondary MCLs, health advisory levels, or other suggested levels as provided in the EPA Drinking Water Standards, although it was not clear that these chemicals originated from nearby hydraulic fracturing activity or from other potential sources.

For the inorganic chemicals that were identified in produced water on EPA's chemical list, noncancer toxicity values (chronic oral RfVs) and ACToR data availability for these chemicals are shown in Table 9-6, and cancer information (OSFs and qualitative cancer classifications) are shown in Table 9-7. As shown in Table 9-6, chronic oral RfVs were available for 26 of these chemicals, ranging from 0.00002 mg/kg-day (phosphorus) to 1.6 mg/kg-day (nitrate). Critical effects for these metals include neurotoxicity, developmental and liver toxicity, hyperpigmentation and keratosis of the skin, and decrements in blood copper status and enzyme activity. Nineteen of the inorganic chemicals in Table 9-6 are regulated as drinking water contaminants under the NPDWR.

All but one of these inorganic chemicals had at least some relevant data available on EPA's ACToR database. However, none of the inorganic chemicals have TOPKAT LOAEL estimates available, as this QSAR model is only able to generate estimates for organic chemicals (Section 9.4.2).

Table 9-6. List of inorganics and TENORM reported in produced water, and respective chronic oral RfVs and OSFs when available.

Includes inorganic chemicals that were identified on EPA's list of chemicals in produced water (Appendix H). Chemicals are ordered from most toxic to least toxic based on chronic oral RfV. Chemicals without chronic oral RfVs were ordered in terms of the number of data points on ACToR. *Indicates chemicals are regulated as drinking water contaminants under the NPDWR.

		Chronic oral RfVs[a]			ACToR
Chemical Name	**CASRN**	**RfV (mg/kg-day)**	**Source of RfV**	**Critical effect[b]**	**# of data points[c]**
Phosphorus	7723-14-0	0.00002	IRIS	Parturition mortality; forelimb hair loss	113
Vanadium	7440-62-2	0.00007	PPRTV	Kidney histopathology	76
Arsenic*	7440-38-2	0.0003	IRIS	Hyperpigmentation and vascular complications	243

Chemical Name	CASRN	Chronic oral RfVs[a]			ACToR
		RfV (mg/kg-day)	Source of RfV	Critical effect[b]	# of data points[c]
Cobalt	7440-48-4	0.0003	PPRTV	Decreased iodine uptake	76
Antimony*	7440-36-0	0.0004	IRIS	Hematological; alterations in glucose and cholesterol	163
Cadmium*	7440-43-9	0.0005	IRIS	Proteinuria	230
Beryllium*	7440-41-7	0.002	IRIS	Intestinal lesions	186
Mercury	7439-97-6	0.002	CICAD	Renal toxicity	177
Lithium	7439-93-2	0.002	PPRTV	Adverse effects in multiple organ systems	43
Chromium (VI)	18540-29-9	0.003	IRIS	None reported	120
Selenium*	7782-49-2	0.005	IRIS	Clinical selenosis	232
Silver	7440-22-4	0.005	IRIS	Argyria	120
Molybdenum	7439-98-7	0.005	IRIS	Increased uric acid levels	73
Iodine	7553-56-2	0.01	CICAD		27
Nitrite*	14797-65-0	0.1	IRIS	Methemoglobinemia	109
Chlorine	7782-50-5	0.1	IRIS	No adverse effect level	116
Manganese	7439-96-5	0.14	IRIS	Central nervous system (CNS) effects	128
Barium*	7440-39-3	0.2	IRIS	Nephropathy	167
Boron	7440-42-8	0.2	IRIS	Decreased fetal weight (developmental)	93
Zinc	7440-66-6	0.3	IRIS	Decreases in erythrocyte Cu, Zn-superoxide dismutase (ESOD) activity in humans	163
Lead*	7439-92-1	0.5 µg/day [d]	CalEPA	Reproductive Toxicity	168
Strontium	7440-24-6	0.6	IRIS	Rachitic bone	67
Iron	7439-89-6	0.7	PPRTV	Adverse gastrointestinal effects	73
Aluminum	7429-90-5	1	PPRTV	Neurotoxicity	88
Chromium (III)	16065-83-1	1.5	IRIS	No effects observed	71

Chemical Name	CASRN	Chronic oral RfVs[a]			ACToR
		RfV (mg/kg-day)	Source of RfV	Critical effect[b]	# of data points[c]
Nitrate*	14797-55-8	1.6	IRIS	Clinical signs of methemoglobinemia in excess of 10%	130
Nickel	7440-02-0				181
Copper*	7440-50-8				163
Thallium*	7440-28-0				136
Chromium	7440-47-3				125
Uranium-238*	7440-61-1				100
Ammonia	7664-41-7				90
Zirconium	7440-67-7				55
Alpha particle*	12587-46-1				55
Fluoride*	16984-48-8				53
Radium*	7440-14-4				52
Beta particle*	12587-47-2				51
Magnesium	7439-95-4				40
Tin	7440-31-5				40
Chloride	16887-00-6				32
Sodium	7440-23-5				31
Sulfate	14808-79-8				27
Potassium	7440-09-7				25
Titanium	7440-32-6				25
Calcium	7440-70-2				24
Radium-226*	13982-63-3				13
Radium-228*	15262-20-1				11
Sulfide	18496-25-8				11
Caesium	7440-46-2				7
Caesium-137	10045-97-3				6
Silicon	7440-21-3				5
Rubidium	7440-17-7				5

Chemical Name	CASRN	Chronic oral RfVs[a]			ACToR
		RfV (mg/kg-day)	Source of RfV	Critical effect[b]	# of data points[c]
Bromide	24959-67-9				2
Sulfite	14265-45-3				1
Uranium-235*	15117-96-1				1
Octasulfur	10544-50-0				

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Value; CalEPA = California Environmental Protection Agency; CICAD = Concise International Chemical Assessment Documents; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] Reference value (RfV): An estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[b] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[c] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

[d] CalEPA MADLs are in units of µg/day, while all other chronic oral RfVs in this table are in units of mg/kg-day.

OSFs were available for 4 of the inorganic chemicals reported in produced water, and 14 are classified as known or probable carcinogens (Table 9-7). OSFs ranged from 15 per mg/kg-day for cadmium to 0.0085 mg/kg-day for lead. Chromium (VI), arsenic, alpha particle, beta particle, radium-226, and radium-288 are all classified as known human carcinogens by all sources reporting in this table. Beryllium and cadmium are both classified as known human carcinogens by IARC and NTP, and as probable human carcinogens by EPA. Lead, cobalt, nickel, nitrate, and nitrite are classified by these sources as possible or probable human carcinogens.

Table 9-7. List of qualitative cancer classifications available for inorganics and NORM that were reported in produced water.

Includes inorganic chemicals that were identified on EPA's list of chemicals in produced water (Appendix H) that classified as known, probable, or possible carcinogens by at least one of the sources in Table 9-1. Chemicals that had OSFs available are ordered in this table from most potent (highest OSF) to least potent (lowest OSF).

Chemical Name	CASRN	OSF[a]		Qualitative Cancer Classifications			
		OSF (per mg/kg-day)	Source of OSF	IRIS[b]	PPRTV[c]	IARC[d]	RoC[e]
Cadmium	7440-43-9	15	CalEPA	B1 (Probable human carcinogen)		1	Known
Arsenic	7440-38-2	1.5	IRIS	A (Human carcinogen)		1	Known

		OSF[a]		Qualitative Cancer Classifications			
Chemical Name	**CASRN**	**OSF (per mg/kg-day)**	**Source of OSF**	**IRIS[b]**	**PPRTV[c]**	**IARC[d]**	**RoC[e]**
Chromium (VI)	18540-29-9	0.5	CalEPA	Inhaled: A (Human carcinogen) Oral: D (Not classifiable as to human carcinogenicity)		1	Known
Lead	7439-92-1	0.0085	CalEPA	B2 (Probable human carcinogen)		2B	RAHC
Alpha particle	12587-46-1					1	
Beryllium	7440-41-7			B1 (Probable human carcinogen)		1	Known
Beta particle	12587-47-2					1	
Radium	7440-14-4					1	
Radium-226	13982-63-3					1	
Radium-228	15262-20-1					1	
Cobalt	7440-48-4				Likely to be carcinogenic to humans	2B	
Nickel	7440-02-0					2B	RAHC
Nitrate	14797-55-8					2A	
Nitrite	14797-65-0					2A	

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; HHBP = Human Health Benchmarks for Pesticides; CalEPA = California Environmental Protection Agency; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): An upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

9.5.4 Organochlorine Pesticides and Polychlorinated Biphenyls (PCBs) in Produced Water

EPA's list of chemicals detected in produced water includes several chemicals that have been banned from commercial use: specifically, organochlorine pesticides and Aroclor 1248, which is a commercial PCB mixture. These chemicals were reported by two of the sources used to compile EPA's chemical list (Appendix H): a technical report prepared by the Gas Technology Institute for the Marcellus Shale Coalition (MSC), which is a drilling industry trade group (Hayes, 2009); and a report by the New York State Department of Environmental Conservation (NYSDEC), which referenced the results of the MSC study (NYSDEC, 2011). These chemicals are listed in Table 9-8 along with their respective noncancer toxicity values (chronic oral RfVs and TOPKAT LOAELs) and availability of relevant toxicological information on ACToR. Cancer information (OSF or qualitative cancer classification) for these chemicals is listed in Table 9-9.

There is uncertainty about why organochlorine pesticides and PCBs were detected, as they are not used in hydraulic fracturing fluids and are not naturally occurring. The MSC study stated the banned substances were detected sporadically and at low concentrations, and suggested they may have originated from laboratory contamination. The NYSDEC report suggested that the banned substances may have been introduced to the shale or the water as a result of drilling or fracturing operations. It is possible that these chemicals were present as legacy contaminants in the source water used for hydraulic fracturing fluid formulation, or were mobilized from the environment near the well. Although these chemicals are notable for their high toxicity, the extent to which these chemicals may be detected in produced water from other hydraulic fracturing sites is not clear.

Chronic oral RfVs for these organochlorine pesticides ranged from 0.000013 mg/kg-day (Heptachlor epoxide) to 0.0005 mg/kg-day (heptachlor), and were all based on liver toxicity. All of these pesticides had TOPKAT LOAEL estimates, and all have relevant data available within EPA's ACToR database.). Heptachlor epoxide, heptachlor, and lindane are regulated as drinking water contaminants under the NPDWR.

Table 9-8. List of organochlorine pesticides and PCBs that were reported in produced water, and their respective chronic oral RfVs, TOPKAT LOAEL estimates, and availability of data in EPA's ACToR database.

Includes banned chemicals that were identified on EPA's list of chemicals in produced water (Appendix H). Chemicals are ordered from most toxic to least toxic based on chronic oral RfV. Chemicals without chronic oral RfVs were ordered in terms of the number of data points on ACToR. *Indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

		Chronic oral RfV[a]			QSAR	ACToR
Chemical Name	**CASRN**	**RfV (mg/kg-day)**	**Source of RfV**	**Critical effect[b]**	**TOPKAT LOAEL[c] (mg/kg)**	**# of data points[d]**
Heptachlor epoxide*	1024-57-3	0.000013	IRIS	Increased liver-to-body weight ratio in both males and females	0.595	168
Aldrin	309-00-2	0.00003	IRIS	Liver toxicity	0.743	166

		Chronic oral RfV[a]			QSAR	ACToR
Chemical Name	CASRN	RfV (mg/kg-day)	Source of RfV	Critical effect[b]	TOPKAT LOAEL[c] (mg/kg)	# of data points[d]
Dieldrin	60-57-1	0.00005	IRIS	Liver lesions	0.442	167
Lindane*	58-89-9	0.0003	IRIS	Liver and kidney toxicity	23.9	238
Heptachlor*	76-44-8	0.0005	IRIS	Liver weight increases in males	0.927	203
beta-Hexachlorocyclohexane	319-85-7				23.9	88
delta-Hexachlorocyclohexane	319-86-8				23.9	22
Aroclor 1248	12672-29-6				21.87	35
p,p'-DDE	72-55-9				14.6	103
Endrin aldehyde	7421-93-4				4.09	27
Endosulfan I	959-98-8				2.27	32
Endosulfan II	33213-65-9				2.27	32

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; QSAR = Quantitative structure-activity relationship; TOPKAT = Toxicity Prediction by Komputer Assisted Technology; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] Reference value (RfV): An estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[b] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[c] TOPKAT lowest-observed-adverse-effect level (LOAEL): The LOAEL is the lowest exposure level at which there are biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group. TOPKAT LOAELs were predicted using a QSAR-based software model, as described in Section 9.4.2.

[d] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

OSFs were available for 7 of the organochlorine pesticides that are classified as known, probable, or possible human carcinogens (Table 9-9). OSFs ranged from 17 per mg/kg-day (aldrin) to 0.34 per mg/kg-day (p,p'-DDE). Aldrin, dieldrin, heptachlor epoxide, heptachlor, beta-hexachlorocyclohexane, and p,p'-DDE are classified as probable or possible carcinogens. Lindane is classified as a known carcinogen by IARC, and as "reasonably anticipated to be a human carcinogen" by RoC.

Table 9-9. List of OSFs and qualitative cancer classifications available for organochlorine pesticides reported in produced water.

Includes banned chemicals that were identified on EPA's list of chemicals in produced water (Appendix H) that are classified as known, probable, or possible carcinogens by at least one of the sources in Table 9-1. Chemicals are ordered in this table from most potent (highest OSF) to least potent (lowest OSF).

		OSF[a]		Qualitative cancer classifications			
Chemical Name	CASRN	OSF (per mg/kg-day)	Source of OSF	IRIS[b]	PPRTV[c]	IARC[d]	RoC[e]
Aldrin	309-00-2	17	IRIS	B2 (Probable human carcinogen)		3	
Dieldrin	60-57-1	16	IRIS	B2 (Probable human carcinogen)		3	
Heptachlor epoxide	1024-57-3	9.1	IRIS	B2 (Probable human carcinogen)			
Heptachlor	76-44-8	4.5	IRIS	B2 (Probable human carcinogen)		2B	
beta-Hexachlorocyclohexane	319-85-7	1.8	IRIS	C (Possible human carcinogen)			
Lindane	58-89-9	1.1	CalEPA			1	RAHC
p,p'-DDE	72-55-9	0.34	IRIS	B2 (Probable human carcinogen)			

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; CalEPA = California Environmental Protection Agency; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): An upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

9.5.5 Methane in Stray Gas

Chapter 6 discussed stray gas as a potential hazard in areas of hydraulic fracturing activity (Text Box 6-3). Stray gas refers to the phenomenon of natural gas (primarily methane, plus lesser amounts of ethane) migrating into shallow groundwater, into water wells, or to the surface (e.g., cellars, streams, or springs). As discussed in Chapter 6, some studies indicate an association between hydraulic fracturing activity and elevated methane concentrations in drinking water, while other studies did not find such a correlation. Potential pathways for migration of stray gas into aquifers include pathways along production wells with casing and/or cement issues, through naturally existing fractures, through induced fractures, or via a route that is some combination of these pathways.

Although ingestion of methane is not considered to be toxic, it has the potential to pose a physical hazard. Methane can accumulate to explosive levels when allowed to exsolve (degas) from groundwater in closed environments. High concentrations of methane may also displace oxygen and act as an asphyxiant (NIOSH, 2000), potentially causing suffocation, loss of consciousness, or symptoms such as headache and nausea. Methane is not a regulated drinking water contaminant. Methane does not have an RfV, OSF, or qualitative cancer classification available from any of the sources consulted by EPA, and did not have a high-confidence TOPKAT LOAEL estimate. Information on methane is available within the ACToR database.

9.5.6 Disinfection Byproducts (DBPs) Formed from Wastewater Constituents

Some of the inorganic constituents of hydraulic fracturing produced water, including chloride, bromine, iodine, and ammonium, can contribute to the formation of DBPs during wastewater treatment (Harkness et al., 2015; Parker et al., 2014). The entry of these constituents into drinking water resources—e.g., as a result of wastewater spills or from the discharge of inadequately treated hydraulic fracturing wastewater—can result in DBPs in finished drinking water from downstream drinking water treatment plants (States et al., 2013). DBPs may also be formed when hydraulic fracturing produced water is treated at a centralized or publicly owned treatment works, and may reach drinking water resources when the treated wastewater is discharged to surface water (Hladik et al., 2014). Currently, there are no data available on the concentrations of DBPs in finished drinking water as related to contributions of DBP precursors from hydraulic fracturing wastewater.

Regulated DBPs such as bromate, chlorite, haloacetic acids, and trihalomethanes are a small subset of the full spectrum of DBPs that include other chlorinated and brominated DBPs as well as nitrogenous and iodated DBPs. Long term exposure to these DBPs can result in an increased risk of cancer, anemia, liver and kidney effects, and central nervous system effects. Some of the unregulated DBPs may be more toxic than their regulated counterparts (Harkness et al., 2015; McGuire et al., 2014; Parker et al., 2014). In addition, brominated forms of DBPs are considered to be more cytotoxic, genotoxic, and carcinogenic than chlorinated species based on studies using rodents, various types of human cells, and a salmonella strain containing human P450 genes (McGuire et al., 2014; Parker et al., 2014; States et al., 2013; Krasner, 2009; Richardson et al., 2007). As with brominated DBPs, there is concern that some iodinated forms of DBPs are more cytotoxic and genotoxic than chlorinated species (McGuire et al., 2014; Parker et al., 2014; Krasner, 2009; Richardson et al., 2007), as evidenced by studies involving rodent research and human cell research (Plewa et al., 2010; Plewa and Wagner, 2009; Richardson et al., 2007). The MCLs (mg/L) for the regulated DBPs are: 0.01 for bromate, 1.0 for chlorite, 0.06 for haloacetic acid, and 0.08 for total trihalomethanes.

9.5.7 Chemicals Detected in Multiple Stages of the Hydraulic Fracturing Water Cycle

As mentioned in Section 9.3 above, there were a total of 77 chemicals on EPA's list that were identified as being used in hydraulic fracturing fluids and detected in produced water. The presence of these chemicals within both of these stages of the hydraulic fracturing water cycle may indicate that these chemicals persist after they are injected into the well. However, this is not necessarily the case, as some of these chemicals (e.g., BTEX, naphthalene, metals) also occur naturally in oil and gas reservoirs. Additionally, the EPA's list of chemicals used in hydraulic fracturing fluids and list of chemicals in produced water were compiled from different sets of sources, and does not provide a matched comparison between the chemicals used in hydraulic fracturing fluid and the chemicals present in produced water at a particular site. There may have been other chemicals in present in produced water that were not detected by these studies due to limitations of analytical chemistry. Thus, the EPA's composited chemical list cannot reliably be used to draw conclusions on the persistence of hydraulic fracturing chemicals following well injection.

Of the 77 chemicals identified in both hydraulic fracturing fluids and produced water, 45 have a chronic oral RfV or OSF available from at least one of the sources in Table 9-1. These 45 chemicals and their respective toxicity values are shown in Table 9-10, with frequency of use data from the EPA FracFocus 1.0 project database provided when available. Eleven of these chemicals are regulated as drinking water contaminants.

Table 9-10. List of 45 chemicals on EPA's list that were used in hydraulic fracturing fluids and detected in produced water and have an RfV or OSF available.

Frequency of use data from the EPA FracFocus 1.0 project database is provided when available. Chemicals with available data from the FracFocus 1.0 project database are ordered from high to low based on frequency of use. Chemicals without frequency of use data are ordered from most toxic to least toxic based on chronic oral RfV. *Indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

			Chronic oral RfVs[b]			OSFs[d]	
Chemical Name	**CASRN**	**% of Disclo-sures[a]**	**RfV (mg/kg-day)**	**Source of RfV**	**Critical Effect[c]**	**OSF (per mg/kg-day)**	**Source of OSF**
Methanol	67-56-1	73%	2	IRIS	Extra cervical ribs		
Ethylene glycol	107-21-1	47%	2	IRIS	Kidney toxicity		
Propargyl alcohol	107-19-7	33%	0.002	IRIS	Renal and hepatotoxicity		
2-Butoxyethanol	111-76-2	23%	0.1	IRIS	Hemosiderin deposition in liver (inhalation study)		
Naphthalene	91-20-3	19%	0.02	IRIS	Decreased mean terminal body weight in males		

Chemical Name	CASRN	% of Disclo-sures[a]	Chronic oral RfVs[b]			OSFs[d]	
			RfV (mg/kg-day)	Source of RfV	Critical Effect[c]	OSF (per mg/kg-day)	Source of OSF
1,2,4-Trimethylbenzene	95-63-6	13%	0.01	IRIS	Decreased pain sensitivity		
Formic acid	64-18-6	11%	0.9	PPRTV	Reproductive Toxicity		
N,N-Dimethylformamide	68-12-2	9%	0.1	PPRTV	Increase in ALT enzyme and liver weight		
Benzyl chloride	100-44-7	6%	0.002	PPRTV	Cardiotoxicity	0.17	IRIS
1,2-Propylene glycol	57-55-6	4%	20	PPRTV	Reduced RBC counts and hyperglycemia		
Xylenes*	1330-20-7	2%	0.2	IRIS	Decreased body weight, increased mortality		
D-Limonene	5989-27-5	2%	0.1	CICAD	Increased liver weight		
1-Butanol	71-36-3	1%	0.1	IRIS	Hypoactivity and ataxia		
Toluene*	108-88-3	0.7%	0.08	IRIS	Increased kidney weight		
Bis(2-chloroethyl) ether	111-44-4	0.7%				1.1	IRIS
2-(2-Butoxyethoxy)ethanol	112-34-5	0.6%	0.03	PPRTV	Changes in red blood cells (RBC)		
1,3,5-Trimethylbenzene	108-67-8	0.5%	0.01	IRIS	Decreased pain sensitivity		
Cumene	98-82-8	0.5%	0.1	IRIS	Increased average kidney weight in female rats		
Iron	7439-89-6	0.4%	0.7	PPRTV	Adverse gastrointestinal effects		
1,2,3-Trimethylbenzene	526-73-8	0.4%	0.01	IRIS	Decreased pain sensitivity		

Chemical Name	CASRN	% of Disclo-sures[a]	Chronic oral RfVs[b]			OSFs[d]	
			RfV (mg/kg-day)	Source of RfV	Critical Effect[c]	OSF (per mg/kg-day)	Source of OSF
Phenol	108-95-2	0.4%	0.3	IRIS	Decreased maternal weight gain		
Ethylbenzene*	100-41-4	0.4%	0.1	IRIS	Liver and kidney toxicity	0.011	CalEPA
1,4-Dioxane	123-91-1	0.3%	0.03	IRIS	Liver and kidney toxicity	0.1	IRIS
Acetone	67-64-1	0.2%	0.9	IRIS	Nephropathy		
Boron	7440-42-8	0.05%	0.2	IRIS	Decreased fetal weight		
o-Xylene*	95-47-6	0.05%	0.2	ATSDR	Neurotoxicity		
Acetophenone	98-86-2	0.04%	0.1	IRIS	General toxicity		
Quinoline	91-22-5	0.02%				3	IRIS
Dichloromethane*	75-09-2	0.02%	0.006	IRIS	Hepatic effects (hepatic vacuolation, liver foci)	0.002	IRIS
Trimethylbenzene	25551-13-7	0.01%	0.01	IRIS	Decreased pain sensitivity		
Benzene*	71-43-2	0.01%	0.004	IRIS	Decreased lymphocyte count	0.015-0.055	IRIS
Bisphenol A	80-05-7	0.01%	0.05	IRIS	Reduced mean body weight		
Aluminum	7429-90-5	0.003%	1	PPRTV	Neurotoxicity		
Hydrazine	302-01-2	0.003%				3	IRIS
Chlorobenzene*	108-90-7	0.003%	0.02	IRIS	Histopathologic changes in liver		
Arsenic*	7440-38-2		0.0003	IRIS	Hyperpigmentation and vascular complications	1.5	IRIS
Acrolein	107-02-8		0.0005	IRIS	Decreased survival		
Chromium (VI)	18540-29-9		0.003	IRIS	None reported	0.5	CalEPA
Tributyl phosphate	126-73-8		0.01	PPRTV	Occasional salivation	0.009	PPRTV

Chemical Name	CASRN	% of Disclo-sures[a]	Chronic oral RfVs[b]			OSFs[d]	
			RfV (mg/kg-day)	Source of RfV	Critical Effect[c]	OSF (per mg/kg-day)	Source of OSF
Di(2-ethylhexyl) phthalate*	117-81-7		0.02	IRIS	Increased relative liver weight	0.014	IRIS
Chlorine	7782-50-5		0.1	IRIS	No adverse effect level		
p-Xylene*	106-42-3		0.2	ATSDR	Neurotoxicity		
Zinc	7440-66-6		0.3	IRIS	Decreases in erythrocyte Cu, Zn-superoxide dismutase (ESOD) activity in humans		
Lead*	7439-92-1		0.5 µg/day [e]	CalEPA	Reproductive toxicity	0.0085	CalEPA
Chromium (III)	16065-83-1		1.5	IRIS			

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Value; HHBP = Human Health Benchmarks for Pesticides; ATSDR = Agency for Toxic Substances and Disease Registry; CalEPA = California Environmental Protection Agency; CICAD = Concise International Chemical Assessment Documents; QSAR = Quantitative structure-activity relationship; TOPKAT = Toxicity Prediction by Komputer Assisted Technology; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] The FracFocus frequency of use data presented in this chapter is based on 35,957 FracFocus disclosures that were deduplicated, within the study time period (January 1, 2011 to February 28, 2013), and with ingredients that have a valid CASRN.

[b] Reference value (RfV): An estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[c] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[d] Oral slope factor (OSF): An upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[e] CalEPA MADLs are in units of µg/day, while all other chronic oral RfVs in this table are in units of mg/kg-day.

9.6 Hazard Evaluation of Selected Subsets of Hydraulic Fracturing Chemicals Using Multi-Criteria Decision Analysis (MCDA): Integrating Toxicity, Occurrence, and Physicochemical Data

Based on the information presented in Section 9.5, it is clear that there are a variety of chemicals used in hydraulic fracturing fluids or detected in produced water that are known to be hazardous to human health. However, there are gaps in our understanding of the potential for human exposure to these chemicals. Although there are subsurface and surface pathways by which these chemicals may be introduced into drinking water resources—including spills, leaks, mechanical integrity failures, intersection of the fracture network with groundwater, or discharge of wastewater, as described in previous chapters of this report—there are significant limitations associated with the publicly available data on these potential impacts, and the potential for human exposure has not been systematically characterized. This makes it difficult to determine which chemicals are of the greatest concern for human exposure in drinking water, and creates a challenge for hazard evaluation.

Although exposure assessment data are limited, some of the chemicals identified by EPA have other data available that might provide preliminary insight into relative hazard potential. This includes data on toxicity, frequency of use in hydraulic fracturing fluids, detected concentrations in produced water, and data on physicochemical properties. By integrating these types of data, we can place the severity of potential impacts (i.e., the toxicity of specific chemicals) into the context of factors that affect the likelihood of impacts (i.e., frequency of use, environmental fate and transport).

Multi-criteria decision analysis (MCDA) is one possible approach that can be used to facilitate data integration. MCDA is a well-established decision support tool, which is used to integrate multiple lines of evidence to develop an overall ranking or classification (Hristozov et al., 2014; Mitchell et al., 2013b; Huang et al., 2011; Linkov et al., 2011). Using MCDA, a problem is approached by dividing it into smaller criteria that need to be evaluated; the criteria are each analyzed individually, and then combined to provide an integrated evaluation. This approach is structured yet flexible, and offers a transparent means of combining information to provide weight of evidence and insight into a complex problem. MCDA has gained increasing popularity as an environmental decision-making tool (Huang et al., 2011). A recent publication by Yost et al. (In Press) described the use of an MCDA framework to evaluate the hazard potential of chemicals associated with hydraulic fracturing.

Here, to demonstrate one possible method for exploring the potential hazards of these chemicals, we use an adaptation of the MCDA framework developed by Yost et al. (In Press) to analyze and rank selected subsets of chemicals that have data available.[1] Chemicals were assigned scores based on toxicity, occurrence, and physicochemical properties that describe transport in water. These scores were then combined to develop a relative ranking of chemicals based on hazard potential.

The MCDA scores provide a preliminary evaluation of hazard potential, and serve as a qualitative metric for making comparison between chemicals when exposure assessment data is limited or unavailable. This analysis is not intended to define whether or not a chemical will present a human health hazard or indicate that one chemical is safer than another, and should not be used in place of

[1] Yost et al. (In Press) used the MCDA framework to analyze and rank the hazards of chemicals used in hydraulic fracturing fluids, using data from the FracFocus 1.0 project database as the metric of occurrence. This chapter uses that same framework for the analysis of chemicals used in hydraulic fracturing fluids. For chemicals detected in produced water, this chapter modifies the MCDA framework by using measured concentration in produced water as the metric of occurrence.

site-specific data on chemical exposures. An overview of the MCDA framework and selection of chemicals for inclusion in the MCDA is described below.

9.6.1 Overview of the MCDA Framework for Hazard Evaluation

The MCDA framework employed in this chapter was designed specifically to fit the scope of EPA's hydraulic fracturing study (Yost et al., In Press). A basic schematic of the model is shown in Figure 9-5, and the methods for assigning scores are outlined below. Under the MCDA framework, each chemical was assigned three scores:

1. A Toxicity Score;
2. An Occurrence Score; and
3. A Physicochemical Properties score.

The three scores were each standardized based on the highest and lowest respective score within the given subset of chemicals, and then summed to develop a Total Hazard Potential Score for each chemical. The Total Hazard Potential Scores reflect a relative ranking of each chemical within the given subset of chemicals, and offer a means of making comparisons between chemicals.

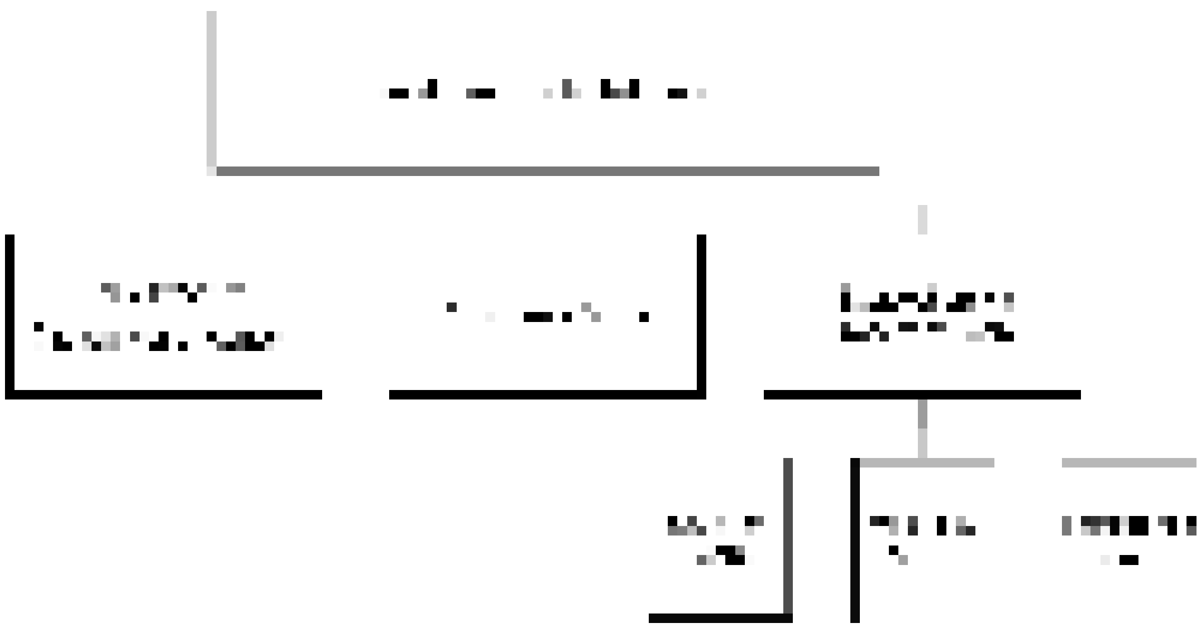

Figure 9-5. Overview of the MCDA framework for hazard evaluation.
Source: Yost et al. (In Press).

9.6.2 Selection of Chemicals for Hazard Evaluation in the MCDA Framework

From the overall list of 1,606 chemicals identified in this assessment, subsets of chemicals were selected for hazard evaluation in the MCDA framework if they had sufficient data for inclusion, using an adaptation of the criteria outlined by Yost et al. (In Press). Specifically, chemicals were selected if they had the following information available:

1. Had a chronic oral RfV or OSF from a US federal source (IRIS, PPRTV, ATSDR, HHBP);
2. Had available data on frequency of use in hydraulic fracturing fluids (data available from the EPA FracFocus 1.0 project database) or measured concentrations in produced water (data available from Appendix E)[1]; and
3. Had data on physicochemical properties available from EPI Suite.

The rationale for applying these criteria is as follows:

1. Federal toxicity values generally undergo more extensive peer review compared to other sources of toxicity values, and are based on the best available scientific information. For this reason, EPA generally prefers to apply RfVs and OSFs from US federal sources for human health risk assessment.
2. Data on frequency of use (in hydraulic fracturing fluids) or measured concentration (in produced water) provide a metric to help assess the likelihood of chemical occurrence in the hydraulic fracturing water cycle.
3. Information on physicochemical properties enables estimation of the likelihood a chemical will be transported in water.

Chemicals used in hydraulic fracturing fluids and chemicals detected in produced water were evaluated separately using the MCDA framework. To explore the different types of toxicity values identified by EPA, two versions of the MCDA were performed on each of these subsets of chemicals: a noncancer MCDA, in which the Toxicity Score is calculated using chronic oral RfVs; and a cancer MCDA, in which the Toxicity Score is calculated using OSFs. For chemicals used in hydraulic fracturing fluids, the noncancer MCDA was repeated for specific subsets of chemicals used in three states that have a significant amount of hydraulic fracturing activity: Texas, Pennsylvania, and North Dakota. Thus, seven iterations of the MCDA were performed: 1-4) noncancer MCDAs for chemicals used in hydraulic fracturing fluids on a national or state-specific basis, 5) a cancer MCDA for chemicals used in hydraulic fracturing fluids, 6) a noncancer MCDA for chemicals detected in produced water, and 7) a cancer MCDA for chemicals detected in produced water.

In total, 42 chemicals used in hydraulic fracturing fluid and 29 chemicals detected in produced water had sufficient information available for inclusion in noncancer MCDAs (Figure 9-6), while 10 chemicals used in hydraulic fracturing fluid and 7 chemicals detected in produced water had sufficient information available for inclusion in cancer MCDAs (Figure 9-7).

[1] Chemicals in produced water were considered for the MCDA if they had average or median measured concentrations from any of the tables in Appendix E. Chemicals with only a maximum or minimum concentration listed in Appendix E were not considered for the MCDA.

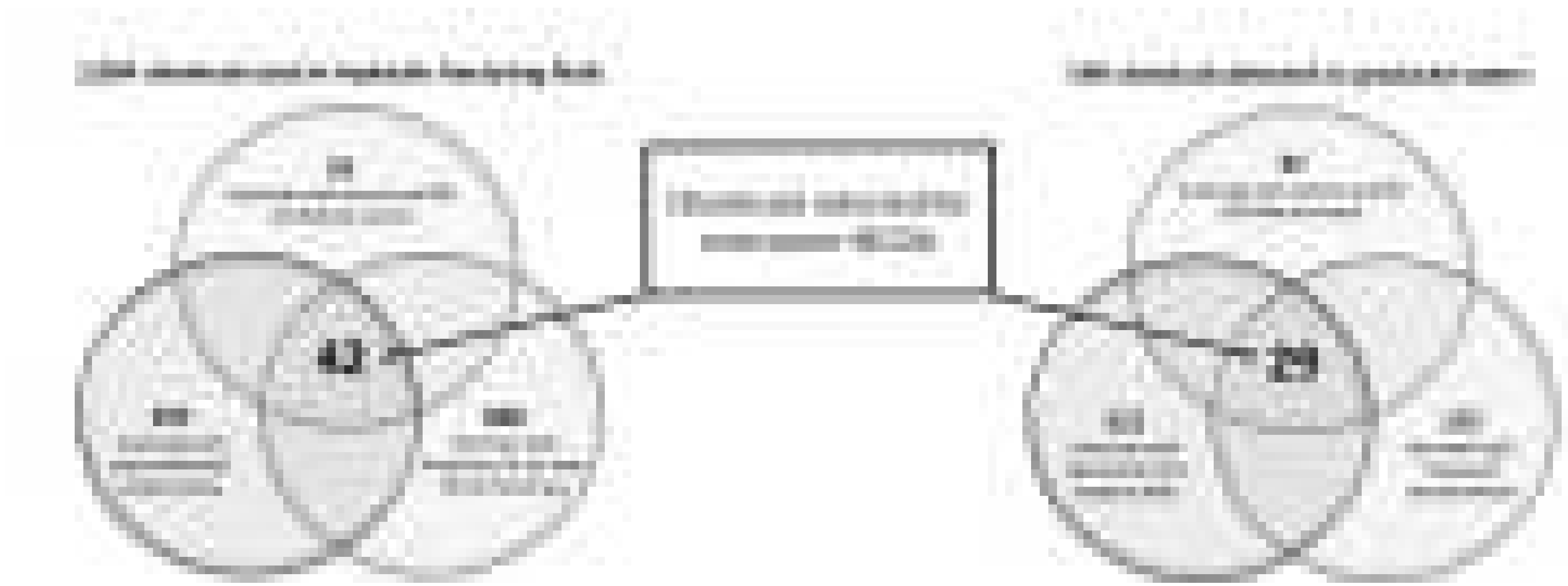

Figure 9-6. The subsets of chemicals selected for hazard evaluation using the noncancer MCDA framework included 42 chemicals used in hydraulic fracturing fluids and 29 chemicals detected in produced water.

For chemicals used in hydraulic fracturing fluids, subsets of these chemicals were also considered in state-specific analyses for Texas (36 chemicals), Pennsylvania (20 chemicals), and North Dakota (21 chemicals).

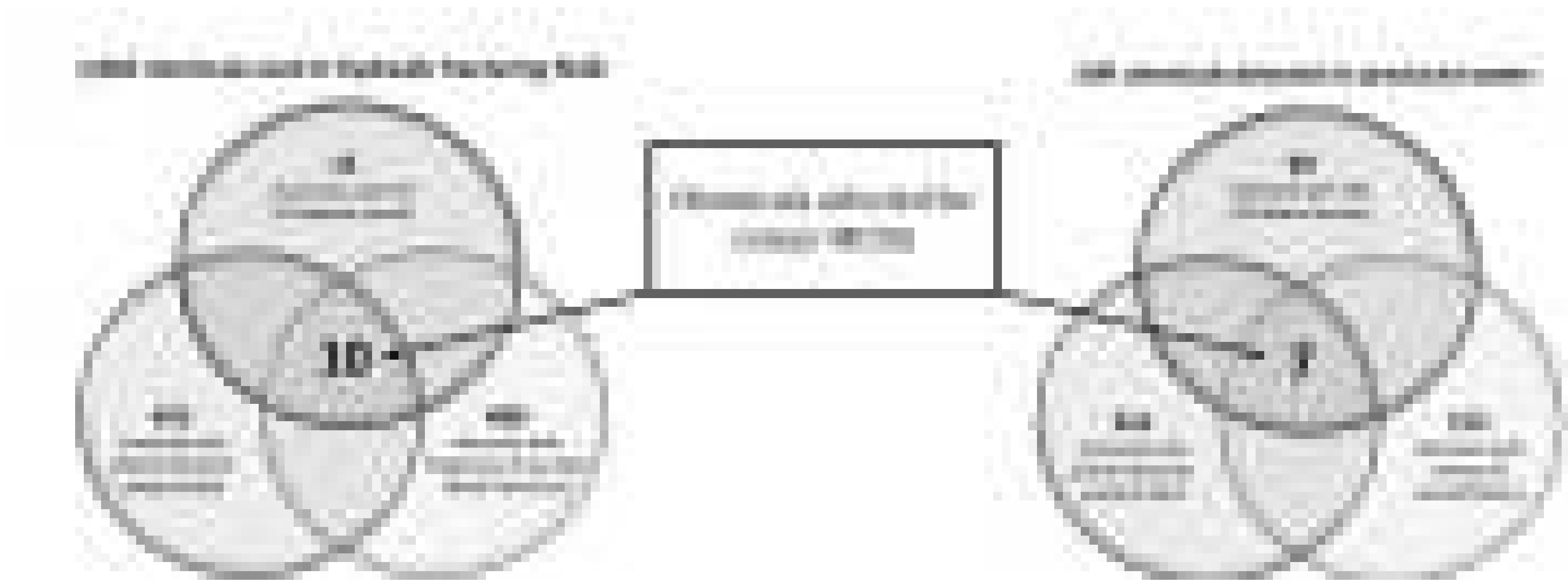

Figure 9-7. The subsets of chemicals selected for hazard evaluation using the cancer MCDA framework included 10 chemicals used in hydraulic fracturing fluids, and 7 chemicals detected in produced water.

9.6.3 Calculation of MCDA Scores

For each iteration of the MCDA, chemicals were assigned scores based on toxicity, occurrence, and physicochemical properties according to the protocol outlined by Yost et al. (In Press). These scores were then standardized to the highest and lowest score within the given subset of chemicals, and then summed to determine a total score and relative ranking for each chemical. The methods used to assign each score and calculate a total score are outlined below.

9.6.3.1 Toxicity Score (Noncancer MCDA)

For each noncancer MCDA, Toxicity Scores were calculated based on chronic oral RfVs from US federal sources (IRIS, PPRTV, ATSDR, and HHBP). If a chemical had a chronic oral RfV available from more than one of these sources, a single value was selected in this order, as described in Section 9.4: HHBP (pesticides), IRIS, PPRTV, ATSDR. Toxicity Scores for the noncancer MCDA were then assigned based on a relative ranking. Within each suite of chemicals considered in this analysis (chemicals used in hydraulic fracturing fluids, or chemicals detected in produced water), RfVs were ranked based on quartiles, and each chemical was assigned a Toxicity Score of 1 to 4 (Table 9-11). Chemicals in the lowest quartile received the highest Toxicity Score, as these chemicals have lower RfVs than other chemicals (i.e., may have lower thresholds for toxicity).

9.6.3.2 Toxicity Score (Cancer MCDA)

For each cancer MCDA, Toxicity Scores were calculated based on OSFs from US federal sources (IRIS, PPRTV, and HHBP). If a chemical had an OSF available from more than one of these sources, a single value was selected in this order, as described in Section 9.4: HHBP (pesticides), IRIS, PPRTV. Toxicity Scores for the cancer MCDA were assigned based on a relative ranking. Within each suite of chemicals considered in this analysis (chemicals used in hydraulic fracturing fluids, or chemicals detected in produced water), OSFs were ranked based on quartiles, and each chemical was assigned a Toxicity Score of 1 to 4 (Table 9-11). Chemicals in the highest quartile received the highest Toxicity Score, as these chemicals have higher OSFs than other chemicals (i.e., are associated with a higher increased risk of cancer per unit of exposure).

9.6.3.3 Occurrence Score

For each of the noncancer and cancer MCDAs, an Occurrence Score was calculated based on the frequency or concentration at which each chemical was reported within the hydraulic fracturing water cycle. For chemicals used in hydraulic fracturing fluids, the Occurrence Score was based on the number of well disclosures for each chemical in the EPA FracFocus 1.0 project database. For chemicals detected in produced water, the Occurrence Score was based on the average or median measured concentration reported in Appendix E. If an average or median concentration of a chemical was reported by multiple studies in Appendix E, the highest of these reported average or median concentrations was used for this calculation. Once a value was determined for each chemical, Occurrence Scores were then assigned based on a relative ranking. Within each suite of chemicals considered in this analysis (chemicals used in hydraulic fracturing fluids, or chemicals detected in produced water), chemical occurrence was ranked based on quartiles, with each chemical assigned an Occurrence Score of 1 to 4 (Table 9-11).

9.6.3.4 Physicochemical Properties Score

For each of the noncancer and cancer MCDAs, a Physicochemical Properties Score was calculated based upon inherent physicochemical properties that describe the likelihood that a chemical will be transported in water. The total Physicochemical Properties Score was calculated as the sum of three subcriteria scores: a Mobility Score, a Volatility Score, and a Persistence Score. The Mobility Score was assessed based upon three physicochemical properties that describe chemical solvency in water: the octanol-water partition coefficient (K_{ow}), the soil adsorption coefficient (K_{oc}), and aqueous solubility. The Volatility Score was assessed based on the Henry's law constant, which describes partitioning of a chemical between water and air. The Persistence Score was assessed based on estimated half-life in water, which describes how long a chemical will remain in water before it is degraded.

For input into the MCDA, experimentally measured physicochemical property values (provided in EPI Suite) were used whenever available. Otherwise, estimated values from EPI Suite were used. To classify these values and assign a score, these numerical values were compared against threshold values (Table 9-11). Each chemical was assigned a Mobility Score, Volatility Score, and Persistence Score (each on a scale of 1 to 4), which were then summed to calculate the Physicochemical Properties Score. The threshold values in Table 9-11 are based upon previously published values employed by existing exposure assessment models, including the EPA's Design for the Environment Alternatives Assessment Criteria for Hazard Evaluation (U.S. EPA, 2011b), the EPA's Pollution Prevention (P2) Framework (U.S. EPA, 2012i), and a peer-reviewed publication by Mitchell et al. (2013b). More details on the Physicochemical Properties Score calculation are provided in the Chapter 9 Annex, Section 9.8.1.

9.6.4 Total Hazard Potential Score

Within each iteration of the MCDA, the three criteria scores (Toxicity, Occurrence, Physicochemical Properties) were each standardized to the dataset by scaling to the highest and lowest respective score within the given subset of chemicals. The following equation was used:

$$S_{x\_final} = (S_x - S_{min}) / (S_{max} - S_{min})$$

in which S_x is the raw score for a particular chemical, S_{max} is the highest observed raw score within the set of chemicals, and S_{min} is the lowest observed raw score within the set of chemicals. S_{x_final} is the standardized score for the chemical. Each standardized score (Toxicity, Occurrence, or Physicochemical Properties) falls on a scale of 0 to 1, and represents a relative ranking within the given subset of chemicals.

The standardized Toxicity Score, Occurrence Score, and Physicochemical Properties Score were summed to calculate a Total Hazard Potential Score for each chemical. The Total Hazard Potential Scores fall on a scale of 0 to 3, with higher scores indicating chemicals that may be more likely to affect drinking water resources. Examples of the Total Hazard Potential Score calculation can be found in the Chapter 9 Annex, Section 9.8.2.

Table 9-11. Thresholds used for developing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score in this MCDA framework.
Adapted from Yost et al. (In Press).

Criteria	Sub-criteria	Value	Score			
			1	2	3	4
Toxicity (Noncancer MCDA)	NA	Chronic oral RfV (mg/kg-day)	>3rd quartile	>2nd quartile to ≤3rd quartile	>1st quartile to ≤2nd quartile	≤1st quartile
Toxicity (Cancer MCDA)	NA	OSF (per mg/kg-day)	<1st quartile	≥1st quartile to <2nd quartile	≥2nd quartile to <3rd quartile	≥3rd quartile
Occurrence	NA	Frequency of use (% of disclosures in EPA's FracFocus 1.0 project database) <u>or</u> Measured concentration in produced water (µg/L; Appendix E)	<1st quartile	≥1st quartile to <2nd quartile	≥2nd quartile to <3rd quartile	≥3rd quartile
Physico-chemical Properties	Mobility	Log K_{OW}	>5	>3 to ≤5	>2 to ≤3	≤2
		Log K_{OC}	>4.4	>3.4 to ≤4.4	>2.4 to ≤3.4	≤2.4
		Aqueous solubility (mg/L)	<0.1	≥0.1 to <100	≥100 to <1000	≥1000
	Volatility	Henry's law constant	$>10^{-1}$	$>10^{-3}$ to $\leq 10^{-1}$	$>10^{-5}$ to $\leq 10^{-3}$	$\leq 10^{-5}$
	Persistence	Half-life in water (days)	<16	≥16 to <60	≥60 to <180	≥180

9.6.5 MCDA Results

For each iteration of the MCDA, we first present the data used for input into the MCDA, including data on toxicity, occurrence, and physicochemical properties. We then present the results of each MCDA, which show a relative ranking of chemicals based on integration of these data. Lastly, we discuss the key limitations of this MCDA approach, which is intended as a preliminary analysis only.

9.6.5.1 Results: Noncancer MCDA for Chemicals Used in Hydraulic Fracturing Fluids

A total of 42 chemicals used in hydraulic fracturing fluids were evaluated in a noncancer MCDA (Table 9-12). Chronic oral RfVs within this suite of chemicals range from 0.001–20 mg/kg-day, with (E)-crotonaldehyde having the lowest chronic oral RfV and 1,2-propylene glycol having the highest. These RfVs were derived based on health effects including immune system effects, changes in body weight, changes in blood chemistry, cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and developmental toxicity. The total UFs used in the derivation of these chronic oral RfVs (Table 9-12) reflect varying degrees of confidence surrounding the data sets for these chemicals. Three of the chemicals with the lowest chronic oral RfVs [(E)-crotonaldehyde, propargyl alcohol, benzyl chloride] have total UFs of 3000, indicating a relatively large amount of uncertainty in these values. Comparatively, chemicals such as benzene, acrylamide, and dichloromethane also have low chronic oral RfVs, but with much less uncertainty reflected in the values.

Figure 9-8 presents the results of a noncancer MCDA for these 42 chemicals in hydraulic fracturing fluids. Of these 42 chemicals, propargyl alcohol received the highest overall Total Hazard Potential Score. Propargyl alcohol was reported in 33% of disclosures nationally in the EPA FracFocus 1.0 project database, making it one of the most widely used chemicals that was considered in this analysis. It has physicochemical properties that are conducive to transport in water, and a low RfV. Given these properties, propargyl alcohol received the highest overall ranking based on hazard potential across all of the metrics that were considered in the MCDA.

Several of the other chemicals that received high Occurrence Scores also received among the highest Total Hazard Potential Scores, including 2-butoxyethanol, naphthalene, 1,2,4-trimethylbenzene, N,N-dimethylformamide, and formaldehyde (reported in 23%, 19%, 13%, 9%, and 7% of disclosures, respectively). Methanol, ethylene glycol, and formic acid (73%, 47%, and 11% of disclosures, respectively) received lower Total Hazard Potential Scores as a result of having higher RfVs. Likewise, didecyldimethylammonium chloride and dodecylbenzenesulfonic acid (8% and 7% of disclosures, respectively) received lower Total Hazard Potential Scores as a result of having higher RfVs and more hydrophobic properties.

The other chemicals that received high Toxicity Scores (i.e., had low chronic oral RfVs) received moderate to high Total Hazard Potential Scores overall. Acrylamide was reported in only 1% of disclosures, but has physicochemical properties that are very conducive to transport in water, and therefore received one of the highest overall Total Hazard Potential Scores. 1,2,4-Trimethylbenzene, benzyl chloride, and epichlorohydrin (13%, 6%, and 1% of disclosures in the EPA FracFocus 1.0 project database, respectively) scored slightly lower than acrylamide with regards to physicochemical properties. Other chemicals, including 1,2,3-trimethylbenzene, 1,3,5-trimethylbenzene, (E)-crotonaldehyde, benzene, dichloromethane, aniline, furfural, and 2-(Thiocyanomethylthio)benzothiazole, received lower overall scores because they are used more infrequently (the trimethylbenzenes were reported in <1% of disclosures, and the rest reported in <0.1% of disclosures).

9.6.5.2 Results: Noncancer MCDA for Chemicals Used in Hydraulic Fracturing Fluids (State-specific analysis for Texas, Pennsylvania, and North Dakota)

To investigate the extent of regional differences and examine the applicability of the MCDA model at the regional scale, we repeated the noncancer MCDA for hydraulic fracturing fluids for subsets of chemicals used in three representative states that have a significant amount of hydraulic fracturing activity: Texas, Pennsylvania, and North Dakota. The chemicals used in these state-specific analyses are subsets of the chemicals used nationally, and are indicated in Table 9-12. Some of the chemicals considered in the national analysis were not included in the state-specific analyses because they were not disclosed to FracFocus 1.0 as used in these states.

Results are presented in Figure 9-9 (Texas), Figure 9-10 (Pennsylvania), and Figure 9-11 (North Dakota). By comparing these results to each other and to the national noncancer MCDA (Figure 9-8), it is evident that there are some regional differences in the Total Hazard Potential Scores, although many chemicals were commonly used and received similar overall rankings.

Methanol, ethylene glycol, and 2-butoxyethanol were among the most frequently reported chemicals in all three state-specific analyses, while other chemicals differed distinctly between states. For instance, propargyl alcohol was frequently reported in Texas (39% of disclosures) and Pennsylvania (58% of disclosures), but not North Dakota (1% of disclosures). Likewise, naphthalene was reported frequently in Texas (14% of disclosures) and North Dakota (43% of disclosures), but not in Pennsylvania (1% of disclosures). The most toxic chemicals (occurring in the lowest quartile of chronic oral RfVs) common among all three states include propargyl alcohol, benzyl chloride, acrylamide, and 1,2,4-trimethylbenzene. Other chemicals receiving high Toxicity Scores in these states include epichlorohydrine (Texas and Pennsylvania), 1,3,5-Trimethylbenzene (Texas and Pennsylvania), 1,4-dioxane (North Dakota), naphthalene (North Dakota), benzene, aniline, and 1,2,3-Trimethylbenzene (Texas).

Overall, in Texas, propargyl alcohol received the highest possible Total Hazard Potential Score, with acrylamide receiving the second highest score. In Pennsylvania, propargyl alcohol also received the highest possible Total Hazard Potential Score, with 2-butoxyethanol receiving the second highest score. In North Dakota, 2-butoxyethanol received the highest Total Hazard Potential Score, with naphthalene receiving the second highest score.

The results of these state-specific MCDAs support the concept presented in Chapter 5 that there is no single hydraulic fracturing fluid formulation, and that the chemicals of most potential concern will vary between regions or even between wells.

Table 9-12. Data on the selected subset of chemicals in hydraulic fracturing fluids used for input into a noncancer MCDA.
Chemicals within the table are ordered from most toxic to least toxic based on chronic oral RfV.

		Noncancer toxicity (chronic oral RfV)[a]			% disclosures in EPA FracFocus 1.0 project database[b]				Mobility			Volatility	Persist-ence
Chemical Name	CASRN	RfV (mg/kg-day)	Total UF	Source	National	TX	PA	ND	Log K_{OW}	Log K_{OC}	Solu-bility (mg/L)	Henry's Law Constant	Half-life in water (days)
(E)-Crotonaldehyde	123-73-9	0.001	3000	PPRTV	0.06%				0.6	0.254	41480	1.94E-05	15
Propargyl alcohol	107-19-7	0.002	3000	IRIS	33%	39%	58%	1%	-0.38	0.28	935500	1.15E-06	15
Benzyl chloride	100-44-7	0.002	3000	PPRTV	6%	7%	5%	0.80%	2.3	2.649	1030	4.12E-04	15
Acrylamide	79-06-1	0.002	30	IRIS	1%	2%	1%	1%	-0.67	0.755	504000	1.70E-09	15
Benzene	71-43-2	0.004	300	IRIS	0.006%	0.01%			2.13	1.75	2000	5.55E-03	37.5
Epichlorohydrin	106-89-8	0.006	1000	PPRTV	1%	0.20%	0.08%		0.45	1	50630	3.04E-05	15
Dichloromethane	75-09-2	0.006	30	IRIS	0.02%				1.25	1.44	10950	3.25E-03	37.5
Aniline	62-53-3	0.007	1000	PPRTV	0.02%	0.05%			0.9	1.6	20820	2.02E-06	15
1,2,4-Trimethylbenzene	95-63-6	0.01	300	IRIS	13%	11%	1%	25%	3.63	2.788	79.59	6.16E-03	37.5
1,3,5-Trimethylbenzene	108-67-8	0.01	300	IRIS	0.5%	0.80%	1%		3.42	2.82	120.3	8.77E-03	37.5
1,2,3-Trimethylbenzene	526-73-8	0.01	300	IRIS	0.4%	0.80%			3.66	2.8	75.03	4.36E-03	37.5
2-(Thiocyanomethyl-thio)benzothiazole	21564-17-0	0.01	300	HHBP	0.006%				3.3	3.528	41.67	6.49E-12	37.5
Furfural	98-01-1	0.01	3000	HHBP	0.003%				0.41	0.784	53580	3.77E-06	15
Naphthalene	91-20-3	0.02	3000	IRIS	19%	14%	1%	43%	3.3	2.96	142.1	4.40E-04	37.5

		Noncancer toxicity (chronic oral RfV)[a]			% disclosures in EPA FracFocus 1.0 project database[b]				Mobility			Volatility	Persist-ence
Chemical Name	**CASRN**	**RfV (mg/ kg-day)**	**Total UF**	**Source**	**National**	**TX**	**PA**	**ND**	**Log K_{OW}**	**Log K_{OC}**	**Solu-bility (mg/L)**	**Henry's Law Constant**	**Half-life in water (days)**
Chlorobenzene	108-90-7	0.02	1000	IRIS	0.003%	0.01%			2.84	2.15	400.5	3.11E-03	15
2-(2-Butoxyethoxy) ethanol	112-34-5	0.03	3000	PPRTV	0.6%	0.40%	4%		0.56	1	71920	7.20E-09	8.67
1,4-Dioxane	123-91-1	0.03	300	IRIS	0.3%	0.50%		0.80%	-0.27	0.421	213900	4.80E-06	15
1,3-Dichloropropene	542-75-6	0.03	100	IRIS	0.02%				2.04	1.82	1994	3.55E-03	37.5
Bisphenol A	80-05-7	0.05	1000	IRIS	0.006%	0.01%			3.32	4.576	172.7	9.16E-12	37.5
Toluene	108-88-3	0.08	3000	IRIS	0.7%	1%			2.73	2.07	573.1	6.64E-03	15
Ethylenediamine	107-15-3	0.09	100	PPRTV	0.01%	0.02%			-2.04	1.172	1000000	1.73E-09	15
2-Butoxyethanol	111-76-2	0.1	10	IRIS	23%	27%	21%	15%	0.83	0.451	64470	1.60E-06	8.67
N,N-Dimethylform-amide	68-12-2	0.1	1000	PPRTV	9%	10%	11%	0.60%	-1.01	0	977900	7.39E-08	15
Didecyldimethylam-monium chloride	7173-51-5	0.1	100	HHBP	8%	7%	12%	0.05%	4.66	5.546	0.9	6.85E-10	15
1-Butanol	71-36-3	0.1	1000	IRIS	1%	2%		0.70%	0.88	0.5	76700	8.81E-06	8.67
Cumene	98-82-8	0.1	1000	IRIS	0.5%	0.80%	1%		3.66	2.844	75.03	1.15E-02	15
Ethylbenzene	100-41-4	0.1	1000	IRIS	0.4%	0.50%		0.10%	3.15	2.23	228.6	7.88E-03	15
Acetophenone	98-86-2	0.1	3000	IRIS	0.04%	0.04%			1.58	1.8	4484	1.04E-05	15
Formaldehyde	50-00-0	0.2	100	IRIS	7%	8%	4%	8%	0.35	0	57020	3.37E-07	15
Xylenes	1330-20-7	0.2	1000	IRIS	2%	3%	1%	0.20%	3.2	2.25	207.2	7.18E-03	15

		Noncancer toxicity (chronic oral RfV)[a]			% disclosures in EPA FracFocus 1.0 project database[b]				Mobility			Volatility	Persist-ence
Chemical Name	**CASRN**	**RfV (mg/ kg-day)**	**Total UF**	**Source**	**National**	**TX**	**PA**	**ND**	**Log K_{OW}**	**Log K_{OC}**	**Solu-bility (mg/L)**	**Henry's Law Constant**	**Half-life in water (days)**
o-Xylene	95-47-6	0.2	30	ATSDR	0.05%	0.1%			3.12	2.25	224.1	5.18E-03	15
Phenol	108-95-2	0.3	300	IRIS	0.4%	0.80%		0.05%	1.46	1.9	26160	3.33E-07	15
2-Methyl-1-propanol	78-83-1	0.3	1000	IRIS	0.3%			4%	0.76	0.465	97120	9.78E-06	15
Dodecylbenzenesul-fonic acid	27176-87-0	0.5	100	HHBP	7%	10%	2%	8%	4.71	4.066	0.8126	6.27E-08	15
Formic acid	64-18-6	0.9	300	PPRTV	11%	14%	8%	11%	-0.54	0	955200	1.67E-07	8.67
Ethyl acetate	141-78-6	0.9	1000	IRIS	0.4%	0.70%			0.73	0.747	29930	1.34E-04	15
Acetone	67-64-1	0.9	1000	IRIS	0.2%	0.02%		1%	-0.24	0.374	219900	3.50E-05	15
Methanol	67-56-1	2	100	IRIS	73%	80%	69%	54%	-0.77	0.44	1000000	4.55E-06	8.67
Ethylene glycol	107-21-1	2	100	IRIS	47%	60%	35%	37%	-1.36	0	1000000	6.00E-08	8.67
Hexanedioic acid	124-04-9	2	300	PPRTV	0.70%	1%			0.08	1.386	167300	4.71E-12	8.67
Benzoic acid	65-85-0	4	1	IRIS	0.06%	0.10%	0.04%		1.87	1.5	2493	3.81E-08	15
1,2-Propylene glycol	57-55-6	20	300	PPRTV	4%	4%	8%	8%	-0.92	0.36	811100	1.29E-08	8.67

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; ATSDR = Agency for Toxic Substances and Disease Registry; HHBP = Human Health Benchmarks for Pesticides; K_{OW} = octanol-water partitioning coefficient; K_{OC} = soil adsorption coefficient

[a] Reference value (RfV): An estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfVs considered in the MCDA include chronic oral reference doses (RfD) from IRIS, PPRTV, and HHBP; and chronic oral minimal risk levels (MRLs) from ATSDR.

[b] The FracFocus frequency of use data presented in this chapter is based on 35,957 FracFocus disclosures that were deduplicated, within the study time period (January 1, 2011 to February 28, 2013), and with ingredients that have a valid CASRN.

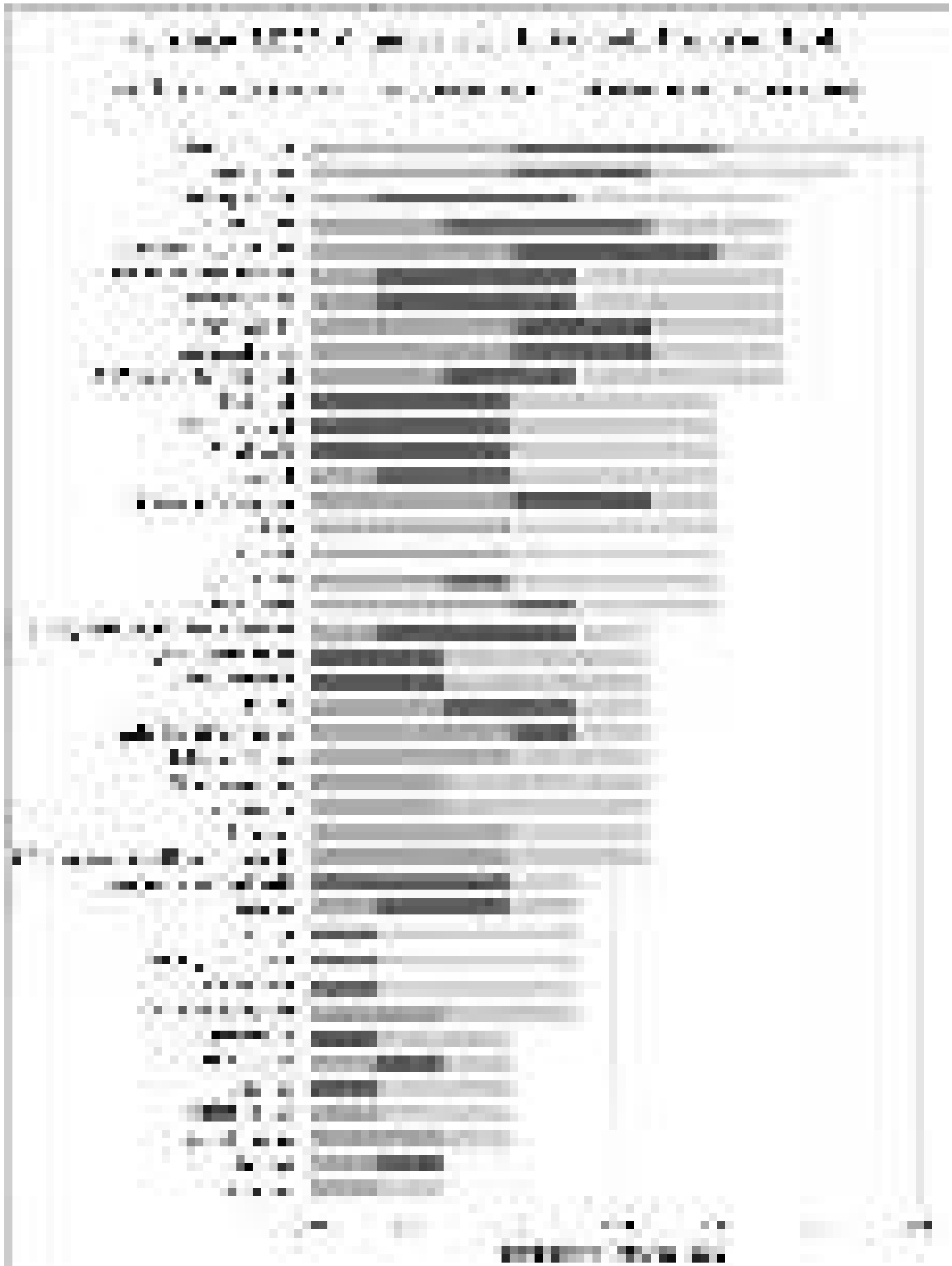

Figure 9-8. Noncancer MCDA results for 42 chemicals used in hydraulic fracturing fluids (national analysis), showing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score for each chemical.

Chemicals are ordered from high to low based on Total Hazard Potential Score. See Section 9.6.4 for details on the calculation.

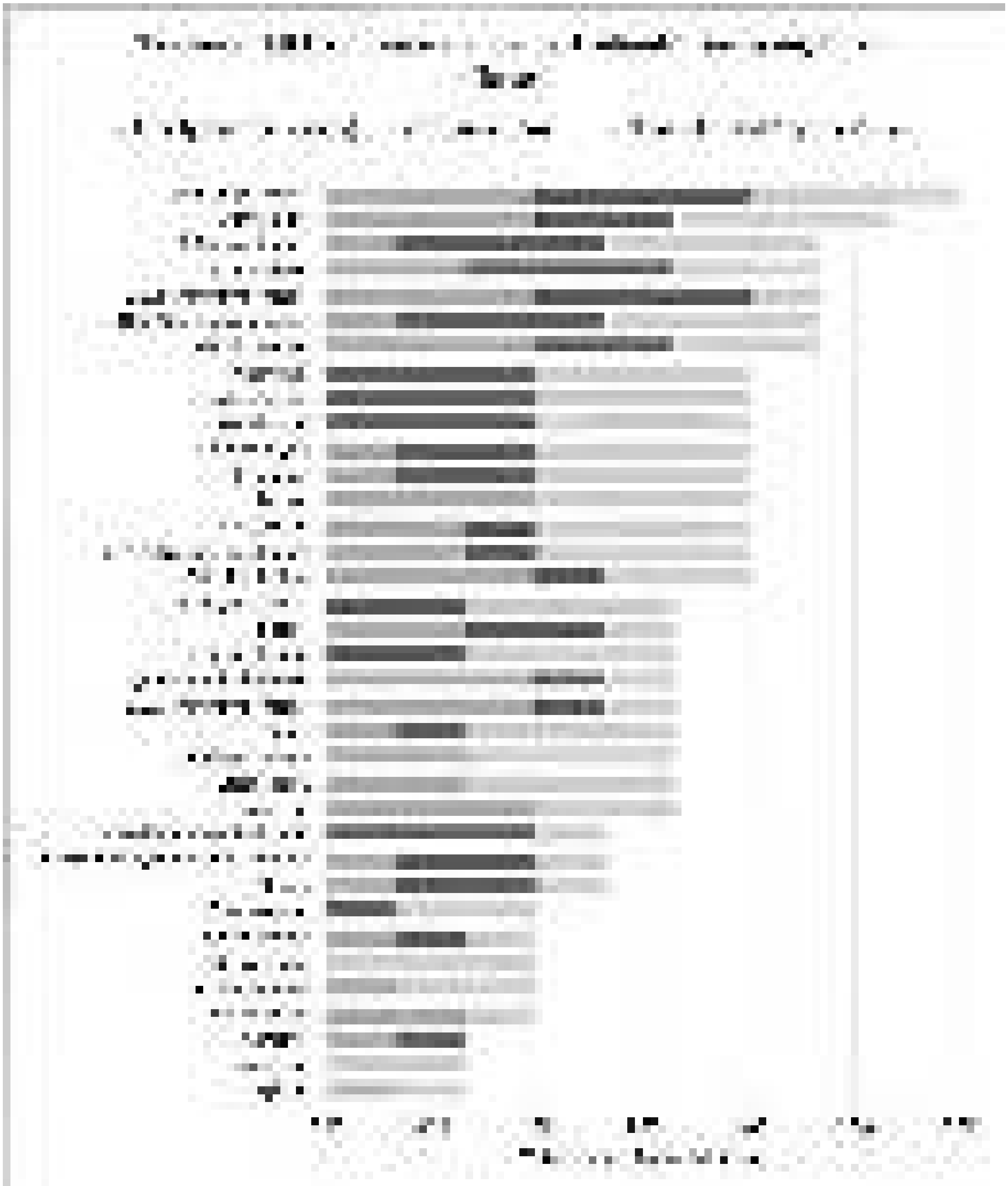

Figure 9-9. Noncancer MCDA results for 36 chemicals used in hydraulic fracturing fluids in Texas (state-specific analysis), showing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score for each chemical.

Chemicals are ordered from high to low based on Total Hazard Potential Score. See Section 9.6.4 for details on the calculation.

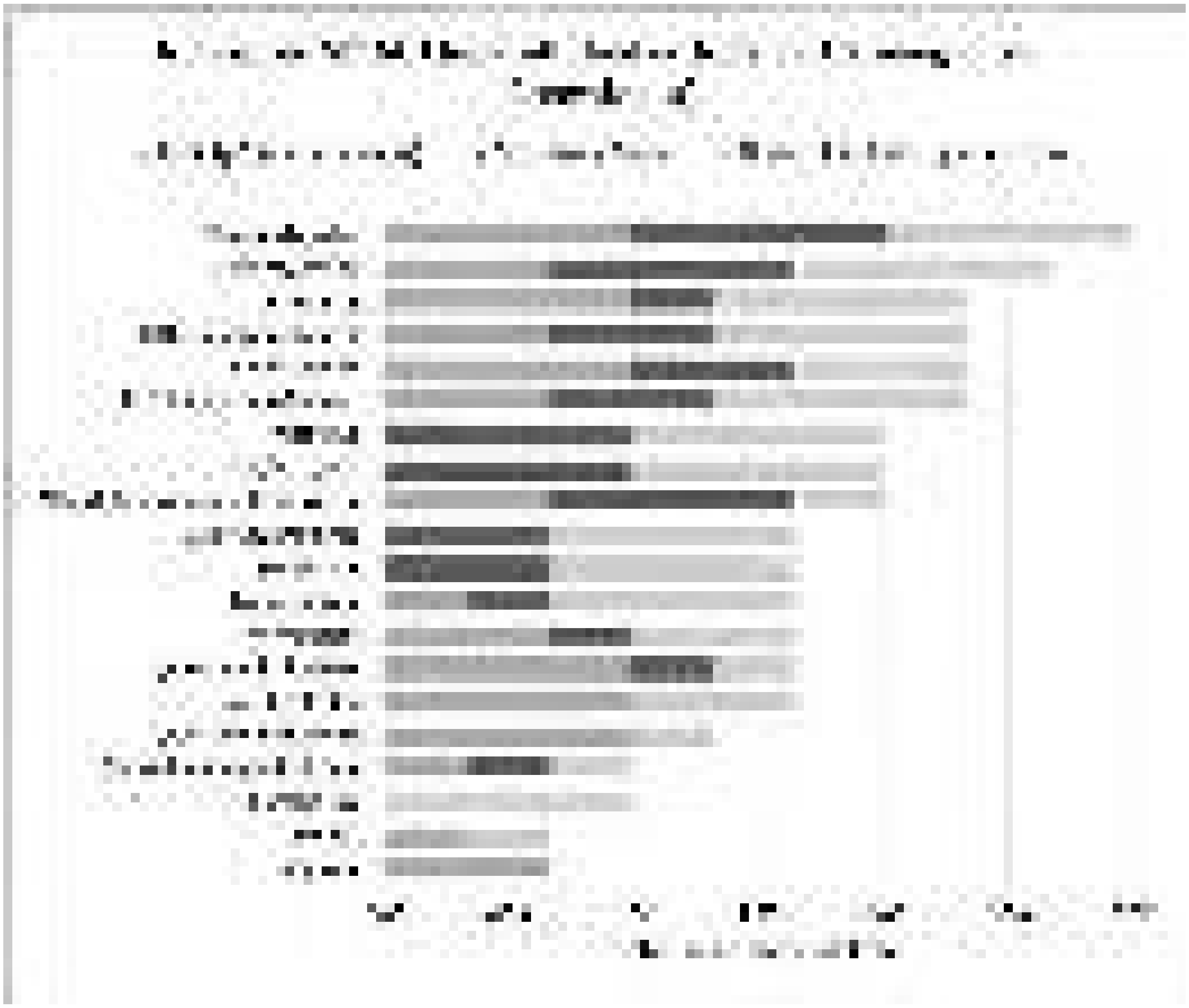

Figure 9-10. Noncancer MCDA results for 20 chemicals used in hydraulic fracturing fluids in Pennsylvania (state-specific analysis), showing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score for each chemical.

Chemicals are ordered from high to low based on Total Hazard Potential Score. See Section 9.6.4 for details on the calculation.

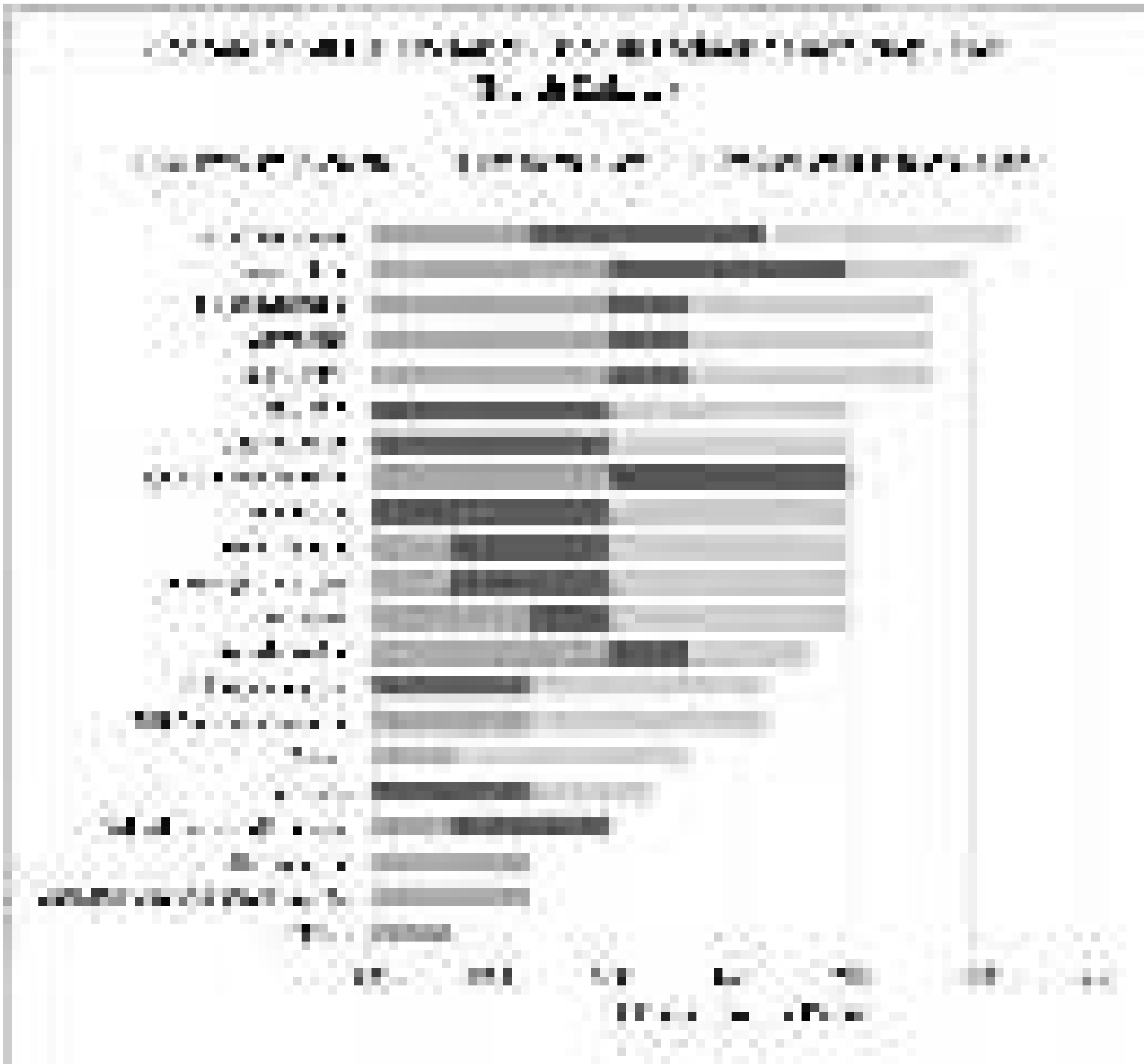

Figure 9-11. Noncancer MCDA results for 21 chemicals used in hydraulic fracturing fluids in North Dakota (state-specific analysis), showing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score for each chemical.

Chemicals are ordered from high to low based on Total Hazard Potential Score. See Section 9.6.4 for details on the calculation.

9.6.5.3 Results: Cancer MCDA for Chemicals Used in Hydraulic Fracturing Fluids

A total of 10 chemicals used in hydraulic fracturing fluids were evaluated in a cancer MCDA (Table 9-13). OSFs for these chemicals ranged from 0.002 to 3 per mg/kg-day, with quinoline having the highest OSF, and dichloromethane having the lowest. Benzene is the only one of these chemicals that is classified as a known human carcinogen by at least one of the sources in Table 9-1, while the other chemicals in this subset are classified as probable carcinogens in humans (Appendix Table G-1e).

Figure 9-12 presents the results from the cancer MCDA for chemicals used in hydraulic fracturing fluids. Of the 10 chemicals that were considered in this analysis, acrylamide received the highest Total Hazard Potential Score. Acrylamide has an OSF of 0.5 per mg/kg-day, which is one of the higher OSFs in this suite of chemicals, and has physicochemical properties that are highly conducive to transport in water. Acrylamide was reported in 1% of disclosures nationally in the EPA FracFocus 1.0 project database. This nevertheless places acrylamide in the top quartile in terms of frequency of use, as none of the chemicals within this subset were used with great frequency on a national basis.

Bis(2-chloroethyl)ether and quinoline, which are the two most potent carcinogens considered in the analysis and received high Toxicity Score, received the second and third highest Total Hazard Potential Scores within this suite of chemicals. Bis(2-chloroethyl)ether was reported in 0.7% of disclosures, while quinoline was reported in 0.02% of disclosures. Both are expected to be readily transported in water.

In addition to acrylamide, the other two chemicals receiving high Occurrence Scores were benzyl chloride and epichlorohydrin (6% and 1% of disclosures, respectively). These two chemicals both received moderate Total Hazard Potential Scores. Benzyl chloride has an OSF of 0.17 per mg/kg-day, while epichlorohydrine has an OSF of 0.0099 per mg/kg-day. Both received lower Physicochemical Properties Scores relative to other chemicals in this analysis, due in part to volatility.

Table 9-13. Data on the selected subset of chemicals in hydraulic fracturing fluids used for input into a cancer MCDA.
Chemicals within the table are ordered from most potent to least potent based on OSF.

		Cancer-specific toxicity (OSF)[a]		% disclosures in EPA FracFocus 1.0 project database[b]	Mobility			Volatility	Persistence
Chemical Name	**CASRN**	**OSF (per mg/kg-day)**	**Source**	**National**	**Log K_{OW}**	**Log K_{OC}**	**Solubility (mg/L)**	**Henry's Law Constant**	**Half-life in water (days)**
Quinoline	91-22-5	3	IRIS	0.02%	2.03	3.1	1711	1.67E-06	15
Bis(2-chloroethyl) ether	111-44-4	1.1	IRIS	0.7%	1.29	1.88	6435	1.70E-05	37.5
Acrylamide	79-06-1	0.5	IRIS	1%	-0.67	0.755	504000	1.70E-09	15
Benzyl chloride	100-44-7	0.17	IRIS	6%	2.3	2.649	1030	4.12E-04	15
1,4-Dioxane	123-91-1	0.1	IRIS	0.3%	-0.27	0.421	213900	4.80E-06	15
Benzene	71-43-2	0.015-0.055[c]	IRIS	0.006%	2.13	1.75	2000	5.55E-03	37.5
1,3-Dichloropropene	542-75-6	0.05	IRIS	0.02%	2.04	1.82	1994	3.55E-03	37.5
Epichlorohydrin	106-89-8	0.0099	IRIS	1%	0.45	1	50630	3.04E-05	15
Aniline	62-53-3	0.0057	IRIS	0.02%	0.9	1.6	20820	2.02E-06	15
Dichloromethane	75-09-2	0.002	IRIS	0.02%	1.25	1.44	10950	3.25E-03	37.5

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; K_{OW} = octanol-water partitioning coefficient; K_{OC} = soil adsorption coefficient

[a] Oral slope factor (OSF): An upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100. OSFs considered in the MCDA include values from IRIS, PPRTV, and HHBP.

[b] The FracFocus frequency of use data presented in this chapter is based on 35,957 FracFocus disclosures that were deduplicated, within the study time period (January 1, 2011 to February 28, 2013), and with ingredients that have a valid CASRN.

[c] IRIS lists the OSF for benzene as a range from 0.015 to 0.055 per mg/kg-day. For input into the MCDA, we used the high end of this range (0.055 per mg/kg-day).

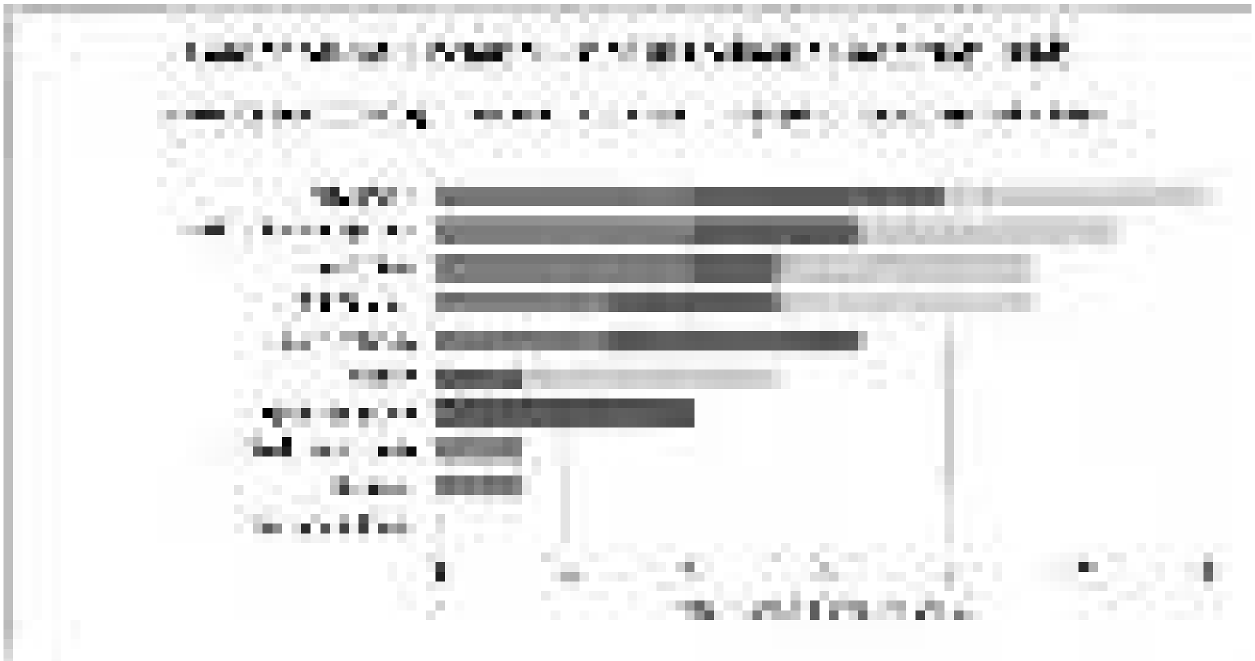

Figure 9-12. Cancer MCDA results for 10 chemicals used in hydraulic fracturing fluids, showing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score for each chemical.

Chemicals are ordered from high to low based on Total Hazard Potential Score. See Section 9.6.4 for details on the calculation.

9.6.5.4 Results: Noncancer MCDA for Chemicals in Produced Water

A total of 29 chemicals detected in produced water were evaluated in a noncancer MCDA (Table 9-14). Of these 29 chemicals, 13 were also included in the noncancer MCDA for hydraulic fracturing fluids. Chronic oral RfVs within this suite of chemicals range from 0.001 to 0.9 mg/kg-day, with pyridine having the lowest chronic oral RfV, and acetone having the highest. Chronic oral exposure to these chemicals may induce a variety of adverse outcomes, including immune system effects, changes in body weight, changes in blood chemistry, pulmonary toxicity, neurotoxicity, liver and kidney toxicity, and reproductive and developmental toxicity. The total UFs used in the derivation of these chronic oral RfVs (Table 9-14) reflect varying degrees of confidence surrounding the data sets for these chemicals.

Figure 9-13 presents the results of a noncancer MCDA for these 29 chemicals detected in produced water. Benzene, pyridine, and naphthalene received the highest Total Hazard Potential Scores, followed by 2-methylnaphthalene. These four chemicals all received high Toxicity Scores and high Occurrence Scores (with maximum average concentrations of 1500 μg/L, 413 μg/L, 238 μg/L, and 1362 μg/L in Barnett, Marcellus, or Powder River Basin produced water, respectively), but received moderate to low Physicochemical Property Scores.

Table 9-14. Data on the selected subset of chemicals detected in produced water used for input into a noncancer MCDA.
Chemicals within the table are ordered from most toxic to least toxic based on chronic oral RfV.

Chemical Name	CASRN	Noncancer toxicity (chronic oral RfV)[a]			Occurrence (concentration in produced water)[b]		Mobility			Volatility	Persist-ence
		RfV (mg/kg-day)	Total UF	Source	Average or Median Conc. (µg/L)	Reference	Log K_{OW}	Log K_{OC}	Solubility (mg/L)	Henry's Law Constant	Half-life in water (days)
Pyridine	110-86-1	0.001	1000	IRIS	413	Table E-11	0.65	1.6	729800	1.10E-05	15
Benzene	71-43-2	0.004	300	IRIS	1500	Table E-13	2.13	1.75	2000	5.55E-03	37.5
2-Methylnaphthalene	91-57-6	0.004	1000	IRIS	1362	Table E-11	3.86	3.6	41.42	5.18E-04	15
1,2,4-Trimethylbenzene	95-63-6	0.01	300	IRIS	173	Table E-11	3.63	2.788	79.59	6.16E-03	37.5
1,3,5-Trimethylbenzene	108-67-8	0.01	300	IRIS	59	Table E-11	3.42	2.82	120.3	8.77E-03	37.5
Chloroform	67-66-3	0.01	1000	IRIS	28	Table E-11	1.97	1.6	2096	3.67E-03	37.5
Tributyl phosphate	126-73-8	0.01	1000	PPRTV	0.26	Table E-12	4	3.371	7.355	1.41E-06	8.67
Naphthalene	91-20-3	0.02	3000	IRIS	238	Table E-11	3.3	2.96	142.1	4.40E-04	37.5
Di(2-ethylhexyl) phthalate[c]	117-81-7	0.02	1000	IRIS	210	Table E-11	7.6	4.94	0.001132	2.70E-07	15
Chlorobenzene[d]	108-90-7	0.02	1000	IRIS	100	Table E-13	2.84	2.15	400.5	3.11E-03	15
2,4-Dimethylphenol	105-67-9	0.02	3000	IRIS	14.5	Table E-11	2.3	2.692	4068	9.51E-07	15
Pyrene	129-00-0	0.03	3000	IRIS	13	Table E-11	4.88	4.9	0.2249	1.19E-05	60
1,4-Dioxane	123-91-1	0.03	300	IRIS	6.5	Table E-11	-0.27	0.421	213900	4.80E-06	15
Fluorene	86-73-7	0.04	3000	IRIS	8.4	Table E-11	4.1	3.614	20.13	1.59E-03	15
Fluoranthene	206-44-0	0.04	3000	IRIS	6.1	Table E-11	5.16	4.8	0.1297	8.86E-06	60
o-Cresol[e]	95-48-7	0.05	1000	IRIS	28.3	Table E-11	1.95	2.486	9066	1.20E-06	15
Toluene	108-88-3	0.08	3000	IRIS	760	Table E-9	2.73	2.07	573.1	6.64E-03	15

		Noncancer toxicity (chronic oral RfV)[a]			Occurrence (concentration in produced water)[b]		Mobility			Volatility	Persist-ence
Chemical Name	**CASRN**	**RfV (mg/kg-day)**	**Total UF**	**Source**	**Average or Median Conc. (µg/L)**	**Reference**	**Log K_{OW}**	**Log K_{OC}**	**Solubility (mg/L)**	**Henry's Law Constant**	**Half-life in water (days)**
Ethylbenzene	100-41-4	0.1	1000	IRIS	2010	Table E-13	3.15	2.23	228.6	7.88E-03	15
Carbon disulfide	75-15-0	0.1	100	IRIS	400	Table E-11	1.94	1.337	2928	1.44E-02	15
Cumene[f]	98-82-8	0.1	1000	IRIS	120	Table E-11	3.66	2.844	75.03	1.15E-02	15
Benzyl alcohol	100-51-6	0.1	1000	PPRTV	81.5	Table E-11	1.1	1.1	41050	3.37E-07	15
Dibutyl phthalate[g]	84-74-2	0.1	1000	IRIS	41	Table E-11	4.5	3.14	2.351	1.81E-06	8.67
Acetophenone	98-86-2	0.1	3000	IRIS	13	Table E-11	1.58	1.8	4484	1.04E-05	15
Diphenylamine	122-39-4	0.1	100	HHBP	5.3	Table E-11	3.5	2.78	63.61	2.69E-06	37.5
Xylenes	1330-20-7	0.2	1000	IRIS	360	Table E-9	3.2	2.25	207.2	7.18E-03	15
Benzyl butyl phthalate	85-68-7	0.2	1000	IRIS	34.3	Table E-11	4.73	3.72	0.9489	1.26E-06	0.04
Phenol	108-95-2	0.3	300	IRIS	63	Table E-11	1.46	1.9	26160	3.33E-07	15
Caprolactam	105-60-2	0.5	100	IRIS	0.75	Table E-12	0.66	1.3892	28720	2.53E-08	14508
Acetone	67-64-1	0.9	1000	IRIS	145	Table E-10	-0.24	0.374	219900	3.50E-05	15

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; HHBP = Human Health Benchmarks for Pesticides; K_{OW} = octanol-water partitioning coefficient; K_{OC} = soil adsorption coefficient

[a] Reference value (RfV): An estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfVs considered in the MCDA include chronic oral reference doses (RfD) from IRIS, PPRTV, and HHBP; and chronic oral minimal risk levels (MRLs) from ATSDR.

[b] From Appendix E.

[c] Di(2-ethylhexyl) phthalate is listed under the name bis(2-ethylhexyl) phthalate in Appendix Table E-11.

[d] Chlorobenzene is listed under the name chloro-benzene in Appendix Table E-13.

[e] o-Cresol is listed under the name 2-methylphenol in Appendix Table E-11.

[f] Cumene is listed under the name isopropylbenzene in Appendix Table E-11.

[g] Dibutyl phthalate is listed under the name dibutyl-n-phthalate in Appendix Table E-11.

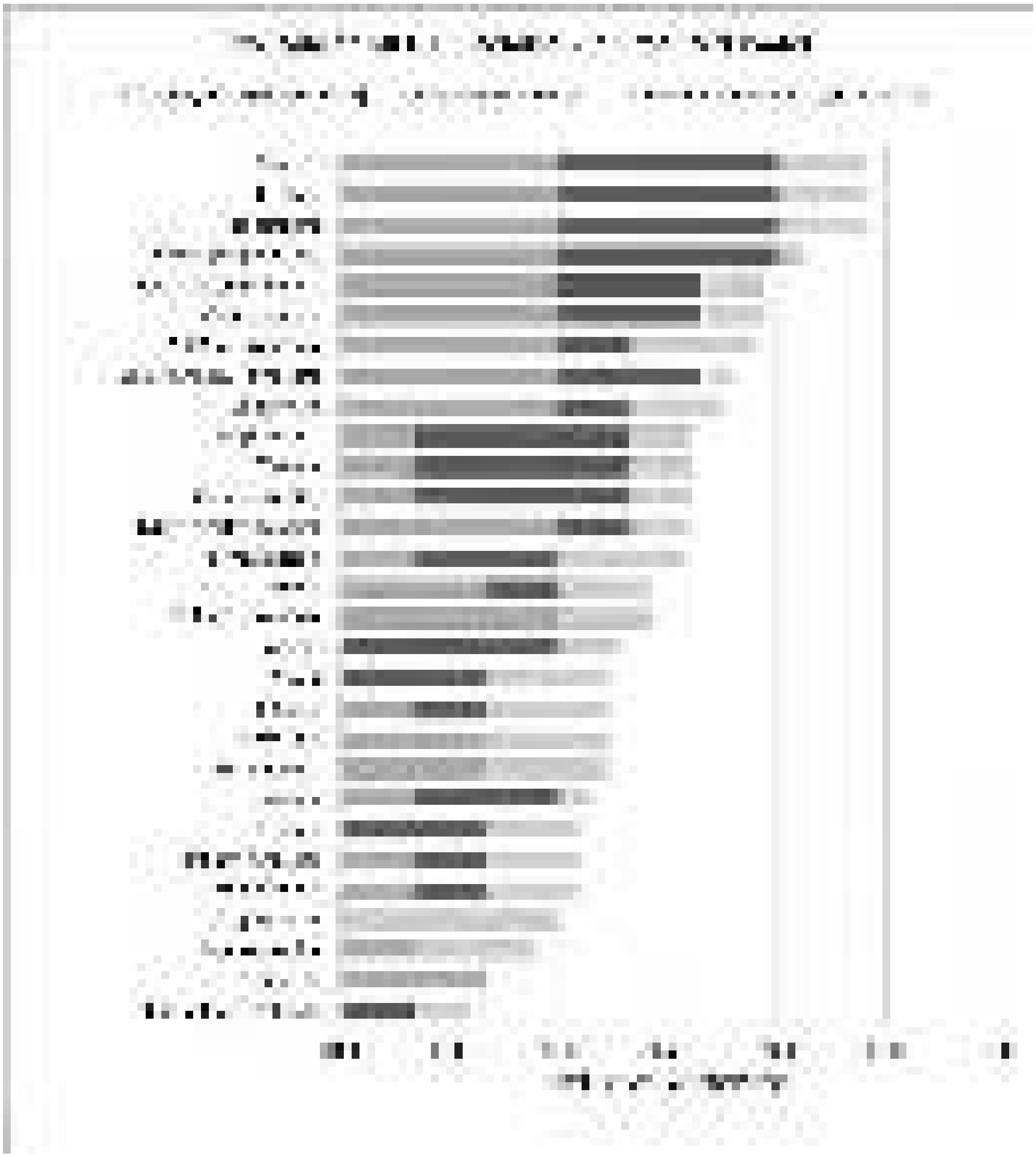

Figure 9-13. Noncancer MCDA results for a subset of 29 chemicals detected in produced water, showing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score for each chemical.

Chemicals are ordered from high to low based on Total Hazard Potential Score. See Section 9.6.4 for details on the calculation.

The other chemicals that received high Toxicity Scores were 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, chloroform, 2,4,-dimethylphenol, tributyl phosphate, di(2-ethylhexyl) phthalate, and chlorobenzene. These chemicals received moderate Total Hazard Potential Scores, as all were detected at lower concentrations compared to other chemicals considered in this analysis and are expected to have moderate transport in water.

The other chemicals that received high Occurrence Scores are ethylbenzene, toluene, xylenes, and carbon disulfide, which were detected at maximum average concentrations of 2010 µg/L, 760 µg/L, 360 µg/L, and 400 µg/L in Barnett, Marcellus, or Powder River Basin produced water. These chemicals received moderate Total Hazard Potential Scores, as all have as all have higher chronic oral RfVs relative to many of the other chemicals in the hazard evaluation, and are all expected to have moderate transport in water relative to the other chemicals.

9.6.5.5 Results: Cancer MCDA for Chemicals in Produced Water

A total of 7 chemicals reported in produced water were evaluated in a cancer MCDA (Table 9-15). OSFs within this suite of chemicals ranged from 7.3 to 0.0049 per mg/kg-day, with benzo(a)pyrene having the highest OSF and N-nitrosodiphenylamine having the lowest. Of these 7 chemicals, benzene and 1,4-dioxane were also included in the cancer MCDA for chemicals used in hydraulic fracturing fluids. Benzene and benzo(a)pyrene are both classified by at least one of the sources in Table 9-1 as a known human carcinogen, while the other chemicals as classified as likely or probable carcinogens in humans (Appendix G: Tables G-1e and G-2e).

Figure 9-14 presents the results of a cancer MCDA for these 7 chemicals in hydraulic fracturing fluids. Benzene and benzo(a)pyrene tied for highest Total Hazard Potential Scores. Of these, benzene was detected at the highest average concentrations in produced water (1500 µg/L in Power River Basin produced water), while benzo(a)pyrene were detected at lower average concentrations (6.7 µg/L in Barnett shale produced water). Benzo(a)pyrine and 1,2-diphenylhydrazine were the most potent carcinogens within this suite of chemicals and received high Toxicity Scores.

The other chemical that received a high Occurrence Score was di(2-ethylhexyl) phthalate, which was detected at an average concentration of 210 µg/L in Barnett Shale produced water. It received a moderate Total Hazard Potential Score because it is hydrophobic and not expected to be readily transported in water.

Table 9-15. Data on the selected subset of chemicals detected in produced water used for input into a cancer MCDA.
Chemicals within the table are ordered from most potent to least potent based on OSF.

		Cancer-specific toxicity (OSF)[a]		Occurrence (concentration in produced water)[b]		Mobility			Volatility	Persistence
Chemical Name	CASRN	OSF (per mg/kg-day)	Source of OSF	Average or Median Conc. (μg/L)	Reference	Log K_{OW}	Log K_{OC}	Solubility (mg/L)	Henry's Law Constant	Half-life in water (days)
Benzo(a)pyrene	50-32-8	7.3	IRIS	6.7	Table E-11	6.13	5.95	0.01038	4.57E-07	60
1,2-Diphenylhydrazine	122-66-7	0.8	IRIS	4.2	Table E-11	2.94	2.98	161.9	4.78E-07	28.17
1,4-Dioxane	123-91-1	0.1	IRIS	6.5	Table E-11	-0.27	0.421	213900	4.80E-06	15
Benzene	71-43-2	0.015-0.055[c]	IRIS	1500	Table E-13	2.13	1.75	2000	5.55E-03	37.5
Di(2-ethylhexyl) phthalate[d]	117-81-7	0.014	IRIS	210	Table E-11	7.6	4.94	0.001132	2.70E-07	15
Tributyl phosphate	126-73-8	0.009	PPRTV	0.26	Table E-12	4	3.371	7.355	1.41E-06	8.67
N-Nitrosodiphenylamine	86-30-6	0.0049	IRIS	8.9	Table E-11	3.13	3.42	94.85	1.21E-06	37.5

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Values; K_{OW} = octanol-water partitioning coefficient; K_{OC} = soil adsorption coefficient

[a] Oral slope factor (OSF): An upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg-day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100. OSFs considered in the MCDA include values from IRIS, PPRTV, and HHBP.

[b] From Appendix E.

[c] IRIS lists the OSF for benzene as a range from 0.015 to 0.055 per mg/kg-day. For input into the MCDA, we used the high end of this range (0.055 per mg/kg-day).

[d] Di(2-ethylhexyl) phthalate is listed under the name bis(2-ethylhexyl) phthalate in Appendix Table E-11.

Figure 9-14. Cancer MCDA results for 7 chemicals detected in produced water, showing the Toxicity Score, Occurrence Score, and Physicochemical Properties Score for each chemical.
Chemicals are ordered from high to low based on Total Hazard Potential Score. See Section 9.6.4 for details on the calculation.

9.6.6 Limitations and Uncertainty of the MCDA Framework

While this MCDA framework provides a simple and transparent tool for exploring the relative hazard potential of chemicals in the hydraulic fracturing water cycle, it is intended only as a preliminary analysis. It is important to acknowledge the limitations of this analysis, as well as the limitations of the parameters that were used for input in the MCDA.

Chronic oral RfVs and OSFs were selected for the MCDA because they are a primary focus of the toxicological evaluation presented in this chapter. We were interested in placing these values in the context of variables that may impact the likelihood of human exposure. These toxicity values were available for a relatively small fraction of chemicals on EPA's list, which limited the number of chemicals considered in the MCDA.

The FracFocus 1.0 data used in the MCDA does not represent a complete record of hydraulic fracturing chemical usage in the United States, as described in more detail in Chapter 5 and in Section 9.3.1. Frequency of use also does not reflect the volume or concentration of chemical usage, and therefore is an incomplete metric for potential exposure. The EPA FracFocus 1.0 project database provides data on the maximum concentration of chemicals in additives and in hydraulic fracturing fluid, as discussed in Section 5.4, but we elected not to use this data in the MCDA because reported concentrations for each chemical varied widely between disclosures (see Table 5-5 and volume estimates in Figure 5-5), making it difficult to determine a chemical concentration to use in an MCDA. Additionally, many chemicals in the EPA FracFocus 1.0 project database did not have valid concentration data; for instance, the maximum concentrations of a chemical in additive often added up to greater than 100%. We therefore elected to focus on frequency of use as a general metric of chemical occurrence in the hydraulic fracturing water cycle.

The produced water concentrations used in the MCDA are based on the compilation of data presented in Appendix E. While this data reflects the findings of recent studies, it does not represent a complete record of chemicals present in produced water, as described in more detail in Chapter 7 and in Section 9.3.2. Concentrations in produced water also do not necessarily reflect the concentrations in treated wastewater, drinking water wells, or residuals in soil or sediment. Concentrations of these chemicals in treated wastewater or well water would likely be more dilute compared to concentrations in produced water. Concentrations in soils or sediments may be higher, particularly for hydrophobic chemicals.

The physicochemical properties from EPI Suite used in the MCDA are useful for making comparison across chemicals, but these values are also subject to uncertainty. Many of the values used in the MCDA were estimated by EPI Suite, and therefore are subject to the inherent limitations of the EPI Suite model (Section 5.8). Chemical fate and transport will be also influenced by environmental and site-specific conditions, which are outside the scope of this analysis. For instance, the half-lives used to develop the Physicochemical Properties Score are estimated values that assume aerobic conditions, and thus may underestimate the expected half-life under anaerobic conditions (e.g., in a groundwater contaminant plume). If chemicals are present in a mixture, as inevitably occurs in hydraulic fracturing fluids and in the subsurface environment, fate and transport will be influenced by changes in solubility or degradation resulting from interactions with other chemicals.

There are also fundamental limitations with regards to the scope of the MCDA. The chemicals used in these analyses may not be representative of chemicals at a specific field site. The analysis only examined organic chemicals, as EPI Suite is not able to estimate physicochemical properties of inorganic chemicals. Additionally, the physicochemical properties used in the MCDA were chosen specifically to reflect chemical transport in water, and therefore do not highlight the potential hazards of hydrophobic or volatile chemicals. Hydrophobic chemicals may serve as long-term sources of pollution by sorbing to soils or sediments at contaminated sites, and volatile chemicals may be hazardous when inhaled. This analysis also does not attempt to address bioavailability or toxicokinetics, which may be influenced by physicochemical properties such as log K_{ow}. For instance, chemicals with log K_{ow} of 2-4 tend to absorb well through biological membranes, while chemicals with log K_{ow} > 4 tend not to absorb well, and those with log K_{ow} of 5-7 tend to bioconcentrate (U.S. EPA, 2012i).

9.6.7 Application of the MCDA Framework for Preliminary Hazard Evaluation

The MCDA framework presented here is intended as a preliminary analysis, and illustrates one possible method for integrating data to explore potential hazards. By combining multiple lines of data, we can stratify chemicals according to estimated hazard potential, and gain preliminary insight into those chemicals that may be of more concern than others to drinking water resources.

Researchers may find this approach useful in their efforts to explore the potential hazards of chemicals present at specific field sites, particularly in instances when exposure assessment data is not available. The MCDA framework is flexible, and could be adapted to incorporate site-specific data on chemical usage, different types of toxicity data, as well as other variables that may be of interest for risk assessment. For instance, rather than focusing on RfVs and OSFs from US federal sources, one could choose to derive the Toxicity Score using other sources of relevant toxicity information. Additionally, one could choose to perform this analysis using different physicochemical property inputs, to highlight chemical interactions with different environmental media (e.g., hydrophobic or volatile chemicals). Researchers could also choose to apply different weights to each of the three criteria considered in this analysis (toxicity, occurrence, physicochemical properties), to reflect expert judgement of each variable's relative importance.

9.7 Synthesis

The overall objective of this chapter was to identify and provide information on the toxicological properties of chemicals used in hydraulic fracturing and of hydraulic fracturing wastewater constituents, and to evaluate the potential hazards of these chemicals for drinking water resources. Toward this end, the EPA developed a list of 1,606 chemicals that are reported to be associated with hydraulic fracturing, separating them into subsets based on whether they were reported to have been used in hydraulic fracturing fluids (1,084 chemicals total) or detected in produced water (599 chemicals total). To evaluate the potential hazards of these chemicals, the EPA compiled chronic oral RfVs, OSFs, and qualitative cancer classifications from selected federal, state, and international sources that met the EPA's criteria for consideration in this assessment. This toxicological information was used to conduct an initial identification of the potential human health hazards associated with several subsets of chemicals identified as being of particular interest in previous chapters of this report. Finally, in order to illustrate how data integration could be used to explore potential hazards, an MCDA framework was used to evaluate selected subsets of chemicals based on toxicity, environmental occurrence, and physicochemical properties affecting chemical transport in water.

9.7.1 Summary of Findings

A major finding of this chapter was that chronic oral RfVs and OSFs were not available for the majority of chemicals that the EPA has identified as being associated with hydraulic fracturing activity, indicating that the majority of these chemicals have not undergone significant toxicological evaluation. Similarly, there have been several recent peer-reviewed studies that have attempted to gather toxicological information for subsets of chemicals that are used in hydraulic fracturing fluids, and they have found that many of these chemicals do not have toxicity values available (Elliott et al., 2016; Wattenberg et al., 2015; Stringfellow et al., 2014; Colborn et al., 2011). Taken together, this suggests a potentially significant knowledge gap exists with respect to the scientific community's understanding of the potential human health impacts of these chemicals. With the limited availability of toxicity values, risk assessment is difficult, and potential impacts on drinking water resources may not be assessed adequately. This lack of toxicity values is not unique to the hydraulic fracturing industry; in fact, it has been estimated that there are tens of thousands of chemicals in commercial use that have not undergone significant toxicological evaluation (Judson et al., 2009).

There are a variety of chemicals associated with hydraulic fracturing known to be hazardous to human health. Chronic oral RfVs or OSFs from the sources considered by the EPA in this assessment were available for 98 (9%) of the 1,084 chemicals used in hydraulic fracturing fluids, and 120 (20%) of the 599 chemicals detected in hydraulic fracturing produced water. Potential hazards associated with chronic oral exposure to these chemicals include carcinogenesis, immune system effects, changes in body weight, changes in blood chemistry, cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and developmental toxicity. Methane is not considered to be toxic when ingested, but may accumulate to explosive levels or act as an asphyxiant. DBPs formed during wastewater treatment can contribute to an increased risk of cancer, anemia, liver and kidney effects, and central nervous system effects, with brominated forms of DBPs considered to be more cytotoxic, genotoxic, and carcinogenic than chlorinated species.

To assess the toxicity of chemicals that lack chronic oral RfVs and OSF, risk assessors will need to turn towards alternative data sources. This chapter explored two alternative data sources that may provide useful information. QSAR-based toxicity estimates—specifically, rat chronic oral LOAEL estimates generated using TOPKAT—were available for many of the chemicals that lacked chronic oral RfVs and OSFs from the sources considered in this assessment, and may be used to rank chemicals based on toxicity when other data are not available. Additionally, many of these chemicals have information available on the EPA's ACToR database, which is an online data warehouse designed to consolidate large and disparate amounts of chemical data. The information available in the ACToR data warehouse ranges from the selected RfVs and OSFs discussed in this assessment, which have undergone extensive peer review, to toxicological data that have undergone little-to-no peer review.

When considering the potential impact of chemicals on drinking water resources and human health, it is important to consider exposure as well as toxicological properties. As discussed in previous chapters of this report and highlighted in this chapter, events such as spills, leaks from storage pits, and discharge of inadequately treated wastewater have led to the entry of hydraulic fracturing-related chemicals into drinking water resources. In some instances, chemical concentrations in surface water or groundwater were in exceedance of MCLs, indicating their presence at levels that could impact human health. While these studies demonstrate the potential entry of these chemicals into drinking water resources, there is a lack of systematic studies examining actual human exposures to these chemicals in drinking water as a result of hydraulic fracturing activity.

In the absence of exposure assessment data, the MCDA framework presented in this chapter provides a preliminary analysis of the relative hazard potential of these chemicals. In this context, occurrence and physicochemical property data were used as metrics to estimate the likelihood that a chemical could reach and impact drinking water, and toxicity data was used as a metric for the potential severity of an impact. This analysis highlighted several chemicals that may be more likely than others to reach drinking water and create a toxicological hazard. Of the chemicals used in hydraulic fracturing fluids that were considered in this analysis, chemicals such as propargyl alcohol stood out as having high potential toxicity, high frequency of use, and physicochemical properties that are conducive to transport in water. Of the chemicals in produced water, chemicals such as benzene, pyridine, 2-methylnaphthalene, and naphthalene stood out as having high potential toxicity, high concentrations in produced water, and physicochemical properties that are conducive to transport in water.

9.7.2 Factors Affecting the Frequency or Severity of Impacts

There are multiple pieces of information that could be taken into account when evaluating the frequency and severity of impacts that these chemicals may have on drinking water resources. This includes knowledge of the chemicals used at a given site, the toxicological and physicochemical properties of these chemicals, the amount of fluid being used and recovered, the likelihood of mechanical integrity failures, the likelihood of spills and other unintentional releases, and the efficiency of chemical removal during wastewater treatment. The MCDA presented in this chapter incorporated parameters that may impact the likelihood of chemical exposure, including frequency of use, measured concentration, and transport in water, and was used to stratify and rank chemicals based on relative hazard potential. However, it should be considered only as a preliminary analysis, and should not be used in place of local data on the concentrations and volumes of chemicals in areas of hydraulic fracturing activity.

Analysis of the chemicals used in hydraulic fracturing fluids indicated that the majority of chemicals on the EPA's list are used in <1% of wells nationally (Figure 9-4). Therefore, potential exposure to the majority of these chemicals is more likely to be a local issue, rather than a national one. Given that the analysis of the EPA FracFocus 1.0 project database presented in this chapter was based on 35,957 disclosures, a chemical used in <1% of wells nationally could still be used in several hundred wells. Chemicals used infrequently on a national basis could still be used more frequently within certain areas or counties, increasing the potential for local exposure to that chemical.

As an example of how an infrequently used chemical could have local impacts, consider (E)-crotonaldehyde, which had one of the lowest chronic oral RfVs among the chemicals considered in the noncancer MCDA for hydraulic fracturing chemicals, and was reported in approximately 0.06% of disclosures in the EPA FracFocus 1.0 project database. If the EPA FracFocus 1.0 project database is a representative sample of all of the wells across the country, then the likelihood of (E)-crotonaldehyde contamination on a national scale is limited. However, this in no way diminishes the likelihood or potential severity of (E)-crotonaldehyde contamination at sites where this chemical is used.

This is in contrast with frequently used chemicals such as methanol. Methanol was reported in 73% of wells in the EPA FracFocus 1.0 project database, and was the most frequently used chemical considered in the noncancer MCDA for chemicals used in hydraulic fracturing fluids. Methanol is soluble and relatively mobile in water, but has a higher chronic oral RfV compared other chemicals considered in this analysis. Therefore, methanol may be expected to have a higher exposure potential on a national basis compared to other chemicals, with a moderate hazard potential due to its relatively high RfV.

Even if no chemicals were added to hydraulic fracturing fluids, there is still a potential for impacts from constituents naturally present in the subsurface which could be brought to the surface in produced water. As described in Section 9.5, many of the naturally occurring chemicals in produced water—e.g., organic chemicals (e.g., BTEX and related hydrocarbons), metals, anions, and TENORM—are hazardous to human health and have been reported in drinking water resources as a result of hydraulic fracturing activity, sometimes at concentrations exceeding MCLs. The constituents of produced water that contribute to the formation of DBPs, specifically bromide, chloride, iodine, and ammonium, are naturally occurring and are characteristic of wastewater from hydraulically fractured wells.

Overall, contamination of drinking water resources depends on site-, chemical-, and fluid-specific factors (Goldstein et al., 2014), and the exact mixture and concentrations of chemicals at a site will depend upon the geology and the chemicals used in the oil and gas extraction processes. Therefore, potential hazard and risk considerations are best made on a site-specific, well-specific basis.

9.7.3 Uncertainties

There are notable uncertainties in the chemical and toxicological data limiting a comprehensive assessment of the potential health impacts of hydraulic fracturing on drinking water resources.

For human health risk assessment, a significant data gap is the lack of chronic oral RfVs and OSFs from sources meeting the EPA's criteria for inclusion in this report. For instance, of the 34 chemicals (excluding water, quartz, and sodium chloride) that were reported in ≥10% of disclosures in the EPA FracFocus 1.0 project database, 9 chemicals have chronic oral RfVs available, and none have OSFs (Table 9-2). Without reliable and peer reviewed toxicity values, comprehensive hazard evaluation and hazard identification of chemicals is difficult, and the ability to consider the potential cumulative effects of exposure to chemical mixtures in hydraulic fracturing fluid or produced water may be limited. Although there are other potential sources of toxicity information for many of these chemicals, some of it may be limited or of lesser quality. Consequently, potential impacts on drinking water resources and human health may not be assessed adequately.

An equally significant data gap is the lack of exposure assessment data for drinking water resources in areas of hydraulic fracturing activity. As discussed in Text Box 9-1, data on exposure potential is a critical component of the risk assessment process, and is necessary for risk characterization. In the absence of exposure assessment information, the MCDA framework presented in this chapter may be useful for exploring the potential hazards of hydraulic fracturing-related chemicals, but should be considered as a preliminary analysis only. The MCDA presented in this chapter considered only a small subset of chemicals that had data available, was limited in scope, and may not be representative of the chemicals that are present at a specific field site. It should be emphasized that this MCDA framework represents just one method that can be used to integrate chemical data for hazard evaluation, and is readily adaptable to include different variables, different weights for the variables, and site-specific considerations.

There is also uncertainty surrounding the EPA's list of chemicals associated with hydraulic fracturing activity. As discussed in Section 5.4 and Section 9.3.1, there is incomplete information available on chemicals used in hydraulic fracturing fluids due to industry use of CBI as well as incomplete reporting of chemical use. For instance, the EPA's analysis of the FracFocus 1.0 project database found that approximately 11% of ingredients were reported as CBI, and that more than 70% of FracFocus 1.0 disclosures contained at least one CBI ingredient. There may also be regional limitations in the disclosures submitted to FracFocus 1.0, as 78% of chemical disclosures in came from five states, and 47% were from Texas (U.S. EPA, 2015a). Despite these limitations, FracFocus remains the most complete source for tracking hydraulic fracturing chemical usage in the United States, and therefore was the best available source for the hazard evaluation in this chapter. Although the sources used to compile the chemical list represented the best available data at the time of this study, it is possible that some of these chemicals are no longer used at all, and many of these chemicals may only be used infrequently. Therefore, it may be possible that significantly fewer than 1,084 chemicals are currently used in abundance. As practices evolve, it is likely that chemicals are used or will be used that are not included on this chemical list. Having a better understanding of the chemicals and formulations, including those that are CBI, along with their frequency of use and volumes, would greatly benefit risk assessment and risk management decisions.

Additionally, the list of produced water chemicals identified in this chapter is almost certainly incomplete. As discussed in Chapter 7, chemicals and their metabolites may go undetected because they were not included in the analytical methodology, or because an analytical methodology was not available. Chemical analysis of produced water can also be challenging because high levels of dissolved solids in produced water and wastewater can interfere with chemical detection. As a result, there are likely chemicals of concern in produced water that have not been detected or reported, and are not included on the chemical list presented in this report.

9.7.4 Conclusions

The EPA identified 1,606 chemicals associated with the hydraulic fracturing water cycle, including 1,084 chemicals used in hydraulic fracturing fluids, and 599 chemicals detected in produced water. Toxicity-based chronic oral RfVs and/or OSFs from sources meeting selection criteria were not available for the majority (89%) of the chemicals on this total list. Thirty-seven percent of chemicals on the EPA's list that are used in hydraulic fracturing fluids lack data on their frequency of use. Current understanding of the chemical composition of produced water is constrained by analytical chemistry limitations and by the likelihood that chemical composition will vary between wells. A limited number of studies have detected these chemicals in surface water, groundwater, or well water near areas of hydraulic fracturing activity, suggesting the potential for human exposure; however, actual human exposures to these chemicals in drinking water resources has not been well characterized. Given the large number of chemicals used or detected in various stages of the hydraulic fracturing water cycle, as well as the large number of hydraulically fractured wells nationwide, this missing chemical information represents a significant data gap.

While it remains challenging to fully understand the toxicity and potential public health impacts of these chemicals for drinking water resources, the toxicological data, occurrence data, and physicochemical data compiled in this report provide a resource for assessing the potential hazards of chemicals in the hydraulic fracturing water cycle. The MCDA framework presented here illustrates one method for integrating these data for a preliminary hazard evaluation, which may be useful when exposure assessment data are not available. While the analysis in this chapter is constrained to the assessment of chemicals on a national scale, this approach is readily adaptable for use on a regional or site-specific basis.

This collection of data provides a tool to inform decisions about protection of drinking water resources. Stakeholders may use these results to prioritize chemicals for hazard assessment or for determining future research priorities. Industry may use this information to prioritize chemicals for replacement with less toxic, persistent, and mobile alternatives.

9.8 Annex

9.8.1 Calculation of Physicochemical Property Scores (MCDA Hazard Evaluation)

Section 9.6.3 describes how Physicochemical Properties Scores for the noncancer and cancer MCDAs were calculated based on three subcriteria which affect the likelihood that a chemical will be transported in water: mobility, volatility, and persistence. Calculation of these subcriteria scores was performed as described by Yost et al. (In Press), as follows:

9.8.1.1 Mobility Score

Chemical mobility in water was assessed based upon three physicochemical properties that describe chemical solvency in water: the octanol-water partition coefficient (K_{OW}), the soil adsorption coefficient (K_{OC}), and aqueous solubility. K_{OW} describes the partitioning of a chemical between water and a carbon-based media (octanol), while K_{OC} described the partitioning of a chemical between water and organic carbon in soil. K_{OW} and K_{OC} are generally represented on a logarithmic scale. Aqueous solubility is the maximum amount of a chemical that will dissolve in water in the presence of pure chemical. Chemicals with low K_{OW}, low K_{OC}, or high aqueous solubility are more likely to solubilize and move with water, and therefore were ranked as having greater potential to affect drinking water resources.

For input into the MCDA, we used experimentally measured values (provided in EPI Suite) whenever available. Otherwise, we used the following estimated values from EPI Suite: log K_{OW} estimated using the KOWWIN™ model, log K_{OC} estimated using the KOCWIN™ Sabljic molecular connectivity method, and aqueous solubility estimated using the WSKOWWIN™ model. Using the thresholds designated in Table 9-11, each of these properties was assigned a score of 1-4. The highest of these three scores (K_{OW}, K_{OC}, or solubility) was designated as the Mobility Score for each chemical.

9.8.1.2 Volatility Score

Chemical volatility was assessed based on the Henry's law constant, which is the ratio of the concentration of a chemical in air to the concentration of that chemical in water. Chemicals with low Henry's law constants are less likely to leave water via volatilization, and were therefore ranked as having greater potential to affect drinking water resources.

For input into the MCDA, we used experimentally measured values (provided in EPI Suite) whenever available. Otherwise, we used Henry's Law constants that were estimated using the EPI Suite HENRYWIN™ model, which generates values using two different methods (group contribution and bond contribution); the lower of these two estimated values was used as input into the MCDA.

Using the thresholds designated in Table 9-11, the Henry's law constant for each chemical was assigned a score of 1-4. This value was designated as the Volatility Score for each chemical.

9.8.1.3 Persistence Score

Chemical persistence was assessed based on estimated half-life in water, which describes how long a chemical will persist in water before it is transformed or degraded. Chemicals with longer half-lives are more persistent, and were therefore ranked as having greater potential to impact drinking water resources.

EPI Suite estimates biodegradation time using the BIOWIN™ 3 model, which provides an indication of a chemical's environmental biodegradation rate in relative terms (e.g., hours, days, weeks, etc.), assuming aerobic conditions. These BIOWIN3 estimates are converted to numerical half-life values for use in EPI Suite's Level III Fugacity model. For input into the MCDA, we used the same estimated half-life in water that is used in the Level III Fugacity model. Using the thresholds designated in Table 9-11, the half-life in water of each chemical was assigned a score of 1-4. This value was designated as the Persistence Score for each chemical.

9.8.1.4 Total Physicochemical Properties Score

For each chemical, the Mobility Score, Volatility Score, and Persistence Score (each on a scale of 1 to 4) were summed to calculate a total Physicochemical Properties Score. Higher Physicochemical Properties Scores indicate chemicals that are more likely to be transported in water, with a maximum possible score of 12.

9.8.2 Example of MCDA Score Calculation

The methods used for MCDA score calculation are described in Section 9.6.3. For an example of how the MCDA scores were calculated, consider benzene, which was included in both the noncancer MCDA (national analysis) and cancer MCDA for chemicals used in hydraulic fracturing fluids. This demonstrates how MCDA scores were calculated for benzene for these two different analyses.

9.8.2.1 Score Calculation for Benzene in Noncancer MCDA for Hydraulic Fracturing Fluids

- **Toxicity Score (Noncancer):** Benzene has a chronic oral RfV of 0.004 mg/kg-day (source: IRIS). Across the 42 chemicals that were considered in the noncancer MCDA (national analysis), chronic oral RfVs ranged from 0.001 mg/kg-day [(E)-crotonaldehyde] to 20 mg/kg-day (1,2-propylene glycol). The chronic oral RfV of benzene falls in the lowest (most toxic) quartile of these chemicals, and therefore benzene was assigned a Toxicity Score of 4. When the results were standardized to the highest Toxicity Score (4) and lowest Toxicity Score (1) within the set of chemicals, benzene was calculated to have a final Toxicity Score of 1, as follows:

$$1 = (4 - 1) / (4 - 1)$$

- **Occurrence Score:** Benzene was used in 0.006% of wells nationally. For the 42 chemicals considered in the national noncancer MCDA, frequency of use ranged from 73%

(methanol) to 0.003% (furfural) of wells nationally. Benzene falls in the lowest quartile with regards to frequency of use, and therefore benzene was assigned an Occurrence Score of 1. When the results were standardized to the highest Occurrence Score (4) and lowest Occurrence Score (1) within the set of chemicals, benzene was calculated to have a final Occurrence Score of 0, as follows:

$$0 = (1 – 1) / (4 – 1)$$

- **Physiochemical Properties Score:** Benzene received a Mobility Score of 4 (log K_{OW} = 2.13; log K_{OC} = 1.75; solubility = 2000 mg/l), a Volatility Score of 2 (Henry's law constant = 0.00555), and a Persistence Score of 2 (half-life in water = 37.5 days). This sums to a Total Physicochemical Properties Score of 8. Within the 42 chemicals considered in the national noncancer MCDA, several chemicals received Total Physicochemical Properties Scores of 9, which was the highest observed score. Cumene received a Total Physicochemical Properties Scores of 6, which was the lowest score. When the results were standardized to the highest (9) and lowest (6) of these scores, benzene was calculated to have a final Total Physicochemical Properties Scores of 0.67, as follows:

$$0.67 = (8 – 6) / (9 – 6)$$

- **Total Hazard Potential Score (Noncancer MCDA):** For benzene, the Toxicity Score (1), Occurrence Score (0), and Physicochemical Properties Score (0.67) were summed to calculate a Total Hazard Potential Score of 1.67. The relative contribution of the three criteria scores to this total score is depicted as a graphic in Figure 9-8.

9.8.2.2 Score Calculation for Benzene in Cancer MCDA for Hydraulic Fracturing Fluids

- **Toxicity Score (Cancer):** Benzene has an OSF of 0.055 per mg/kg-day (source: IRIS). Within the entire set of 10 chemicals that was considered in the cancer MCDA, OSFs ranged from 3 (quinoline) to 0.002 (dichloromethane) per mg/kg-day. The OSF of benzene falls in the second quartile of these scores, and therefore was assigned a Toxicity Score of 2. When the results were standardized to the highest Toxicity Score (4) and lowest Toxicity Score (1) within the set of chemicals, benzene was calculated to have a final Toxicity Score of 0.33, as follows:

$$0.33 = (2 – 1) / (4 – 1)$$

- **Occurrence Score:** As described in the noncancer MCDA above, benzene was used in 0.006% of wells nationally. This was the lowest frequency of use among the 10 chemicals that were considered in the cancer MCDA, with benzyl chloride (used in 6% of wells) having the highest. Benzene therefore falls in the lowest quartile with regards to frequency of use, and was assigned an Occurrence Score of 1. When the results were standardized to the highest Occurrence Score (4) and lowest Occurrence Score (1) within the set of chemicals, benzene was calculated to have a final Occurrence Score of 0, as follows:

$$0 = (1 - 1) / (4 - 1)$$

- **Physiochemical Properties Score:** As described in the noncancer MCDA above, benzene received a Total Physicochemical Properties Score of 8. Within the 10 chemicals that were considered in the cancer MCDA, all chemicals either received a Total Physicochemical Properties Score of 8 or 9. When the results were standardized to these high and low scores, benzene was calculated to have a final Total Physicochemical Properties Scores of 0 as follows:

$$0 = (8 - 8) / (9 - 8)$$

- **Total Hazard Potential Score (Cancer MCDA):** The Toxicity Score (0.33), Occurrence Score (0), and Physicochemical Properties Score (0) were summed to calculate a Total Hazard Potential Score of 0.33. The relative contribution of the three criteria scores to this total score is depicted as a graphic in Figure 9-12.

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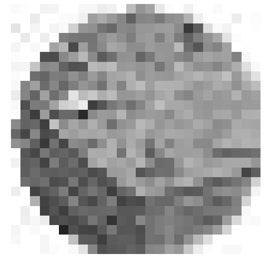
Chapter 10. Synthesis



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10. Synthesis

Introduction

The goals of this report were to assess the potential for activities in the hydraulic fracturing water cycle to impact the quality or quantity of drinking water resources, and to identify factors affecting the frequency or severity of those impacts. Overall, we conclude activities in the hydraulic fracturing water cycle can impact drinking water resources under some circumstances. Impacts can range in frequency and severity, depending on the combination of hydraulic fracturing water cycle activities and local- or regional-scale factors. The following combinations of activities and factors are more likely than others to result in more frequent or more severe impacts:

- Water withdrawals for hydraulic fracturing in times or areas of low water availability, particularly in areas with limited or declining groundwater resources;
- Spills during the management of hydraulic fracturing fluids and chemicals or produced water that result in large volumes or high concentrations of chemicals reaching groundwater resources;
- Injection of hydraulic fracturing fluids into wells with inadequate mechanical integrity, allowing gases or liquids to move to groundwater resources;
- Injection of hydraulic fracturing fluids directly into groundwater resources;
- Discharge of inadequately treated hydraulic fracturing wastewater to surface water resources; and
- Disposal or storage of hydraulic fracturing wastewater in unlined pits, resulting in contamination of groundwater resources.

These conclusions are based on cases of identified impacts and other data, information, and analyses presented in this report. Cases of impacts were identified for all stages of the hydraulic fracturing water cycle. Identified impacts generally occurred near hydraulically fractured oil and gas production wells and ranged in severity, from temporary changes in water quality to contamination making private drinking water wells unusable. The inherent characteristics of groundwater resources make them more vulnerable to impacts from activities in the hydraulic fracturing water cycle compared to surface water.

We see the identification of factors affecting the frequency or severity of impacts, and uncertainties and data gaps in this report as particularly useful for decision makers. Factors often can be managed, changed, or used to identify areas for specific monitoring or modification of practices. Thus, in the short-term, information on factors can help decision makers reduce current vulnerabilities of drinking water resources to activities in the hydraulic fracturing water cycle. In the longer term, reducing the uncertainties and filling the data gaps could enhance science-based decisions to protect drinking water resources in the future.

The purpose of this chapter is to synthesize for decision makers the information on factors, uncertainties, and data gaps presented in this assessment. In Section 10.2, we focus on factors increasing or decreasing the frequency or severity of impacts at each stage of the hydraulic fracturing water cycle. In Section 10. 3, we discuss major uncertainties and data gaps identified in this assessment. Finally, in Section 10.4, we discuss potential uses for this assessment.

10.1 Factors Affecting the Frequency or Severity of Impacts

10.1.1 Water Acquisition

Groundwater and surface water resources serve as both sources of water for hydraulic fracturing and public and private drinking water supplies. Thus, water withdrawals for hydraulic fracturing can impact the quantity or quality of drinking water resources under certain circumstances. Since, by definition, every water withdrawal affects water quantity, we focused in this assessment not on all water withdrawals per se, but rather on those with the potential to limit the availability of drinking water or alter its quality. Whether a withdrawal has this potential depends upon a combination of factors at the local scale. Factors can either increase or decrease the frequency or severity of impacts. In this section on water acquisition, we combine our discussion of frequency and severity because all of the factors we discuss in this section affect both frequency and severity in a similar fashion (i.e., either increase both frequency and severity, or decrease both frequency and severity).

10.1.1.1 Frequency and Severity

The local balance between water withdrawals and water availability is the most important factor determining whether water acquisition impacts are likely to occur or be severe. Impacts are more likely to be frequent or severe where or when hydraulic fracturing water withdrawals are relatively high and water availability is low. In contrast, the same amount of water withdrawn can have a negligible effect if withdrawn in an area of—or at a time of—higher water availability. For this reason, it is important not to focus solely on the amount withdrawn, but the balance between water withdrawals and availability in place and time.

For this assessment, we developed county-level estimates of water use (i.e., water withdrawals) for hydraulic fracturing, which were then compared to an index of readily available fresh water. This readily available fresh water index included unappropriated surface water and groundwater, and appropriated water potentially available for purchase (Tidwell et al., 2013) (Text Box 4-2).[1] In the majority of counties where hydraulic fracturing takes place, hydraulic fracturing water use was less than 1% of this index of readily available fresh water. We did find, however, a small number of counties with higher percentages. There were 45 counties out of the almost 400 surveyed where hydraulic fracturing water use was above 10% of the index. Of these counties, 35 exceeded 30%, and 17 of these counties had hydraulic fracturing water use exceeding the index. All of the counties in this latter category are located in Texas.

[1] In the western United States, water is generally allocated by the principle of prior appropriation—that is, first in time of use is first in right. New development must use unappropriated water or purchase appropriated water from vested users. In the index of readily available fresh water, it was assumed 5% of appropriated irrigated water could be purchased. See Text Box 4-2 for more details about this analysis.

This does not mean impacts to drinking water quantities occurred or will occur in these counties, nor does it mean that impacts did not or will not occur in counties with relatively low percentages. To truly determine whether impacts occurred, water withdrawals and availability need to be compared at the scale of the drinking water resource. For instance, groundwater withdrawals for hydraulic fracturing could affect water levels in nearby private water wells. As a national assessment, we could not often examine impacts at this local scale, although we did cite studies of local impacts where available. Nevertheless, our county level assessment does point to places where the potential for impacts is higher. This information may be useful to focus efforts on reducing the fresh water demand of hydraulic fracturing.

Beyond our county level assessment, we conclude that declining groundwater resources are particularly vulnerable to water quantity and quality impacts from withdrawals. Groundwater recharge rates can be low, and groundwater withdrawals are exceeding recharge in areas of the country (Konikow, 2013). When withdrawals exceed recharge, the result is declining water levels. For this reason, water levels in some aquifers in the United States have declined substantially over the last century (Konikow, 2013). Although irrigated agriculture is often the dominant user of groundwater, hydraulic fracturing withdrawals now also contribute to declining groundwater levels in some areas (e.g., southern Texas; Steadman et al., 2015; Scanlon et al., 2014b) Cumulative groundwater withdrawals can also impact water quality by mobilizing chemicals, such as uranium, from naturally occurring sources in the surrounding rock into the groundwater (DeSimone et al., 2014).

In certain instances, state and local governments have encouraged or mandated the use of surface water in place of groundwater, as evidenced in both Louisiana and North Dakota. In 2008, the state of Louisiana asked oil and gas companies to switch from groundwater to surface water to mitigate stress on the Carrizo-Wilcox aquifer, a critical source of drinking water in the region. Likewise, the state of North Dakota requested the oil industry obtain water from the Missouri river system, and not from stressed groundwater sources. By contrast, surface water availability is limited in other regions and cannot provide an alternative source of water (e.g., western Texas).

Among surface water sources, small streams are particularly vulnerable to impacts. This is the case across the country, even in the eastern United States where surface water is generally more plentiful. An EPA study of the Susquehanna River Basin in northeastern Pennsylvania found that the smallest streams (with less than 10 mi^2 of contributing area–i.e., the watershed area drained by the stream) would be the most likely to be impacted from water withdrawals in the absence of protective passby flows; see discussion below and U.S. EPA (2015e).[1] While the amount of contributing area varies by geographic location due to differences in runoff, the finding that the smallest streams are the most vulnerable to withdrawals holds across all landscapes.

Not only does water availability vary from one location to another, but it can also vary temporally at a given location, often due to variations in precipitation. Because of this dynamic, long-term or seasonal drought can increase the frequency or severity of impacts from withdrawals by decreasing water availability. The EPA study of the Susquehanna River Basin found even larger streams (up to

[1] Passby flows are low stream flow thresholds below which withdrawals are not allowed.

600 mi^2 of contributing area) would be vulnerable to impacts at times of drought, again absent passby flows (U.S. EPA, 2015e). Dry conditions can also stress groundwater supplies by simultaneously increasing water demand (e.g., irrigation water demand increases in dry conditions) while also decreasing groundwater recharge. Much of the western United States has experienced extended periods of drought over the last decade. Climate change is likely to exacerbate these conditions in certain locations (Meixner et al., 2016).

Conversely, there are factors that can reduce the frequency or severity of impacts. Reuse of hydraulic fracturing wastewater (i.e., produced water managed for reuse, treatment and discharge, or disposal), for example, can reduce demands on fresh water resources.[1] Reuse does not appear to be driven by water scarcity, but rather by the cost of disposal. Operators are likely to dispose of wastewater when it is less expensive than reuse. For instance, greater reuse of wastewater occurs in the Marcellus Shale in Pennsylvania than in the Barnett Shale in Texas, even though water availability is generally higher in the Marcellus region (Figure 10-1). The general lack of disposal wells in Pennsylvania means disposing of wastewater requires trucking to Ohio or other locations with disposal wells. Because of this expense, operators reuse substantial proportions of their wastewater, in contrast to the Barnett Shale where disposal wells are readily available.

The reuse of wastewater to offset fresh water use in hydraulic fracturing is often limited by the amount of wastewater available. The volume of produced water from a single well can be relatively small compared to the volume needed to fracture a well (Figure 10-1a). This means produced water would need to be aggregated from multiple wells to equal the volume needed to hydraulically fracture an additional well. For instance, it would take 10 wells to make enough water to fracture an 11th well if, as has been shown in the Marcellus Shale in Pennsylvania, produced water volumes are 10% of injected volumes (Figure 10-1a). Thus, reuse is a factor that can reduce fresh water demand, but not eliminate it in most cases. Nevertheless, even a marginal decline in fresh water demand can make a difference in the frequency or severity of impacts.

The use of brackish groundwater is also a factor reducing fresh water demand, in some cases to a much greater degree than reuse. In the Permian Basin in western Texas, for instance, brackish water makes up 30 to 80% of water used for hydraulic fracturing, and 20% in the Eagle Ford Shale in southern Texas (Nicot et al., 2012). Our county level estimates suggest brackish water availability could entirely meet current hydraulic fracturing water demand in Texas and many other locations.[2] In 35 counties nationally, hydraulic fracturing water use equaled or exceeded 30% of an index of fresh water availability; when brackish water and wastewater were considered in addition to fresh water availability, only two counties equaled or exceeded 30% (Text Box 4-2).

[1] Hydraulic fracturing wastewater is produced water that is managed using practices that include, but are not limited to, reuse in subsequent hydraulic fracturing operations, treatment and discharge, and injection into disposal wells. The term is being used in this study as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes (see Chapter 8 and Appendix J, the Glossary, for more detail).

[2] Brackish water for the purposes of this analysis ranged from 3,000 to 10,000 ppm of total dissolved solids (TDS), and from 50 to 2,500 ft (15-760 m) below the surface (Tidwell et al., 2013). (See Text Box 4-2 for more details.)

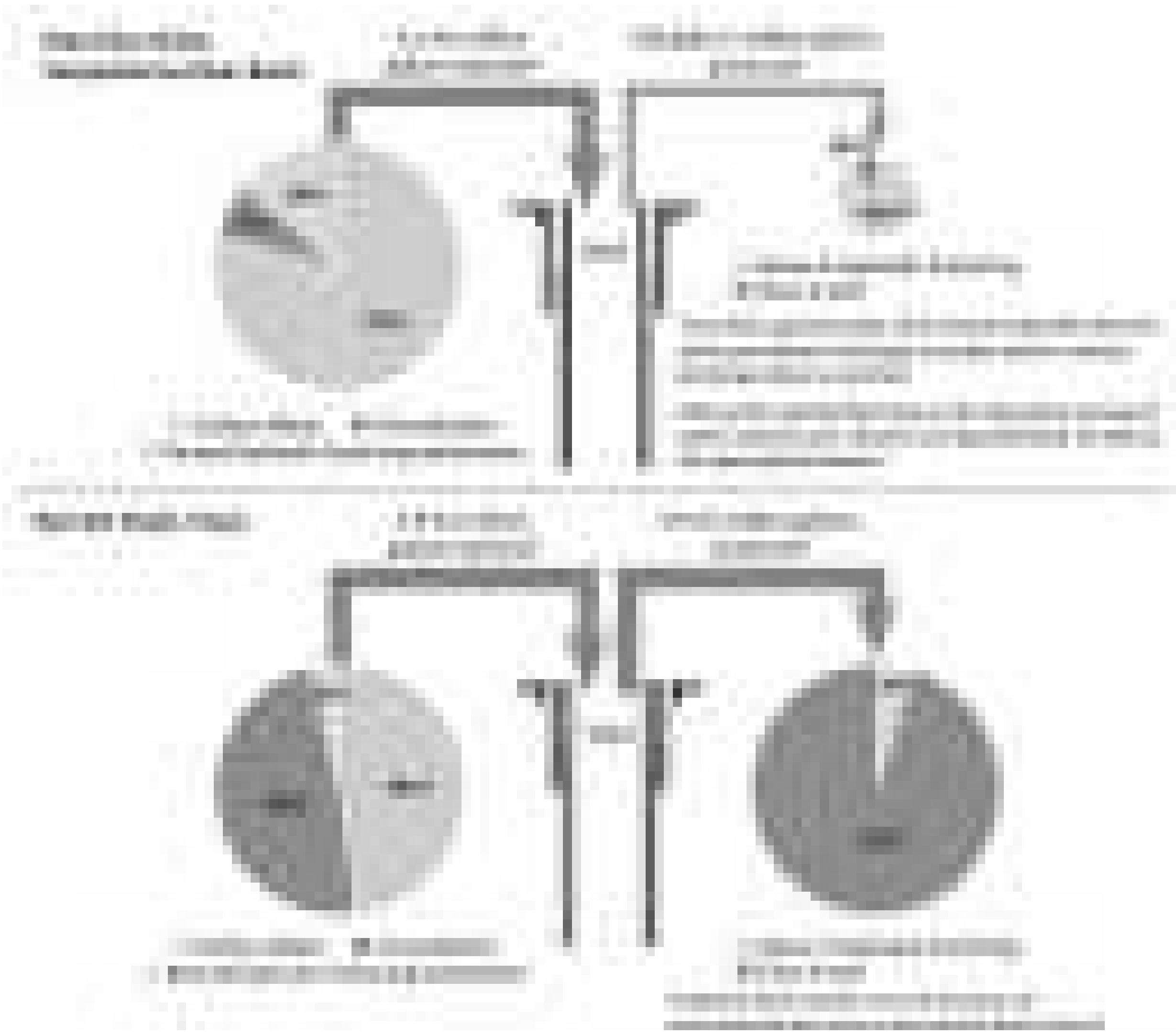

Figure 10-1. Water budgets representative of practices in (top) the Marcellus Shale in the Susquehanna River Basin in Pennsylvania and (bottom) the Barnett Shale in Texas.

Pie size and arrow thickness represent the relative volume of water as it flows through the hydraulic fracturing water cycle. Water budgets illustrative of typical water management practices in the Marcellus Shale in the Susquehanna River Basin between approximately 2008 and 2013 and the Barnett Shale in Texas between approximately 2011 and 2013. They do not represent any specific well. Sources for the top figure (a) Tables 4-1 and 4-2 (SRBC, 2016)—note, surface water, groundwater, and reuse values of 92%, 8%, and 16% in table normalized to 79%, 7%, 14%, respectively, for this chart (this was done to represent reuse on the same chart as surface water and groundwater—in the original tabular values, reuse is expressed as a percentage of total water used, and surface water and groundwater are expressed in percentages relative to each other); (b) Appendix Table B-5 (U.S. EPA, 2015a); (c) Table 7-2 (Ziemkiewicz et al., 2014)—note: produced water volumes estimated from percentages applied to volumes injected, and value from the West Virginia portion of the Marcellus Shale used in this chart since it was the longest term measurement of produced water volumes; (d) Figure 8-4 (PA DEP, 2015a) and Table 8-6 (Ma et al., 2014; Shaffer et al., 2013). Sources for the bottom figure: (e) Tables 4-1 and 4-2 (Nicot et al., 2014; Nicot et al., 2012)—note, surface water, groundwater, and reuse values of 50%, 50%, and 5% in the tables normalized to 48%, 48%, and 4%, respectively, for this chart (see reason for this above); (f) Appendix Table B-5 (U.S. EPA, 2015a; Nicot et al., 2012; Nicot et al., 2011)—note: see median value for Fort Worth Basin; (g) Table 7-2 (Nicot et al., 2014); (h) Table 8-6 (Nicot et al., 2012)—note, percentage going to disposal wells estimated by subtracting reuse values from 100%.

Finally, passby flows can be a factor reducing the frequency or severity of surface water impacts. Passby flows are low stream flow thresholds below which withdrawals are not allowed. This management practice has been shown to be protective of streams from over-withdrawals in the Susquehanna River Basin in northern Pennsylvania (U.S. EPA, 2015e). This is likely most important for protecting aquatic life in smaller streams, but may also aid in protecting drinking water supplies.

10.1.2 Chemical Mixing and Produced Water Handling

Like water acquisition, activities in the chemical mixing and produced water handling stages of the hydraulic fracturing water cycle can impact drinking water in some instances. We combine our discussion of the two stages here because activities in these stages both affect drinking water resources primarily through spills. The chemical mixing stage encompasses management of fluids on the well pad to create hydraulic fracturing fluid. Chemicals are mixed with a base fluid, typically water, and then injected into the production well. After the pressure is released post-fracturing, produced water flows from the well and needs to be collected and managed in the produced water handling stage.

Chemical mixing and produced water handling activities can impact drinking water resources through spills of chemicals used to make hydraulic fracturing fluid, hydraulic fracturing fluid itself, or produced water reaching surface water or groundwater.[1] There is some information on spill frequencies—although limited—and spill severities are most often uncharacterized. Nevertheless, we could identify factors affecting the frequency or severity of impacts from chemical mixing or produced water spills. In the section below, we discuss these factors, with those affecting frequency first, followed by those affecting severity. We discuss each of the factors individually, but spill events in reality exhibit combinations of these factors. These factors can interact to increase or decrease the frequency or severity of a spill beyond the effect of an individual factor.

10.1.2.1 Frequency

An impact on the quality of a drinking water resource from a spill first depends on a spill occurring. Most spill frequency estimates are of spills in total, and not the subset reaching drinking water resources. Spill estimates from three states (Colorado, North Dakota, and Pennsylvania) ranged from 0.4 to 12.2 reported spills per 100 hydraulically fractured wells (Appendix C.4).[2] The estimates from Pennsylvania and Colorado included hydraulic fracturing chemicals, fluids, and produced water; while the North Dakota estimate was based on spills of hydraulic fracturing chemicals and fluids only.[3] Spill rates can also be expressed on a per-active-well basis. This may be

[1] In Chapter 5 and elsewhere in this assessment, the chemicals added to the base fluid (most often water) and proppant (most often sand) are referred to as "additives" since this is the term used in FracFocus. Here, this chapter simply refers to them as "chemicals." It does this to discuss chemicals in a unified manner in this combined section on chemical mixing and produced water.

[2] Since most wells are not reported hydraulically fractured in databases, these estimates used spudded, completed, or installed wells as proxies for hydraulically fractured wells. (See Appendix Section C.4 for more detail.)

[3] These estimates from Pennsylvania and Colorado also included spills of diesel fuel and drilling muds, which could not be separated out from the total frequency estimate even though they were generally out-of-scope of this assessment (diesel fuel was in scope if used in hydraulic fracturing fluid).

more appropriate for produced water spills since they can occur years or even decades after hydraulic fracturing. An analysis of North Dakota produced water spills found there were approximately 5 to 7 spills of produced water per 100 active wells between 2010 and 2015 (Appendix E.5). We conclude from these data that spills do occur in both the chemical mixing and produced water stages of the hydraulic fracturing water cycle, generally in the range of 1 to 10% of hydraulically fractured or active wells.

Not all spills, however, reach and therefore impact a drinking water resource. In U.S. EPA (2015m), 32 of the 457 (7%) spills characterized were reported to have reached surface water or groundwater. The California Office of Emergency Services estimated 18% of produced water spills reached waterways between January 2009 and December 2014 (CCST, 2015b). It is unclear if this estimate included groundwater, or was limited to surface water. If, however, roughly 5 to 20% of spills reach surface water or groundwater (encompassing the U.S. EPA and California estimates above), we would expect a spill to occur and reach a drinking water resource at approximately 0.05 to 2% of active or hydraulically fractured wells.[1] This estimate of spills reaching drinking water resources would be broadly consistent with estimates from the limited number of published studies addressing this topic (e.g., Brantley et al., 2014; Gross et al., 2013).[2] If a 0.05 to 2% frequency rate is applied to the estimates of approximately 275,000 to 370,000 new wells hydraulically fractured nationally between 2000 and part of 2013 and 2000 and part of 2014, respectively (Chapter 3), we would expect roughly 140 to 7,400 spills to reach a drinking water resource during this almost 14-to-15 year time-period. This would be approximately 10 to 500 spills per year reaching a drinking water resource, dividing by the respective time periods. This large range reflects the high uncertainty of these estimates and the lack of data on this topic.

Despite the data limitations and uncertainties surrounding estimates of spills, we can with more certainty identify factors likely affecting the frequency of spills reaching drinking water resources. These factors include spill characteristics, encompassing the volume of the chemical spilled; factors related to the environmental fate and transport of the spill, such as properties of the chemical spilled and characteristics of the site where the spill occurred; and finally, factors related to spill prevention and response.

Everything else being equal, a larger volume spill will be more likely to reach a drinking water resource than a smaller spill (U.S. EPA, 2015m). On-site spills in the chemical mixing and produced water handling stages are typically in the hundreds of gallons (U.S. EPA, 2015m). Larger spills, though less common, do occur. Well blowouts, pipeline leaks, and impoundment failures are sources of some of the largest individual spill volumes. Well blowouts were responsible for the

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[1] Estimated by multiplying the 1 to 10% spill rate for active or hydraulically fractured wells by 5% to 20% for spills reaching drinking water, and then reconverting to a percentage by multiplying by 100.

[2] Brantley et al. (2014) estimated approximately 0.4 to 0.8 spills per 100 hydraulically fractured wells reached surface water in Pennsylvania between 2008 to September 2013. These were spills of 400 gal (1,514 L) or more, containing hydraulic fracturing chemicals, fluids, or produced water. This might be an underestimate of spills reaching surface water since spill volumes were limited to only 400 gal (1,514 L) or more. In estimate of the frequency of spills reaching groundwater, Gross et al. (2013) examined oil and produced water spills between July 2010 and July 2011 in Weld County, Colorado. They counted 77 such spills reaching groundwater, approximately 0.4% of the nearly 18,000 active wells in the county.

highest volume spills on average in 2015 in North Dakota. In Bradford County, Pennsylvania, a well blowout resulted in a spill of approximately 10,000 gal (38,000 L) of produced water into a tributary of Towanda Creek, a state designated trout fishery. The largest volume spill identified in this assessment occurred in North Dakota, where approximately 2.9 million gal (11 million L) of produced water spilled from a broken pipeline and impacted surface water and groundwater. Though relatively rare compared to smaller volume spills, these types of spills are more likely to reach—and therefore impact—a drinking water resource because they are of larger volumes.

By this same principle, produced water spills are more likely to impact drinking water resources than chemical mixing spills. In an analysis of on-site spills, the median volume of produced water spills was approximately twice as large as that in the chemical mixing stage (990 versus 420 gal, or 3,750 versus 1,590 L; U.S. EPA (2015m)). Additionally, offsite, large pipeline spills of produced water can occur. It is possible that spills of produced water are larger, in part, because they are less likely to be stopped as quickly as spills in the chemical mixing stage. Spills in the chemical mixing stage are likely to occur when people are on-site, and so the spills can be quickly addressed. In contrast, spills of produced water may occur when no one is on-site or, in the case of pipelines, near the off-site location of the spill. This may delay a response, allowing larger volumes to spill, increasing the likelihood of the spill reaching a drinking water resource.

Properties of the chemicals spilled also affect the frequency of impacts. We identified or estimated chemical and physical properties for almost half of the chemicals used in hydraulic fracturing fluids between 2006 and 2013 (455 of the 1,084 chemicals). These were individual organic chemicals, not inorganic chemicals, polymers, or mixtures. Volatility, solubility, and hydrophobicity/hydrophilicity are three properties, among others, affecting whether a spill reaches a drinking water resource (hydrophobic chemicals tend to repel or fail to mix with water, while hydrophilic chemicals tend to mix with water). The vast majority of organic chemicals in hydraulic fracturing fluid do not readily volatilize or evaporate, meaning these chemicals tend to remain in water if spilled. These chemicals also vary widely in their solubility and hydrophobicity/hydrophilicity, defying a general characterization. Nevertheless, of the 20 chemicals most frequently used according to our analysis of FracFocus, most are highly soluble and hydrophilic, meaning they will be mobile if spilled (Chapter 5). For example, methanol, isopropanol, and ethylene glycol are all likely to travel quickly through the environment. Thus, these chemicals may more frequently reach drinking water because of two unrelated, yet compounding factors: relatively high frequency of use in hydraulic fracturing operations and relatively high mobility in the environment.

Site characteristics are also an important factor determining whether a spill reaches a drinking water resource (Figure 10-2). Site characteristics facilitating infiltration to groundwater are of particular concern, since spills into groundwater are more likely to have severe impacts than those into surface water (discussed in the severity section below). More permeable, sandier soils allow greater infiltration of spilled fluids, whereas less permeable soils with more clay content can greatly slow infiltration. More permeable rock also facilitates infiltration and movement of spills through preferential flow paths—for example, in fractured or karst bedrock. Thus, sandier soils and more permeable rock can increase the potential for spills to reach groundwater.

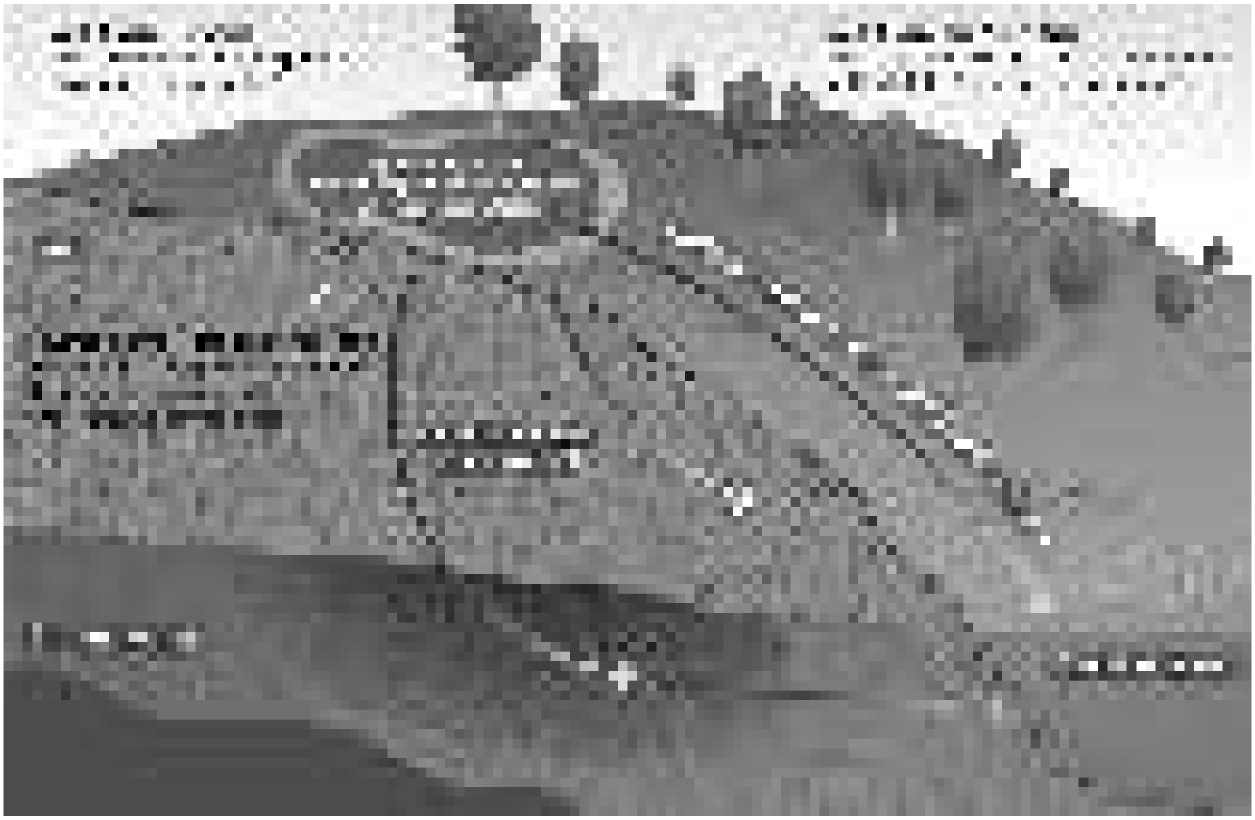

Figure 10-2. Fate and transport schematic for a spill of chemicals, hydraulic fracturing fluid, or produced water.

Schematic shows the potential paths, transport processes, and factors governing potential impacts of spills to drinking water resources.

There are spill prevention and response factors that reduce the frequency of impacts to drinking water resources from spills. Spill containment systems include primary, secondary, and emergency containment systems. Primary containment systems are the storage units, such as tanks or totes. Secondary containment systems, such as liners and berms installed during site set-up, are intended to contain spilled fluids until they can be cleaned up. Emergency containment systems, such as berms, dikes, and booms, can be implemented temporarily in response to a spill. Remediation is the action taken to clean up a spill and its affected environmental media. One of the most commonly reported remediation activities is the removal of spilled fluid and/or affected media, typically soil (U.S. EPA, 2015m). Other remediation methods include the use of absorbent material, vacuum trucks, flushing the affected area with water, and neutralizing the spilled material (U.S. EPA, 2015m). It was beyond the scope of this assessment to evaluate the implementation and efficacy of spill prevention practices and spill response activities.

10.1.2.2 Severity

In addition to frequency, there are also factors affecting the severity of an impact on a drinking water resource from a spill. For a given concentration, a larger volume spill will be more severe than a smaller spill (see frequency section above for discussion of spill volumes). In addition to volume, the concentration and toxicity of the chemicals reaching a drinking water resource affect severity, as well as site characteristics.

A spill with higher chemical concentrations will be more severe than a more dilute spill of equal volume. In the chemical mixing stage, chemicals are stored in concentrated form on-site, prior to diluting with a base fluid. Approximately 3,000 to 30,000 gal (11,000 to 114,000 L) of chemicals are used per well on average, with up to twice that amount stored on site. If multiple wells are fractured per site, tens to hundreds of thousands of gallons of chemicals are likely stored in containers at a single site during the hydraulic fracturing of these wells. These storage containers are a relatively frequent source of spills during the chemical mixing stage. Spills from these storage containers, even if low in volume, may be severe if they reach a drinking water resource because they often contain concentrated chemicals.

In the produced water handling stage, the severity of impacts from a spill also increases with higher concentrations, especially if the spill reaches groundwater (see site characteristics below). Produced water can vary substantially in chemical concentrations, including total dissolved solids (TDS), metals, radioactive isotopes, and organic chemicals. Within the Marcellus Shale, for example, produced water can range in TDS from less than 1,500 mg/L to over 300,000 mg/L (Rowan et al., 2011). By comparison, the average salinity concentration for seawater is 35,000 mg/L. The more concentrated the produced water, the more likely impacts will be severe if a spill reaches a drinking water resource. When a spilled fluid has greater concentrations of TDS than groundwater, the higher-density fluid can move downward through the groundwater resource. Depending on the flow rate and other properties of the groundwater, impacts from produced water spills can last for years.

In addition to concentration, the toxicity of chemicals affects the severity of the impact if they enter a drinking water resource. There were 37 chemicals listed in 10% or more of all FracFocus disclosures between January 1, 2011 and February 28, 2013. Of these 37 chemicals, nine had chronic oral reference values meeting the criteria used in this assessment. [1] These nine chemicals are associated with health effects including liver toxicity, kidney toxicity, developmental toxicity, reproductive toxicity, and/or carcinogenesis. Chemicals used in hydraulic fracturing fluids and detected in produced water will vary from site to site, so human health hazards are best evaluated on a site-specific basis. Nevertheless, the multi-criteria decision analysis (MCDA) presented in Chapter 9 highlighted certain chemicals that may have greater hazard potential. Propargyl alcohol, 2-butoxyethanol, and N,N-dimethylformamide are three such chemicals having relatively greater hazard potential in the MCDA based on toxicity, frequency of use in hydraulic fracturing fluids, and mobility in water.

Many of the chemicals in produced water are also known or suspected to cause cancer and/or non-cancer health effects in humans. Associated health effects include liver toxicity, kidney toxicity, neurotoxicity, reproductive and developmental toxicity, and carcinogenesis, based on the produced

[1] The analysis of toxicity presented in Chapter 9 included chemicals regardless of accompanying concentration data in FracFocus, and therefore listed 37 chemicals that were reported in 10% or more disclosures. Comparatively, Chapter 5 listed 35 chemicals that had valid concentration data from FracFocus and were reported in 10% of more disclosures.

water chemicals having chronic oral reference values meeting the criteria used in this assessment. Benzene, pyridine, and naphthalene are three of the chemicals highlighted in the MCDA as having relatively greater hazard potential based on toxicity, measured concentrations in produced water, and mobility in water.

We did not evaluate trends in chemical use by toxicity (e.g., the trends in the use of less toxic chemicals). However, a more recent study of FracFocus data evaluated disclosures dating from March 9, 2011, to April 13, 2015 (Dayalu and Konschnik, 2016; Konschnik and Dayalu, 2016). When compared to the list of 1,084 chemicals used in hydraulic fracturing operations between 2005 and 2013 compiled for this assessment (Appendix H), an additional 263 chemicals were identified (Chapter 5). Only one of these 263 chemicals was reported in more than 1% of disclosures. This comparison of chemical lists does not address potential shifts in volumes of chemicals used, but it does suggest that a shift to new types of chemicals–less toxic or otherwise–did not occur between 2013 and early 2015.

Finally, site characteristics also affect the severity of the impact. Spills into groundwater are likely to be more severe than spills into surface water, everything else being equal. This is not to say that spills into surface water cannot be severe, especially in the immediate vicinity of the spill. For instance, a tank overflowed on a well site in Kentucky spilling fluid into a nearby stream at concentrations sufficient to kill fish in the area (Papoulias and Velasco, 2013). Chemicals can also associate with stream sediments, forming a source of long-term contamination (e.g., radium). In general, however, surface water dilutes a spilled chemical much more rapidly than groundwater. Groundwater often moves slowly between areas of recharge and discharge. Groundwater movement can be as slow as one foot per year or even one foot per decade (Alley et al., 1999). Because of this dynamic, chemicals from multiple spills can accumulate over time in groundwater. Multiple chemical mixing and produced water spills, even if individually small, may impact a groundwater resource in aggregate. Additionally, groundwater contamination may not be as readily apparent as that in surface water because of the need to install monitoring wells to detect contamination in groundwater. Lastly, groundwater can be difficult and expensive to remediate, adding to the severity of impacts if spills reach groundwater (Alley et al., 1999).

10.1.3 Well Injection

Like the water acquisition, chemical mixing, and produced water handling stages, activities in the well injection stage of the hydraulic fracturing water cycle can affect drinking water resources in some instances. The well injection stage involves the injection of hydraulic fracturing fluids through the production well and into the targeted rock formation at sufficient pressure to fracture the rock. There are two fundamental pathways outlined in this assessment by which activities in the well injection stage have the potential to affect drinking water resource quality. They are: (1) fluid (meaning, liquid or gas) movement into a drinking water resource through defects or deficiencies in the production well casing and/or cement; and (2) fluid movement into a drinking water resource through the fracture network. The fluids potentially affecting drinking water resources include hydraulic fracturing fluids, hydrocarbons (including methane gas), and naturally occurring brines. The drinking water resources impacted directly in this stage are almost always groundwater resources, rather than surface water.

Though we could not in this assessment quantify an overall frequency of groundwater quality impacts from the well injection stage, we can describe factors which make impacts more or less frequent or more or less severe, as we did for other stages. We describe these factors below, first with frequency and then severity. Within the frequency discussion, we address factors by each pathway type.

10.1.3.1 Frequency

Pathway #1: Fluid movement into a drinking water resource through defects or deficiencies in the production well casing and/or cement.

To reach and then fracture the production zone, an oil or gas well must first be drilled and constructed down through the subsurface rock formations, often containing an overlying drinking water resource. Since the well passes through the drinking water resource, this means defects or deficiencies in the production well can lead to unintended movement of fluid into the drinking water resource. This can occur regardless of the vertical separation between the drinking water and the production zone.

The relatively brief hydraulic fracturing phase will likely impose the highest stresses to which the well will be exposed during its entire life. If the well cannot withstand the stresses experienced during hydraulic fracturing, the casing or cement can fail, resulting in the loss of mechanical integrity and the unintended movement of fluids into the surrounding environment.

A few studies have estimated rates of mechanical integrity failure of production wells resulting in the loss of *all* barriers protecting the groundwater or in contamination of groundwater in areas with hydraulic fracturing activity (Table 10-1). The estimates are all approximately equal to or less than 1% of wells drilled or hydraulically fractured over varying time frames. For most of these estimates, it is not possible to tell whether failures occurred during hydraulic fracturing or at some other point in the well's life, with the exception of the EPA's Well File Review (U.S. EPA, 2015n). If the failure rate from the Well File Review (0.5%) is applied to the estimates of 275,000 to 370,000 new wells hydraulically fractured nationally between 2000 and part of 2013 and 2000 and part of 2014, respectively (Chapter 3), we would expect roughly 1,370 to 1,850 mechanical integrity failures during this time-period (almost 14 to 15 years). Dividing by each time period yields approximately 100 to 125 mechanical integrity failures per year on average, resulting in the loss of all barriers protecting the groundwater during hydraulic fracturing. These estimates also have a high degree of uncertainty like the spills estimates. This not only stems from the lack of certainty about failure rates, but also uncertainties surrounding the estimates of the number of wells hydraulically fractured (Chapter 3). These are likely low estimates because they do not include mechanical integrity failures occurring outside of the hydraulic fracturing process (e.g., during the production phase), nor do they consider failures in re-fractured wells.

Table 10-1. Literature estimates of mechanical integrity failure rates resulting in contamination of groundwater or failure of all well barriers, potentially exposing the groundwater.

Citation	Mechanical Integrity Failure Rate (%)	Geographic Scope	Key Findings & Description of Mechanical Integrity Failure[a,b]
Fleckenstein et al. (2015)	0.06	Colorado-Wattenberg Field	An overall catastrophic failure rate of 0.06% was found for 16,828 wells studied (out of 17,948 total wells) drilled in the Wattenberg Field between 1970 and 2013. The timing of the failures was unknown, but most of the failures occurred in the older wells. The Wattenberg Formation is 4,400 ft (1,300m) below surface and typically is hydraulically fractured. A catastrophic failure was considered to have occurred when there was contamination of drinking water aquifers (i.e., the presence of thermogenic gas in a drinking water well) and evidence of a well defect such as exposed intermediate gas formations or casing leaks.
Considine et al. (2012)	0.06	Pennsylvania	Two wells were cited between 2008 and 2011 by PA DEP for causing methane migration into an aquifer. In this same time period, 3,533 wells were drilled.
Brantley et al. (2014)	0.12–1.1	Pennsylvania	Based on positive determination letters (PDLs) for violations that occurred between 2008 and 2012, Brantley et al. estimated between 7 and 64 problematic unconventional wells contaminated 85 properties. Since PDLs are tied to drinking water wells and not gas wells, Brantley et al. made assumptions about how many unconventional gas wells were represented by each PDL. This equates to problematic unconventional gas wells compromising approximately 0.1 to 1% of the 6,061 wells spudded between 2008 and 2012.[c] Not all of these PDLs may be due to mechanical integrity failures–they could also be due to other causes, such as spills, or methane migration from natural or other anthropogenic sources.
Vidic et al. (2013)	0.25	Pennsylvania	Of the 6,466 wells studied, 16 received notices regarding contamination of groundwater with gas or other fluids from the PA DEP associated with incidents that occurred between 2008 and 2013.

Citation	Mechanical Integrity Failure Rate (%)	Geographic Scope	Key Findings & Description of Mechanical Integrity Failure[a,b]
U.S. EPA (2016c)	0.5	National	In an estimated 0.5% of the approximately 28,500 hydraulic fracturing jobs surveyed, a failure occurred during hydraulic fracturing, such that there was no additional barrier between the annular space with fluid and the protected drinking water resource. While it could not definitively be determined whether fluid movement into the protected drinking water resource occurred, in these cases, all of the protective barriers intended to prevent such fluid migration had failed, leaving the groundwater source vulnerable to contamination.

[a] Note: While some information is available on the age of the wells studied, it is unclear whether the failure occurred during the hydraulic fracturing event, with the exception of the U.S. EPA (2016c) study. In that study, the failures occurred during hydraulic fracturing.

[b] While the Pennsylvania studies did not specifically identify whether the wells were involved in hydraulic fracturing operations, a significant portion of Pennsylvania's recent oil and gas activity is in the Marcellus Shale; therefore, many of the wells in these studies were most likely used for hydraulic fracturing.

[c] Spudding refers to starting the well drilling process by removing rock, dirt, and other sedimentary material with the drill bit (U.S. EPA, 2013f).

Not all wells are equally likely to lose mechanical integrity; instead, there are factors that make some wells more likely to experience a mechanical integrity failure than others. Well design and construction are two such factors. First, a primary element of well design is the placement of at least one additional layer of casing (besides the production casing) from the surface through the lowest depth of the drinking water resource. This additional casing provides redundancy if the production casing fails. In a study of 731 saltwater injection wells in the Williston Basin in North Dakota, Michie and Koch (1991) found the risk of aquifer contamination from leaks into the drinking water resource was 7 in 1,000,000 injection wells if a surface casing, in addition to the production casing, was set deep enough to cover the drinking water resource. The risk increased to 6,000 per 1,000,000 wells (or 6 in 1,000) if this additional casing was not set deeper than the bottom of the drinking water resource.

Second, fully cementing casing(s) through the entire drinking water resource affects the frequency of impacts. Uncemented sections of surface casing increase the frequency of fluid leaks from the well that can reach groundwater (Fleckenstein et al., 2015; Watson and Bachu, 2009). The EPA's Well File Review estimated that a portion of the protected groundwater resource identified by well operators was uncemented in 3% of the wells surveyed (U.S. EPA, 2015n). With approximately 25,000 to 30,000 new wells hydraulically fractured a year (Chapter 3), this percentage means 750 to 900 of the wells used in hydraulic fracturing operations annually might lack this protection. Adding re-fractured wells would increase the estimate of wells lacking this protection. Knowing the depth of the groundwater resource at the point of drilling and then setting and cementing casings below the lowest part of the drinking water resource can reduce the frequency or likelihood of an impact.

Third, the well's casing, cement, and components need to be designed and constructed to withstand the stresses applied to the well during hydraulic fracturing. In an example of inadequate well construction, hydraulic fracturing of a gas well with insufficient and improperly placed cement in Bainbridge Township, Ohio led to gas contamination of 26 domestic water supply wells and an explosion in the basement of one of the nearby homes. This was due in part to a failure to cement through the over-pressured gas formations and proceeding with the fracturing operation without adequate cement (ODNR, 2008). In another case, casings at an oil well near Killdeer, North Dakota, ruptured in 2010 following a pressure spike during hydraulic fracturing, allowing fluids to escape to the surface. Brine and tert-butyl alcohol were detected in two nearby water wells. Following an analysis of potential sources, the only potential source consistent with the conditions observed in the two impacted water wells was the ruptured well (U.S. EPA, 2015i).

In addition to well design and construction, the degradation or corrosion of well components can also increase the frequency of impacts to drinking water quality. Older wells exhibit more integrity problems as cement and casings age. The EPA's Well File Review estimated at least 10% of the wells represented in the national survey were greater than five years old at the time of hydraulic fracturing. Hydraulic fracturing or re-fracturing older wells has the potential to increase the frequency of casing or cement failures allowing unintended fluid migration into drinking water resources.

Confirming well mechanical integrity can reduce the frequency of water quality impacts. Pressure testing the casing used for hydraulic fracturing prior to the job can help detect problematic casing—and provide an opportunity to make needed repairs if necessary. Monitoring the annular space behind the casing used for hydraulic fracturing during the hydraulic fracturing job can detect well component failure in real time and signal for an immediate shut down. Based on the EPA's Well File Review study, casing pressure testing occurred at slightly less than 60% of the approximately 28,500 hydraulic fracturing jobs represented in that time frame (primarily 2009-2010) and annulus monitoring took place during slightly more than 50% of these same jobs, implying these activities did not always occur (U.S. EPA, 2016c). It is unclear whether the frequency of these practices have changed since this time period.

Pathway #2: Fluid movement into a drinking water resource through the fracture network.

The other potential pathway for fluid movement into a drinking water resource is through the fracture network. This could occur indirectly if the fracture network extends to a nearby well or its fracture network, or to another permeable subsurface feature, such as natural fractures or faults, which then allow the fluid to reach an underground drinking water resource. It could also occur directly by the fracture network extending out of the production zone into a drinking water resource, or hydraulic fracturing into a drinking water resource itself.[1] Key factors affecting the frequency of this pathway are the presence, distance, and condition of nearby wells; and the vertical

[1] Hereafter, fractures extending out of the production zone are referred to as "out-of-zone" fractures, consistent with Chapter 6.

separation distance and the characteristics of the intervening rock between the production zone and the drinking water resource.

Nearby wells (often called offset wells) can be a pathway for fluid movement, with hydraulic fracturing fluid from one production well moving through the subsurface and entering another nearby oil or gas well or its fracture network. These events are commonly referred to as "well communication events" or "frac hits." The communication event might simply be registered as an increase in pressure in the nearby well; yet there is also the possibility of damage to the nearby well or its components, causing a surface spill or a subsurface release of fluids. The EPA's Well File Review found 1% of the wells represented in the study experienced a frac hit, and the EPA spills report identified 10 spills attributed to well communication events (U.S. EPA, 2015m, n). It is unknown whether any fluid reached a drinking water resource from these spills. Where active nearby wells exist, operators of those wells can shut them in temporarily during the nearby hydraulic fracturing to reduce the possibility of spills or damage to their wells, and therefore, the potential for drinking water resource contamination.

The distance to the nearby well can affect the frequency of these communication events. In one study, the likelihood of a frac hit was less than 10% in hydraulically fractured wells more than 4,000 ft (1,219 m) apart, while nearly 50% in wells less than 1,000 ft (300 m) apart (Ajani and Kelkar, 2012). Distance was measured from the mid-point of each horizontal lateral. Thus, the closer the nearby wells, the more likely a communication event.

If nearby wells are in good condition and can withstand an increase in pressure, then an impact is unlikely to occur. However, if the nearby well is not able to withstand the pressure of the fluid, well components may fail and allow fluid to move into a drinking water resource. Because of this, nearby older or abandoned wells are of particular concern. In older wells near a hydraulic fracturing operation, plugs and cement may have degraded over time; in some cases, abandoned wells may never have been plugged properly. This can be a significant issue in areas with legacy (i.e., historic) oil and gas exploration. A Pennsylvania Department of Environmental Protection (PA DEP) report cited three cases where migration of natural gas had been caused by well communication events via old, abandoned wells (PA DEP, 2009c). In Tioga County, Pennsylvania, following hydraulic fracturing of a shale gas well, an abandoned well nearby produced a 30 ft (9 m) geyser of brine and gas for more than a week (Dilmore et al., 2015). Various studies estimate the number of abandoned wells in the United States to be significant. For example, the Interstate Oil and Gas Compact Commission (IOGCC, 2008) estimates that approximately 1 million wells were drilled in the United States prior to a formal regulatory system, and the status and location of many of these wells are unknown. Hydraulic fracturing operators can reduce the possibility of impacts by identifying nearby wells, and if necessary, plugging or otherwise addressing deficiencies in these wells.

If nearby wells serve as a pathway, fluid movement can bypass layers of intervening rock. In the absence of this pathway, however, vertical distance and the intervening rock between the production zone and the drinking water resource are factors affecting the possible movement of fluid into a drinking water resource. The extension of fractures out of the oil and/or gas production zone can—and does—occur. Examples have been reported in Greene County, Pennsylvania (Hammack et al., 2014); at the Killdeer site in Dunn County, North Dakota (U.S. EPA, 2015i); and in other wells within the Bakken Shale (Arkadakskiy and Rostron, 2013; Arkadakskiy and Rostron, 2012; Peterman et al., 2012). In a study across several major shale formations, Davies et al. (2012) found upward vertical fracture growth was often on the order of tens-to-hundreds of feet. One percent of the fractures had a fracture height greater than 1,148 ft (350 m), and the maximum fracture height among all of the data reported was 1,929 ft (588 m). This would suggest that substantial vertical separation could preclude out-of-zone-fractures from directly reaching the drinking water resource, although these measurements were only conducted in shale formations and the extension of fractures is not the only way the drinking water resource could be contaminated from out-of-zone fractures (see below). A modeling study also suggests fractures are unlikely to extend from the production zone directly to a shallow drinking water resource in a deep Marcellus-like environment (Kim and Moridis, 2015).

Not all fracturing occurs, however, with substantial vertical separation between the production zone and the drinking water resource (Figure 10-3). The EPA's Well File Review found that 20% of wells used for hydraulic fracturing had less than 2,000 ft (600 m) between the shallowest point of fracturing and the base of the protected groundwater resource (U.S. EPA, 2015n). In coalbed methane (CBM) plays, typically shallower than shale gas plays, these separation distances can be smaller. For example, in the Raton Basin of southern Colorado and northern New Mexico, approximately 10% of CBM wells have less than 675 ft (206 m) of separation between the production zone and the depth of local water wells. In certain areas of the basin, this distance is less than 100 ft (31 m) (Watts, 2006). Many of these areas are shallower in depth, and fracture growth has been shown to be primarily horizontal, rather than vertical, at less than 2,000 ft (600 m) from the surface (Fisher and Warpinski, 2012). Nevertheless, the possibility of an out-of-zone fracture reaching a drinking water resource is more likely in a setting with less vertical separation than with more.

Even if an out-of-zone fracture does not extend into a drinking water resource, it could connect to other permeable subsurface features, such as natural fractures or faults, which could then connect to a drinking water resource. Thus, properties of the intervening rock can also make this pathway more or less frequent or likely. For instance, in the Pavillion gas field in Wyoming, there are no laterally-continuous confining layers to prevent upward movement of fluids into the groundwater (Digiulio and Jackson, 2016). While flow of subsurface fluids generally tends to be downward, local areas of upward flow have been observed (Digiulio and Jackson, 2016).

There are cases of hydraulic fracturing without vertical separation between the drinking water resource and the production zone (Figure 10-3). The co-location of the oil or gas formation with the drinking water resource is the factor affecting the frequency of an impact in these cases. Directly fracturing into a drinking water resource causes an impact because it changes the quality of the resource by introducing hydraulic fracturing fluids. The EPA's Well File Review found an estimated 0.4% of the wells represented in the study had perforations used for hydraulic fracturing shallower than the base of the protected groundwater resource, as reported by well operators (U.S. EPA, 2015n). The EPA's Well File Review did not examine these instances by formation type. This practice may be concentrated in locations in western states, especially in CBM plays. Examples include the Raton Basin in Colorado (U.S. EPA, 2015k), the San Juan Basin of Colorado and New

Mexico (U.S. EPA, 2004a), and the Powder River Basin of Montana and Wyoming (Dahm et al., 2011; ALL Consulting, 2004; U.S. EPA, 2004a). This is a concern in the short term (should there be people using these drinking water resources currently) and the long term (if drought or other conditions necessitate the future use of these drinking water resources). For the most part in this chapter, we focused on factors which can be managed, changed, or used to identify areas to target monitoring efforts. In this situation, hydraulic fracturing directly into a drinking water resource would need to cease if it was decided the resulting impacts to drinking water resource quality were unacceptable.

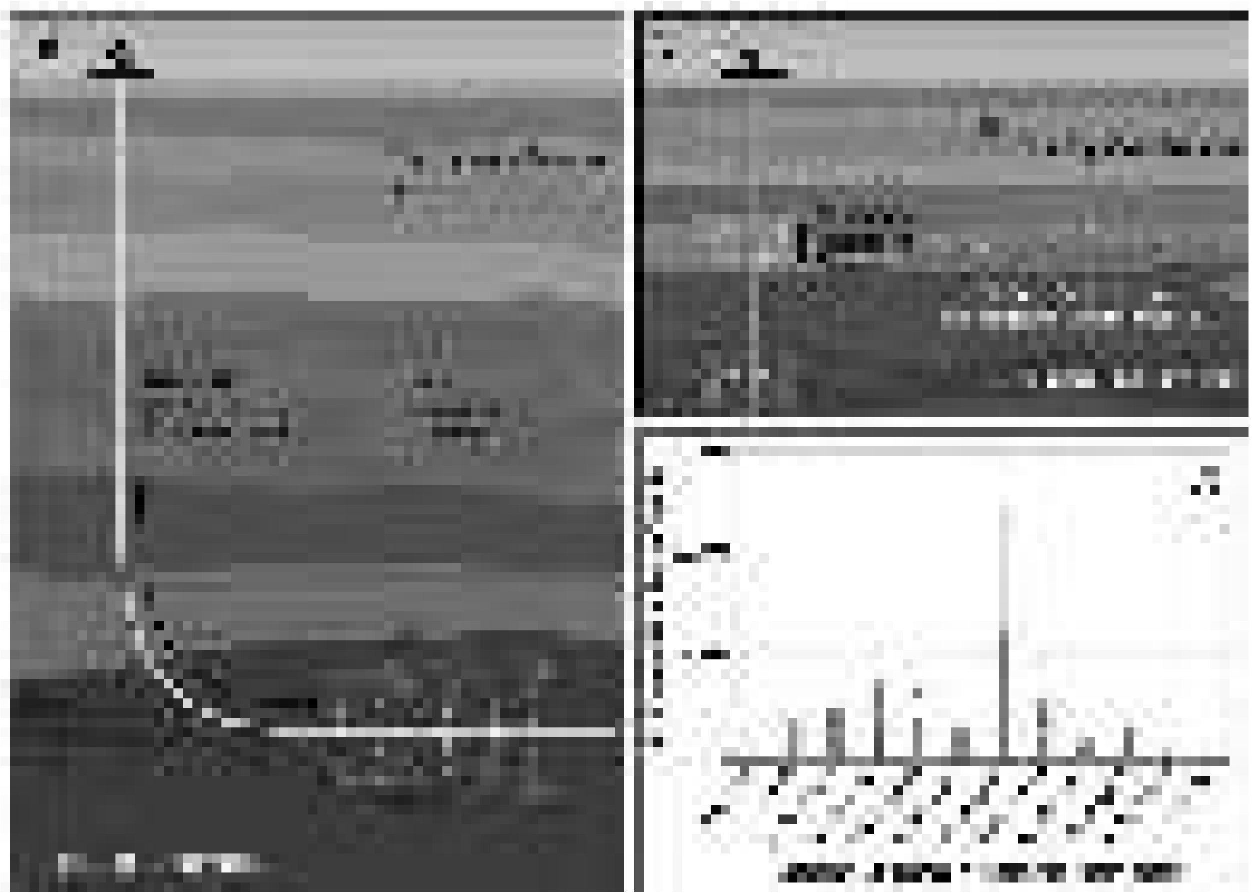

Figure 10-3. Separation in measured depth between drinking water resources and hydraulically fractured intervals in wells.

In panel (a), the oil- and gas-bearing formation (dark gray) being hydraulically fractured is much deeper than the depth where drinking water resources (light blue) exist, and hence a comparatively large separation distance exists. In panel (b), there are two oil- and gas-bearing formations (dark gray and grayish blue) being hydraulically fractured. The shallower formation has no separation distance, because the water also contained in this formation is a drinking water resource. Panel (b) also shows another subsurface drinking water zone at a shallower depth (light blue). Multiple groundwater zones of varying qualities can exist between the production zone and the surface. These two panels depict end-member cases of separation distance: from large separation distances to no separation distance. The graph in panel (c) illustrates the distribution of separation distances among the approximately 23,000 oil and gas production wells hydraulically fractured by nine service companies between 2009 and 2010 (U.S. EPA, 2015n). Error bars in the panel (c) display 95% confidence intervals.

Lastly, the presence of gas, as opposed to liquids, in the subsurface may be a factor affecting the frequency of impacts from fluid movement via defects or deficiencies in the well (pathway #1), or through the fracture network (pathway #2). The low density of gas compared to liquids makes it buoyant, which creates an upward drive toward the surface. Thus, gas found in the subsurface, such as methane, can exploit pathways in a well (such as along a well lacking mechanical integrity), or in the surrounding rock (such as induced or naturally occurring fractures). If a pathway exists and gas is present, it can reach groundwater used for drinking. Consequently, gases could be more likely to contaminate drinking water resources than liquids (Li et al., 2016a).

10.1.3.2 Severity

The well injection chapter (Chapter 6) focused primarily on the potential for impacts to occur and factors affecting frequency. By contrast, we have little-to-no information on factors affecting the severity of impacts for this stage of the hydraulic fracturing water cycle. Severity would likely be affected by the chemical composition of the fluid entering the drinking water resource; the volume of the fluid; the duration in which that volume is delivered; and the concentration of the fluid and its specific components, among other factors. Logically, the relatively simple pathway of a mechanical integrity failure might result in the highest fluid volume delivered to a drinking water resource over a short period of time—e.g., contamination of water wells in Bainbridge Township, Ohio. By contrast, fluid movement through a fracture network, then through the intervening rock, and finally into a drinking water resource may take a longer time and deliver a comparatively lower volume. Even in this case, however, the impacts could still be severe if the fluid movement was to go undetected and unaddressed.

10.1.4 Wastewater Disposal and Reuse

The last stage of the hydraulic fracturing water cycle is wastewater disposal and reuse. Produced water from hydraulically fractured oil or gas production wells is managed predominantly through disposal in underground Class II wells. Secondarily, it is disposed of via other practices, such as discharge to surface waters or disposal in pits or evaporation ponds, or reused in other hydraulic fracturing operations. Activities in the wastewater disposal and reuse stage of the hydraulic fracturing water cycle can impact drinking water resources in some instances. Two such activities are: the discharge of inadequately treated wastewater to surface water, and the storage or disposal of wastewater in unlined pits or impoundments leading to contamination of surface water or groundwater. In this section, we address factors increasing or decreasing the frequency or severity of impacts from these activities. As in the water acquisition section, we combine our discussion of frequency and severity here.

10.1.4.1 Frequency and Severity

Discharge of inadequately treated wastewater has impacted surface water. The quality of the wastewater discharged is a factor affecting the frequency and severity of impacts. This factor is a function of the chemical characteristics of the wastewater prior to treatment (i.e., the composition and concentration of chemicals in the wastewater) and the efficacy of the treatment process. The pre-2011 treatment of Marcellus wastewater in Pennsylvania illustrates this combination. In

Pennsylvania before 2011, wastewater from shale gas operations was treated at centralized waste treatment facilities (CWTs) and publicly owned treatment works (POTWs). The POTWs and some CWTs at the time were not equipped to adequately treat high TDS wastewater. This resulted in wastewater discharges containing elevated levels of TDS, including bromide and iodide, to surface waters.

The elevated levels of TDS raised concerns about the formation of disinfection byproducts (DBPs) after treatment at downstream drinking water facilities. Disinfection byproducts are formed when organic material comes in contact with disinfectants (e.g., chlorine, chloramine, chlorine dioxide or ozone). Many DBPs have long-term health effects including an increased risk of cancer, anemia, liver and kidney effects, and central nervous system effects. Of particular concern are DBPs formed in the presence of bromide or iodide, which are considered particularly toxic. Management of DBPs places a burden on downstream drinking water utilities. Concerns regarding elevated TDS (in particular high bromide) and the potential for formation of DBPs led the PA DEP to take steps in 2010 and 2011 to route Marcellus Shale wastewater away from POTWs and CWTs (that could not treat for TDS) to alternate options such as disposal via injection wells, on-site reuse, or reuse after limited treatment at CWTs. By 2014, only a small percentage (approximately less than 1%) of Marcellus wastewater went to CWTs permitted to discharge to surface waters (Figure 10-1). Additionally, the new EPA pretreatment standards prohibit oil and gas operators from sending unconventional oil and gas wastewater directly to POTWs (U.S. EPA, 2016d).

The combination of wastewater composition and inadequate treatment have also resulted in the discharge of other constituents such as barium, strontium, and radium into surface waters in Pennsylvania. Marcellus Shale wastewater contains radium, naturally occurring in the subsurface formation. Radium has been found in stream sediments at discharge points for POTWs and CWT facilities that have accepted Marcellus Shale wastewater. The ratio of radium isotopes (radium-228 to radium-226) in these sediments is consistent with ratios in Marcellus Shale wastewater (Warner et al., 2013a). Radium-226, with a half-life of approximately 1,600 years, causes long-term contamination. The practice of management of wastewaters via POTWs and CWTs without TDS removal has declined, yet it remains uncertain whether the discharge of radionuclides to surface waters from the oil and gas industry in Pennsylvania has ceased entirely (PA DEP, 2015b).

The storage or disposal of wastewater in pits or impoundments can also lead to contamination of surface water or groundwater resources. This can occur via surface spills or overflows. It can also occur via infiltration into the soil and percolation to groundwater through the pit itself. Whether the pit or impoundment is lined is an important factor affecting the frequency and severity of impacts on groundwater due to subsurface leaching. Historically, unlined pits have been used to dispose of wastewater via percolation (or evaporation). While this practice has been banned in most states, it is allowed in certain locations or instances (e.g., storage of wastewater, but not disposal) as of July 2016. Even where prohibited, unpermitted unlined disposal or storage pits exist. For example, approximately 1,000 unlined storage or disposal pits of oil and gas wastewater are located in the Central Valley Region of California (California State Water Resources Control Board, 2016; Esser et al., 2015). Of these, approximately 60% were still active as of July 2016, and roughly 20% of those pits lacked permits (CA Water Board, 2016).

Unlined pits have been shown to cause contamination of drinking water resources. The presence of BTEX (benzene, toluene, ethylbenzene, and xylenes) and other organics in groundwater are linked to pits in California and New Mexico (California Regional Water Quality Control Board Central Valley Region, 2015; Sumi, 2004; Eiceman, 1986). Groundwater impacts downgradient of an unlined pit in Oklahoma included high salinity (3500-25,600 mg/L) and the presence of volatile organic compounds (Kharaka et al., 2002). Impacts can also occur in the case of disposal of relatively low TDS wastewater (Healy et al., 2011; Healy et al., 2008). For example, a CBM wastewater impoundment in the Powder River Basin of Wyoming resulted in high concentrations of TDS, chloride, nitrate, and selenium in the groundwater (Healy et al., 2011; Healy et al., 2008). Total dissolved solids exceeded 100,000 mg/L in one groundwater sample, despite the much lower concentrations (2,300 mg/L) in the wastewater being discharged (Healy et al., 2008). Most of the solutes found in the groundwater did not originate with the CBM wastewater, but rather resulted from dissolution of previously existing salts and minerals in the subsurface. Lining pits or using closed-loop systems (i.e., tanks) can decrease the frequency of such impacts.

10.1.5 Summary

In the above section, we synthesized the information in this assessment by discussing factors increasing or decreasing the frequency or severity of impacts from activities in the hydraulic fracturing water cycle. We focused particularly on factors that could be managed, changed, or used to identify locations for additional monitoring or alteration of practices. Based on the information reviewed, we conclude the following combinations of activities and factors are more likely than others to result in more frequent or more severe impacts:

- Water withdrawals for hydraulic fracturing in times or areas of low water availability, particularly in areas with limited or declining groundwater resources;
- Spills during the management of hydraulic fracturing fluids and chemicals or produced water that result in large volumes or high concentrations of chemicals reaching groundwater resources;
- Injection of hydraulic fracturing fluids into wells with inadequate mechanical integrity, allowing gases or liquids to move to groundwater resources;
- Injection of hydraulic fracturing fluids directly into groundwater resources;
- Discharge of inadequately treated hydraulic fracturing wastewater to surface water resources; and
- Disposal or storage of hydraulic fracturing wastewater in unlined pits, resulting in contamination of groundwater resources.

Conversely, the scientific literature and data provide evidence that certain factors can reduce the frequency or severity of impacts. Based on the information reviewed in this assessment, we conclude the following factors are likely to reduce the frequency or severity of impacts:

- Passby flows, or low-flow criteria, for surface water withdrawals, and the use of brackish groundwater or reused wastewater as substitutes for fresh water withdrawals;

- Implementation of spill prevention and response measures;
- Design and placement of well casing and cement able to withstand the stresses imposed by hydraulic fracturing (including identifying the depth of the drinking water resource at the point of drilling, and setting and cementing casings through the entire drinking water resource);
- Confirming mechanical integrity of oil and gas wells prior to, during, and after hydraulic fracturing, and correcting deficiencies if necessary;
- Identification of active or abandoned wells near hydraulic fracturing operations and, if necessary, adjustment of the operations to minimize well-to-well communication and/or plugging improperly abandoned wells;
- The use of treatment technologies to remove TDS, and other constituents, such as radium, when present prior to discharge; and
- Storage of wastewater in lined pits or the use of closed-loop systems instead of pits.

The above factors are not the only factors that can reduce the frequency or severity of impacts, yet are the ones most emphasized by the information reviewed for this assessment. It should be noted that the above factors reduce, but do not completely eliminate, the possibility of an impact occurring. In the case of hydraulic fracturing directly into a drinking water resource or disposal of wastewater via unlined pits, we did not identify factors which could reduce the frequency or severity of impacts, besides restricting the activity itself.

10.2 Uncertainties and Data Gaps

In this assessment, we identified impacts on drinking water resources in all stages of the hydraulic fracturing water cycle and described the factors affecting the frequency or severity of impacts. The major conclusions presented above (in Section 10.2.5) are the strongest conclusions based on data and information synthesized for the assessment.

There were also many areas within the assessment for which strong conclusions could not be reached. This was because of the lack of publicly available data and large uncertainties in available sources of information. Below, we provide perspective on what data gaps and uncertainties prevented us from drawing additional strong conclusions about the potential for impacts and/or the factors affecting the frequency or severity of impacts.

We encountered uncertainties associated with, and gaps in, aggregated, publicly accessible information about both activities in the hydraulic fracturing water cycle and groundwater data. In general, comprehensive information on the location of activities in the hydraulic fracturing water cycle is lacking, either because it is not collected, not publicly-available, or prohibitively difficult to aggregate. Thus, we lacked complete information on the geographic locations of well sites (both new and existing) where the chemical mixing, well injection, and produced water handling stages take place; the depth(s) of zones that have been hydraulically fractured in these wells; where water is being acquired (i.e., the source water) for hydraulic fracturing; and where hydraulic fracturing wastewater is treated and/or disposed. FracFocus provided data on well locations, and water and other chemicals used at those locations. However, reporting to FracFocus at the time period studied was not always required, making it difficult to determine the completeness or representativeness of the information.

In addition, there are uncertainties about where groundwater resources are located. This includes the thickness of the resource, from its top to its lowest depth, and its relation to the shallowest depth where hydraulic fracturing occurred. If comprehensive data about the locations of both drinking water resources and activities in the hydraulic fracturing water cycle were available, it would have been possible to more completely identify areas in the United States where hydraulic fracturing-related activities and drinking water resources overlap.

There are also uncertainties and data gaps related to chemicals used in hydraulic fracturing fluid and those detected in produced water. Some chemicals and chemical mixtures remain undisclosed because of confidential business information (CBI) claims. Well operators claimed at least one chemical as CBI at more than 70% of disclosures reported to FracFocus between 2011 and early 2013. Data suggests this practice is increasing. Konschnik and Dayalu (2016) reported that 92% of FracFocus disclosures submitted between approximately March 2011 and April 2015 included at least one chemical claimed as confidential. When chemicals are claimed as CBI, there is no public means of accessing information on these chemicals. Furthermore, many of the chemicals and chemical mixtures disclosed, or those detected in produced water, lack information on properties affecting their movement, persistence, and toxicity in the environment should they be spilled. Better information on these chemicals would allow for a more robust evaluation of potential human health hazards posed, and thus a better understanding about the severity of impacts should the chemicals reach drinking water resources.

In places where we know hydraulic fracturing water cycle activities have occurred, data to assess impacts are often either not collected or are not publicly available in accessible forms. Specifically, local water quality monitoring and well mechanical integrity integrity data are not consistently collected or readily available. In particular, sufficient baseline data on local water quality are needed to quantify any changes post-hydraulic fracturing. There are exceptions to this, for example, the state of California recently implemented a plan to make water quality monitoring information public (Text Box 10-1). In general, however, the limited amount of data collected before, during, and after hydraulic fracturing activities and made public, reduces the ability to determine whether hydraulic fracturing affected drinking water resources.

Text Box 10-1. Hydraulic Fracturing and Groundwater Quality Monitoring in California.

In July 2015, the California Water Resources Control Board adopted Senate Bill 4 (SB4), *Model Criteria for Groundwater Monitoring in Areas of Oil and Gas Well Stimulation*. This resolution directed the establishment of a "comprehensive regulatory groundwater monitoring and oversight program...in order to assess the potential effects of well stimulation treatment activities on the state's groundwater resources" (California State Water Resources Control Board, 2015). The adoption of SB4 concluded a multi-year process, which incorporated stakeholder engagement, review by the public, and consultation with an expert scientific panel.

(Text Box 10-1 is continued on the following page.)

Text Box 10-1 (continued). Hydraulic Fracturing and Groundwater Quality Monitoring in California.

The recommendations of the expert panel informed the creation and implementation of SB4 with respect to criteria "to be used for assessment, sampling, analytical testing, and reporting of water quality associated with oil and gas well stimulation activities" (Esser et al., 2015).

The resolution requires two different scales of groundwater monitoring for different purposes. First, it requires well-by-well (also called "area-specific") groundwater monitoring by well operators. This includes groundwater monitoring conducted for all hydraulic fracturing projects initiated after July 2015. Each oil or gas production well operator must submit a design and timeline for monitoring groundwater resources in proximity to its proposed well. The State Water Resources Control Board approves the monitoring plan before hydraulic fracturing can proceed. The groundwater monitoring plan must include:

- The installation of monitoring wells within 0.5 miles of the wellhead. At least one monitoring well must be upgradient of the production well and two monitoring wells must be downgradient. Should the production well penetrate more than one protected groundwater resource (as defined by the resolution), monitoring wells must facilitate sampling of at least one that is shallow and one that is deep.
- A monitoring timeline that includes sampling prior to production well construction and hydraulic fracturing, as well as semi-annual sampling after completion.
- A list of water quality parameters and constituents to be monitored, including TDS, specific metals, and specific organic compounds.

The area-specific monitoring requirements also include submission of information by well operators about geologic and human-made features in the subsurface that could serve as pathways for impacts to groundwater, aspects of production well construction, and hydraulic fracturing fluid composition.

Second, a regional groundwater monitoring program will document trends in baseline water quality and locate protected groundwater state-wide. In addition to monitoring for trends in groundwater quality related to activities at well sites, it will also be designed to detect trends related to impacts from wastewater disposal practices.

All data from the monitoring programs will be publicly accessible in a state-maintained database. The database is intended to support public health, scientific, and academic needs, as well as future "investigation, assessment, and research relevant to oil and gas development impacts on groundwater quality" (Esser et al., 2015).

Together, the data and information collected and made publicly available as part of the area-specific and regional groundwater monitoring in California will help fill data gaps identified in this section of the assessment by locating groundwater resources, monitoring drinking water resources at spatial and temporal scales relevant for detecting impacts from activities in the hydraulic fracturing water cycle, and distinguishing impacts from hydraulic fracturing activities from the impacts of other potential sources.

In the cases where effects are suspected, it is often difficult to separate the potential effects of hydraulic fracturing activities from effects of broader oil and gas industry activities and other industries or causes. The use of long-lasting, mobile tracer chemicals added to hydraulic fracturing fluids to monitor for impacts has been proposed (Kurose, 2014), but has received relatively little attention in the scientific literature as of mid-2016. Instead, measured changes in water quality parameters can be associated with, but not necessarily diagnostic of, impacts from hydraulic fracturing activities. For instance, measurable changes in methane levels, TDS, ratios of geochemical constituents, and isotopic ratios might suggest impacts from hydraulic fracturing but could also be from other sources, either natural or anthropogenic. To try to assign a cause, these measurements often have to be followed with further collection of evidence supporting or refuting hydraulic fracturing activities as the cause of the changes. (See Text Box 10-2 for discussion of causal assessments.)

Text Box 10-2. Causal Assessment and Hydraulic Fracturing Water Cycle Activities.

A number of recent studies have conducted regional-scale assessments of trends in water quality in areas with hydraulic fracturing activity, showing either no trend or trends linked temporally or spatially with hydraulic fracturing (Burton et al., 2016; Hildenbrand et al., 2016; Hildenbrand et al., 2015; Siegel et al., 2015; Darrah et al., 2014; Fontenot et al., 2013; Warner et al., 2013b) Regional assessments can be important for integrating information over broader scales, and for posing hypotheses about how hydraulic fracturing water cycle activities may impact drinking water resources. Oftentimes, however, activities in the hydraulic fracturing water cycle are merely one of several possible causes of an observed change in water quality or quantity at a specific site. In this case, more thorough, site-specific investigations are often necessary. Causal assessment (also called causal analysis) involves collecting multiple kinds of evidence to determine which of several possible causes of contamination is most likely.

Causal assessment requires several steps. First, the spatial and temporal scope of the issue is defined, including a description of all the possible causes of an observed impact, in this case the change in quality or quantity of a drinking water resource. Once this is done, evidence is collected and assembled to support or refute the potential causes. Evidence indicating how a potential cause and an observed effect are related in time can help support or refute potential causes. Other kinds of evidence can also be useful in identifying a cause, including: determining whether the composition and volume of a leaked, spilled, or treated and discharged fluid are capable of causing observed impacts on water quality; and determining whether a physical pathway between a well or well site exists by measuring the mechanical integrity of hydraulically fractured wells and/or establishing the presence/absence of a contaminant plume.

Ideally, the evidence helps exclude possible causes of the reported contamination, narrowing down the list of potential causes to a single cause. Causal assessments can take a long time to complete and can require a lot of resources and expertise. In some situations, available data and resources are simply not sufficient to definitively identify the cause. Nevertheless, causal assessments conducted in a consistent and transparent way can help enable the identification of the likely cause(s) of contamination of drinking water resources.

The retrospective case studies conducted by the EPA under the Study Plan are examples of scientific investigations using a multiple lines of evidence approach consistent with the principles of causal assessment (U.S. EPA, 2015i, j, l, 2014f, g). These case studies were cited throughout this report. For instance, as noted previously, the Killdeer, North Dakota case study found that an inner string of casing burst during hydraulic fracturing of an oil well, resulting in a release of hydraulic fracturing fluids and formation fluids that impacted a groundwater resource (U.S. EPA, 2015i). Following an analysis of potential sources, the only potential source consistent with the conditions observed was the ruptured well (U.S. EPA, 2015i).

Regardless of whether a single cause can be determined, actions can still be taken to mitigate one or more potential causes of contamination. Information gained once the suspected activity has been halted or at least reduced could elucidate whether hydraulic fracturing operations are more or less likely to have been the source of the contamination.

Many members of the public are interested in understanding the national frequency of impacts to drinking water resources from activities across the entire hydraulic fracturing water cycle. Because of the significant data gaps and uncertainties in the available data, it was not possible to estimate the national frequency of impacts to drinking water resources from activities in the hydraulic fracturing water cycle collectively. We were, however, able to estimate impact frequencies in some, limited cases within the larger hydraulic fracturing water cycle (i.e., spills of hydraulic fracturing fluids or produced water, and mechanical integrity failures). These more specific estimates had a high degree of uncertainty, but were the best estimates that could be provided with the data and literature currently available.

Finally, it should be recognized that this assessment is a snapshot in time. Our understanding of the factors affecting the frequency or severity of impacts may change in the future as industry practices evolve and new information becomes available.

10.3 Use of this Assessment

This assessment contributes to the understanding of the potential impacts to drinking water resources by activities in the hydraulic fracturing water cycle and the factors influencing those impacts. The scientific information presented can be used by federal, tribal, state, and local officials; industry; and the public to better understand and address vulnerabilities of drinking water resources to activities in the hydraulic fracturing water cycle.

The uncertainties and data gaps identified throughout this assessment could be used to identify future data collection efforts. Data collection efforts could include, for example, surface water and groundwater monitoring programs in areas with hydraulically fractured oil and gas production wells; collection and the public dissemination of data on the condition of hydraulically fractured wells; or targeted research programs to better characterize the environmental fate and transport and human health hazards associated with chemicals in the hydraulic fracturing water cycle. Data collected and analyzed through new data collection efforts may identify new factors increasing or decreasing the frequency or severity of impacts.

In the near term, decision-makers could focus their attention on the combinations of activities and factors that we conclude are more likely than others to result in more frequent or more severe impacts (Section 10.2.5). By focusing attention on the above combinations, impacts to drinking water resources from activities in the hydraulic fracturing water cycle can be prevented or reduced.

Overall, the practice of hydraulic fracturing is expanding and continues to change. Oil and gas production associated with hydraulic fracturing was insignificant in 2000, but by 2015 it accounted for an estimated 51% of U.S. oil production and 67% of U.S. gas production (EIA, 2016c, d). The number of wells drilled and hydraulically fractured is likely to continue to increase in the coming decades (EIA, 2014a). The work of evaluating potential impacts from combinations of activities and factors in the hydraulic fracturing water cycle will need to keep pace with this industry and as new scientific studies are produced. This assessment provides a foundation for those efforts, while offering information to support the reduction of current vulnerabilities of drinking water resources.

Chapter 11. References

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Hyperlinks to the reference citations throughout this document will take you to the ORD National Center for Environmental Assessment HERO database (Health and Environmental Research Online) at https://hero.epa.gov/hero. HERO is a database of scientific literature used by the U.S. EPA in the process of developing selected science assessments.

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Front cover (top): Illustrations of activities in the hydraulic fracturing water cycle.
From left to right: Water Acquisition, Chemical Mixing, Well Injection, Produced Water Handling, and Wastewater Disposal and Reuse.

Front cover (bottom): Aerial photographs of hydraulic fracturing activities.
Left: Near Williston, North Dakota. Image ©J Henry Fair / Flights provided by LightHawk.
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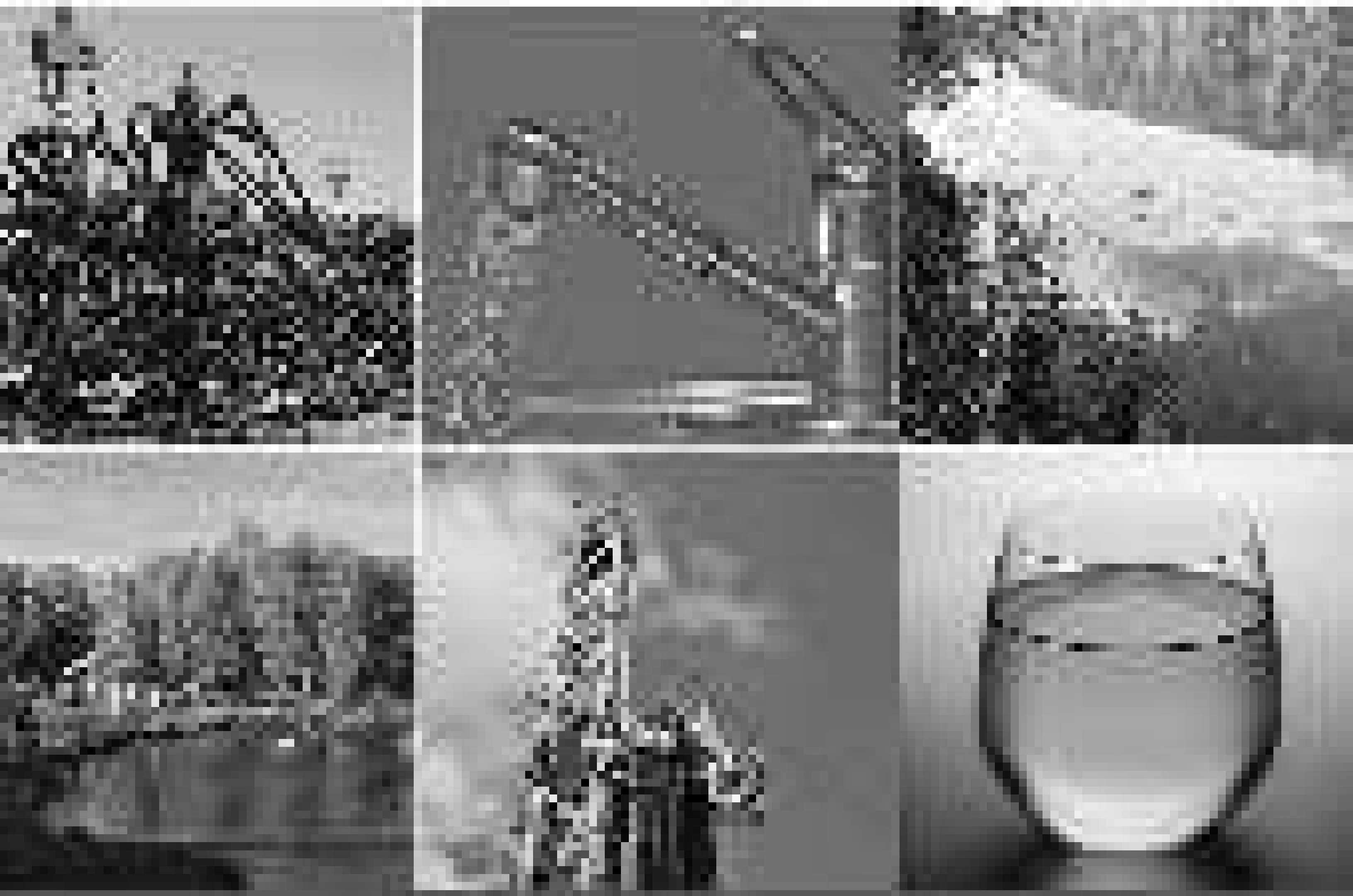

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U.S. Environmental Protection Agency
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Evaluation of Impacts to Underground Sources of Drinking Water by Hydraulic Fracturing of Coalbed Methane Reservoirs

Final

Office of Water
Office of Ground Water and Drinking Water (4606M)
EPA 816-R-04-003
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June 2004

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United States Environmental Protection Agency
Office of Water
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Drinking Water Protection Division
Prevention Branch
1200 Pennsylvania Avenue, NW (4606M)
Washington, DC 20460

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Executive Summary

> A USDW is defined as an aquifer or a portion of an aquifer that:
> A. 1. *Supplies any public water system; or*
> 2. *Contains sufficient quantity of groundwater to supply a public water system; and*
> i. *currently supplies drinking water for human consumption; or*
> ii. *contains fewer than 10,000 milligrams per liter (mg/L) total dissolved solids (TDS); and*
> B. *Is not an exempted aquifer.*
>
> ***NOTE***: Although aquifers with greater than 500 mg/L TDS are rarely used for drinking water supplies without treatment, the Agency believes that protecting waters with less than 10,000 mg/L TDS will ensure an adequate supply for present and future generations.

The U.S. Environmental Protection Agency (EPA, or the Agency) conducted a study that assesses the potential for contamination of underground sources of drinking water (USDWs) from the injection of hydraulic fracturing fluids into coalbed methane (CBM) wells. To increase the effectiveness and efficiency of the study, EPA has taken a phased approach. Apart from using real world observations and gathering empirical data, EPA also evaluated the theoretical potential for hydraulic fracturing to affect USDWs. Based on the information collected and reviewed, EPA has concluded that the injection of hydraulic fracturing fluids into CBM wells poses little or no threat to USDWs and does not justify additional study at this time. EPA's decision is consistent with the process outlined in the April, 2001 Final Study Design, which is described in Chapter 2 of this report.

The first phase of the study, documented in this report, is a fact-finding effort based primarily on existing literature to identify and assess the potential threat to USDWs posed by the injection of hydraulic fracturing fluids into CBM wells. EPA evaluated that potential based on two possible mechanisms. The first mechanism was the direct injection of fracturing fluids into a USDW in which the coal is located, or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system). The second mechanism was the creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

EPA also reviewed incidents of drinking water well contamination believed to be associated with hydraulic fracturing and found no confirmed cases that are linked to fracturing fluid injection into CBM wells or subsequent underground movement of fracturing fluids. Although thousands of CBM wells are fractured annually, EPA did not find confirmed evidence that drinking water wells have been contaminated by hydraulic fracturing fluid injection into CBM wells.

EPA has determined that in some cases, constituents of potential concern (section ES-6) are injected directly into USDWs during the course of normal fracturing operations. The use of diesel fuel in fracturing fluids introduces benzene, toluene, ethylbenzene, and xylenes (BTEX) into USDWs. BTEX compounds are regulated under the Safe Drinking Water Act (SDWA).

Given the concerns associated with the use of diesel fuel and the introduction of BTEX constituents into USDWs, EPA recently entered into a Memorandum of Agreement (MOA) with three major service companies to voluntarily eliminate diesel fuel from hydraulic fracturing fluids that are injected directly into USDWs for CBM production (USEPA, 2003). Industry representatives estimate that these three companies perform approximately 95 percent of the hydraulic fracturing projects in the United States. These companies signed the MOA on December 15, 2003 and have indicated to EPA that they no longer use diesel fuel as a hydraulic fracturing fluid additive when injecting into USDWs.

ES-1 How Does CBM Play a Role in the Nation's Energy Demands?

CBM production began as a safety measure in underground coalmines to reduce the explosion hazard posed by methane gas (Elder and Deul, 1974). In 1980, the U.S. Congress enacted a tax credit for non-conventional fuels production, including CBM production, as part of the Crude Oil Windfall Profit Act. In 1984, there were very few CBM wells in the U.S.; by 1990, there were almost 8,000 CBM wells (Pashin and Hinkle, 1997). In 1996, CBM production in 12 states totaled about 1,252 billion cubic feet, accounting for approximately 7 percent of U.S. gas production (U.S. Department of Energy, 1999). At the end of 2000, CBM production from 13 states totaled 1.353 trillion cubic feet, an increase of 156 percent from 1992. During 2000, a total of 13,973 CBM wells were in production (GTI, 2001; EPA Regional Offices, 2001). According to the U.S. Department of Energy, natural gas demand is expected to increase at least 45 percent in the next 20 years (U.S. Department of Energy, 1999). The rate of CBM production is expected to increase in response to the growing demand.

In evaluating CBM production and hydraulic fracturing activities, EPA reviewed the geology of 11 major coal basins throughout the United States (Figure ES-1). The basins shown in red have the highest CBM production volumes. They are the Powder River Basin in Wyoming and Montana, the San Juan Basin in Colorado and New Mexico, and the Black Warrior Basin in Alabama. Hydraulic fracturing is or has been used to stimulate CBM wells in all basins, but it has not frequently been used in the Powder River, Sand Wash, or Pacific Coal Basins. Table ES-1 provides production statistics for 2000 and information on hydraulic fracturing activity for each of the 11 basins in 2000.

Figure ES-1. Major United States Coal Basins

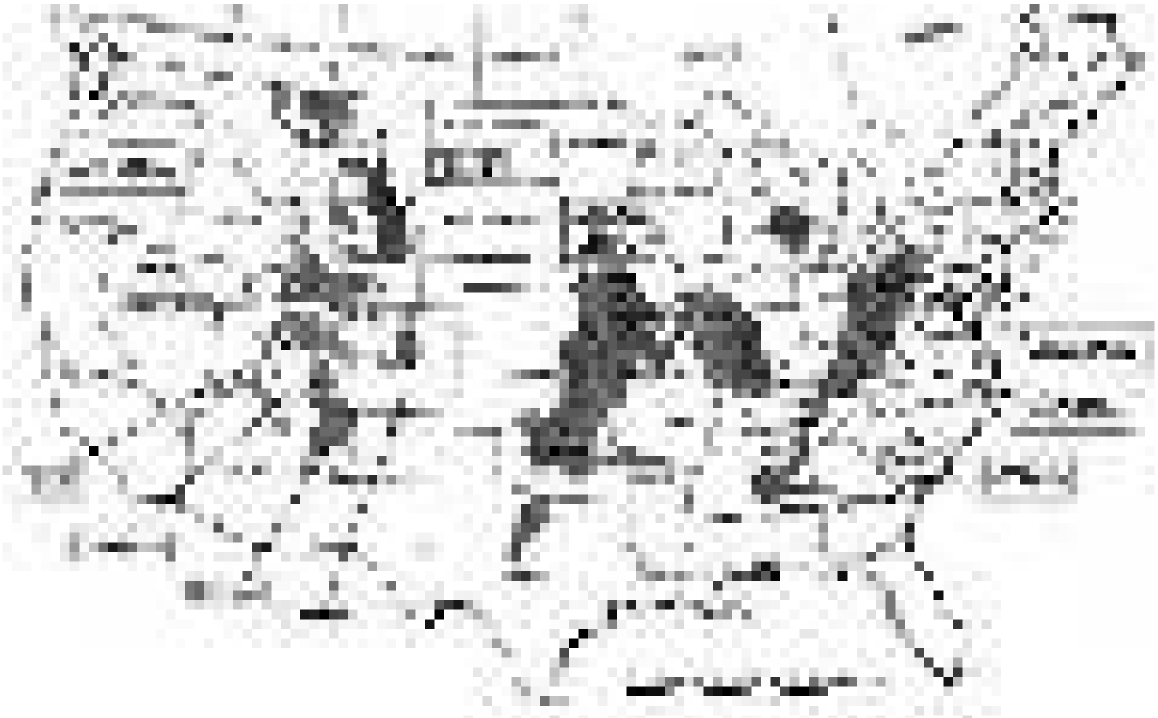


Table ES-1. Coal Basins Production Statistics and Activity Information in the U.S.

Basin	Number of CBM Producing Wells (Year 2000)*	Production of CBM in Billions of Cubic Feet (Year 2000)*	Does Hydraulic Fracturing Occur?
Powder River	4,200	147	Yes (but infrequently)
Black Warrior	3,086	112	Yes
San Juan	3,051	925	Yes
Central Appalachian	1,924	52.9	Yes
Raton Basin	614	30.8	Yes
Uinta	494	75.7	Yes
Western Interior	420	6.5	Yes
Northern Appalachian	134	1.41	Yes
Piceance	50	1.2	Yes
Pacific Coal	0	0	Yes (but infrequently)
Sand Wash	0	0	Yes (but infrequently)

* Data provided by the Gas Technology Institute and EPA Regional Offices. Production figures include CBM extracted using hydraulic fracturing and other processes.

ES-2 What Is Hydraulic Fracturing?

CBM gas is not structurally trapped in the natural fractures in coalbeds. Rather, most of the methane is adsorbed to the coal (Koenig, 1989; Winston, 1990; Close, 1993). To extract the CBM, a production well is drilled through the rock layers to intersect the coal seam that contains the CBM. Next, fractures are created or existing fractures are enlarged in the coal seam through which the CBM can be drawn to the well and pumped to the surface.

Figure ES-2 illustrates what occurs in the subsurface during a typical hydraulic fracturing event. This diagram shows the initial fracture creation, fracture propagation, proppant placement, and the subsequent fracturing fluid recovery/groundwater extraction stage of the CBM production process. The actual extraction of CBM generally begins after a period of fluid recovery/groundwater extraction. The hydraulically created fracture acts as a conduit in the rock or coal formation, allowing the CBM to flow more freely from the coal seams, through the fracture system, and to the production well where the gas is pumped to the surface.

To create or enlarge fractures, a thick fluid, typically water-based, is pumped into the coal seam at a gradually increasing rate and pressure. Eventually the coal seam is unable to accommodate the fracturing fluid as quickly as it is injected. When this occurs, the pressure is high enough that the coal fractures along existing weaknesses within the coal (steps 1 and 2 of Figure ES-1). Along with the fracturing fluids, sand (or some other propping agent or "proppant") is pumped into the fracture so that the fracture remains "propped" open even after the high fracturing pressures have been released. The resulting proppant-containing fracture serves as a conduit through which fracturing fluids and groundwater can more easily be pumped from the coal seam (step 3 of Fig. ES-1).

To initiate CBM production, groundwater and some of the injected fracturing fluids are pumped out (or "produced" in the industry terminology) from the fracture system in the coal seam (step 4 of Figure ES-1). As pumping continues, the pressure eventually decreases enough so that methane desorbs from the coal, flows toward, and is extracted through the production well (step 5 of Figure ES-1). In contrast to conventional gas production, the amount of water extracted declines proportionally with increasing CBM production. In some basins, huge volumes of groundwater are extracted from the production well to facilitate the production of CBM.

Figure ES-2. A Graphical Representation of the Hydraulic Fracturing Process in Coalbed Methane Wells

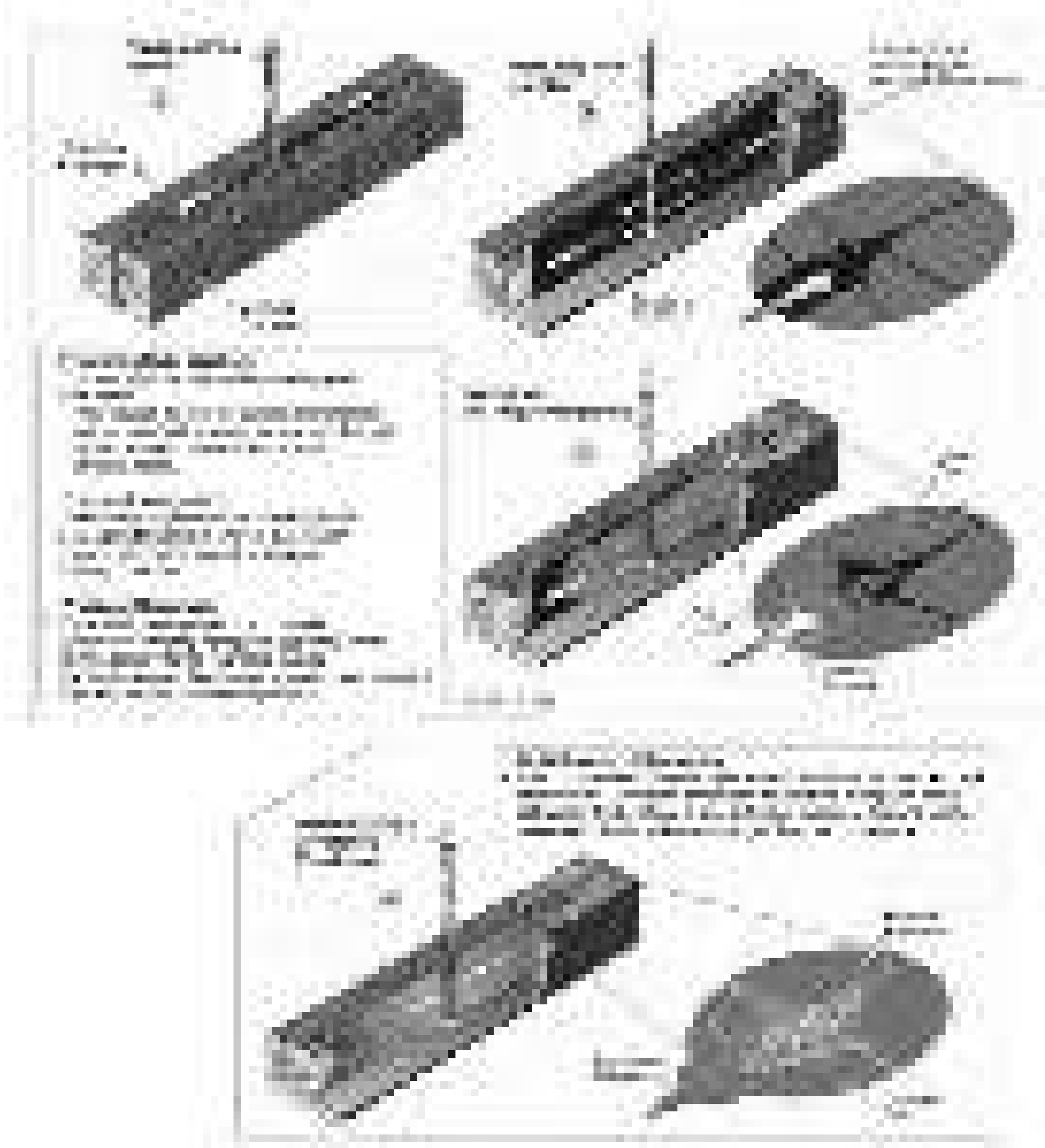


Figure ES-2. A Graphical Representation of the Hydraulic Fracturing Process in Coalbed Methane Wells (Continued)

ES-3 Why Did EPA Evaluate Hydraulic Fracturing?

SDWA requires EPA and EPA-authorized states to have effective programs to prevent underground injection of fluids from endangering USDWs (42 U.S.C. 300h et seq.). Underground injection is the subsurface emplacement of fluids through a well bore (42 U.S.C. 300h(d)(1)). Underground injection endangers drinking water sources if it may result in the presence of any contaminant in underground water which supplies or can reasonably be expected to supply any public water system, and if the presence of such a contaminant may result in such system's noncompliance with any national primary drinking water regulation (i.e., maximum contaminant levels (MCLs)) or may otherwise adversely affect the health of persons (42 U.S.C. 300h(d)(2)). SDWA's regulatory authority covers underground injection practices, but the Act does not grant authority for EPA to regulate oil and gas production.

In 1997, the Eleventh Circuit Court ruled, in *LEAF v. EPA* [LEAF v. EPA, 118F.3d 1467 (11th Circuit Court of Appeals, 1997)], that because hydraulic fracturing of coalbeds to produce methane is a form of underground injection, Alabama's EPA-approved Underground Injection Control (UIC) Program must effectively regulate this practice. In the wake of the Eleventh Circuit's decision, EPA decided to assess the potential for hydraulic fracturing of CBM wells to contaminate USDWs. EPA's decision to conduct this study was also based on concerns voiced by individuals who may be affected by CBM development, Congressional interest, and the need for additional information before EPA could make any further regulatory or policy decisions regarding hydraulic fracturing.

The Phase I study is tightly focused to address hydraulic fracturing of CBM wells and does not include other hydraulic fracturing practices (e.g., those for petroleum-based oil and gas production) because: (1) CBM wells tend to be shallower and closer to USDWs than conventional oil and gas production wells; (2) EPA has not heard concerns from citizens regarding any other type of hydraulic fracturing; and (3) the Eleventh Circuit litigation concerned hydraulic fracturing in connection with CBM production. The study also does not address potential impacts of non-injection related CBM production activities, such as impacts from groundwater removal or production water discharge. EPA did identify, as part of the fact-finding process, citizen concerns regarding groundwater removal and production water.

ES-4 What Was EPA's Project Approach?

Based on public input, EPA decided to carry out this study in discrete phases to better define its scope and to determine if additional study is needed after assessing the results of the preliminary phase(s). EPA designed the study to have three possible phases, narrowing the focus from general to more specific as findings warrant. This report describes the findings from Phase I of the study. The goal of EPA's hydraulic fracturing Phase I study was to assess the potential for contamination of USDWs due to the injection of hydraulic fracturing fluids into CBM wells and to determine based on these findings, whether further study is warranted.

Phase I is a fact-finding effort based primarily on existing literature. EPA reviewed water quality incidents potentially associated with CBM hydraulic fracturing, and evaluated the theoretical potential for CBM hydraulic fracturing to affect USDWs. EPA researched over 200 peer-reviewed publications, interviewed approximately 50 employees from industry and state or local government agencies, and communicated with approximately 40 citizens and groups who are concerned that CBM production affected their drinking water wells.

For the purposes of this study, EPA assessed USDW impacts by the presence or absence of documented drinking water well contamination cases caused by CBM hydraulic fracturing, clear and immediate contamination threats to drinking water wells from CBM hydraulic fracturing, and the potential for CBM hydraulic fracturing to result in USDW contamination based on two possible mechanisms as follows:

1. The direct injection of fracturing fluids into a USDW in which the coal is located (Figure ES-3), or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. The creation of a hydraulic connection between the coalbed formation and an adjacent USDW (Figure ES-4).

Figure ES-3. Hypothetical Mechanisms - Direct Fluid Injection into a USDW (Where Coal Lies Within a USDW or USDWs)

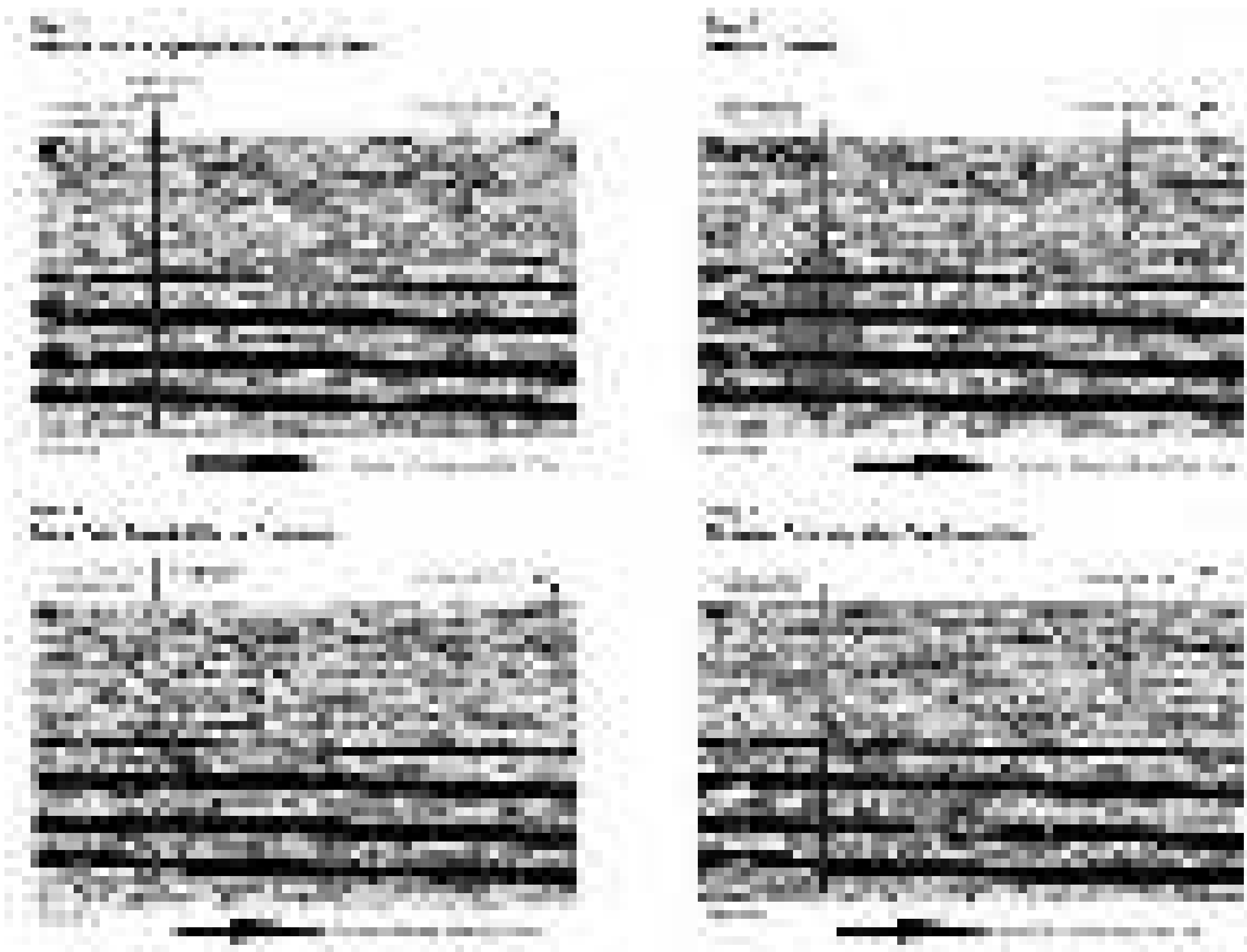
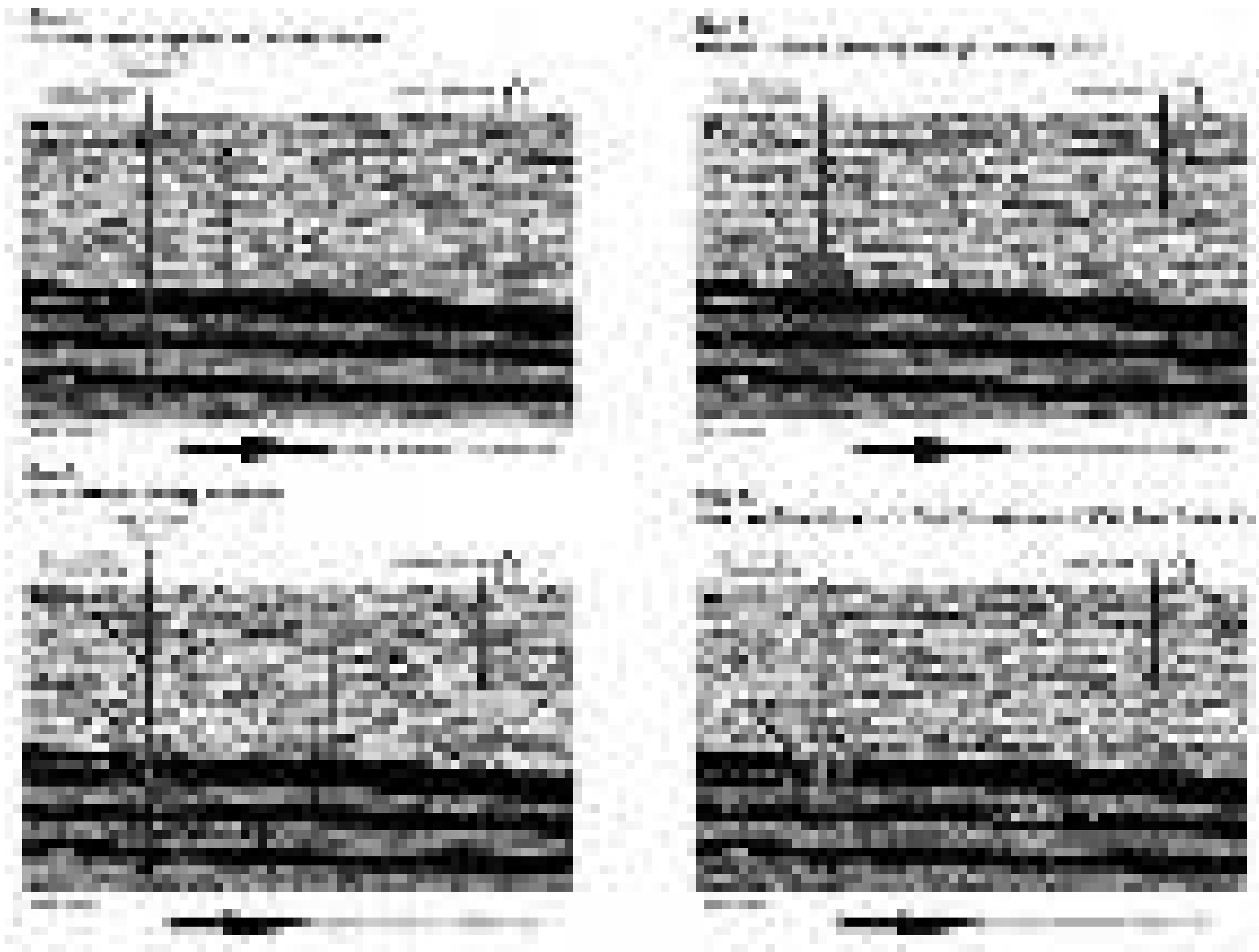


Figure ES-4. Hypothetical Mechanisms - Fracture Creates Connection to USDW



ES-5 How Do Fractures Grow?

In many CBM-producing regions, the target coalbeds occur within USDWs, and the fracturing process injects “stimulation” fluids directly into the USDWs. In other production regions, target coalbeds are adjacent to the USDWs (i.e., either higher or lower in the geologic section). Because shorter fractures are less likely to extend into a USDW or connect with natural fracture systems that may transport fluids to a USDW, the extent to which fractures propagate vertically influences whether hydraulic fracturing fluids could potentially affect USDWs.

The extent of the fractures is difficult to predict because it is controlled by the characteristics of the geologic formation (including the presence of natural fractures), the fracturing fluid used, the pumping pressure, and the depth at which the fracturing is being performed. Fracture behavior through coals, shales, and other geologic strata commonly present in coal zones depends on site-specific factors such as the relative thickness and in-situ stress differences between the target coal seam(s) and the surrounding geologic strata, as well as the presence of pre-existing natural fractures. Often, a high stress contrast between adjacent geologic strata results in a barrier to fracture propagation. An example of this would be where there is a geologic contact between a coalbed and an overlying, thick, higher-stress shale.

Another factor controlling fracture height can be the highly cleated nature of some coalbeds. In some cases, highly cleated coal seams will prevent fractures from growing vertically. When the fracturing fluid enters the coal seam, it is contained within the coal seam's dense system of cleats and the growth of the hydraulic fracture will be limited to the coal seam (see Appendix A).

Deep vertical fractures can propagate vertically to shallower depths and develop a horizontal component (Nielsen and Hansen, 1987, as cited in Appendix A: DOE, Hydraulic Fracturing). In the formation of these "T-fractures," the fracture tip may fill with coal fines or intercept a zone of stress contrast, causing the fracture to turn and develop horizontally, sometimes at the contact of the coalbed and an overlying formation. (Jones et al., 1987; Morales et al., 1990). For cases where hydraulically induced fractures penetrate into, or sometimes through, formations overlying coalbeds, they are most often attributed to the existence of pre-existing natural fractures or thinly inter-bedded layering.

ES-6 What Is in Hydraulic Fracturing Fluids?

Fracturing fluids consist primarily of water or inert foam of nitrogen or carbon dioxide. Other constituents can be added to fluids to improve their performance in optimizing fracture growth. Components of fracturing fluids are stored and mixed on-site. Figures ES-5 and ES-6 show fluids stored in tanks at CBM well locations.

During a hydraulic fracturing job, water and any other additives are pumped from the storage tanks to a manifold system placed on the production wells where they are mixed and then injected under high pressure into the coal formation (Figure ES-6). The hydraulic fracturing in CBM wells may require from 50,000 to 350,000 gallons of fracturing fluids, and from 75,000 to 320,000 pounds of sand as proppant (Holditch et al., 1988 and 1989; Jeu et al., 1988; Hinkel et al., 1991; Holditch, 1993; Palmer et al., 1991, 1993a, and 1993b). More typical injection volumes, based on average injection volume data provided by Halliburton for six basins, indicate a maximum average injection volume of 150,000 gallons of fracturing fluids per well, with a median average injection volume of 57,500 gallons per well (Halliburton, Inc., 2003).

Figure ES-5. Water used for the fracturing fluid is stored on-site in large, upright storage tanks and in truck-mounted tanks.

EPA reviewed material safety data sheets to determine the types of additives that may be present in fracturing fluids. Water or nitrogen foam frequently constitutes the solute in fracturing fluids used for CBM stimulation. Other components of fracturing fluids contain benign ingredients, but in some cases, there are additives with constituents of potential concern. Because much more gel can be dissolved in diesel fuel as compared to water, the use of diesel fuel increases the efficiency in transporting proppant in the fracturing fluids. Diesel fuel is the additive of greatest concern because it introduces BTEX compounds, which are regulated by SDWA.

A thorough discussion of fracturing fluid components and fluid movement is presented in Chapter 4.

Figure ES-6. The fracturing fluids, additives, and proppant are pumped from the storage tanks to a manifold system placed on the wellhead where they are mixed just prior to injection.

ES-7 Are Coalbeds Located within USDWs?

EPA reviewed information on 11 major coal basins to determine if coalbeds are co-located with USDWs and to understand the CBM activity in the area. If coalbeds are located within USDWs, then any fracturing fluids injected into coalbeds have the potential to contaminate the USDW. As described previously, a USDW is not necessarily currently used for drinking water and may contain groundwater unsuitable for drinking without treatment. EPA found that 10 of the 11 basins may lie, at least in part, within USDWs. Table ES-2 identifies coalbed basin locations in relation to USDWs and summarizes evidence used as the basis for the conclusions.

ES-8 Did EPA Find Any Cases of Contaminated Drinking Water Wells Caused by Hydraulic Fracturing in CBM Wells?

EPA did not find confirmed evidence that drinking water wells have been contaminated by hydraulic fracturing fluid injection into CBM wells. EPA reviewed studies and follow-up investigations conducted by state agencies in response to citizen reports that CBM production resulted in water quality and quantity incidents. In addition, EPA received reports from concerned citizens in each area with significant CBM development. These complaints pertained to the following basins:

- San Juan Basin (Colorado and New Mexico);
- Powder River Basin (Wyoming and Montana);
- Black Warrior Basin (Alabama); and
- Central Appalachian Basin (Virginia and West Virginia).

Examples of concerns and claims raised by citizens include:

- Drinking water with strong, unpleasant taste and odor.
- Impacts on fish, and surrounding vegetation and wildlife.
- Loss of water in wells and aquifers, and discharged water creating artificial ponds and swamps not indigenous to region.

Water quantity complaints were the most predominant cause for complaint by private well owners. After reviewing data and incident reports provided by states, EPA sees no conclusive evidence that water quality degradation in USDWs is a direct result of injection of hydraulic fracturing fluids into CBM wells and subsequent underground movement of these fluids. Several other factors may contribute to groundwater problems, such as various aspects of resource development, naturally occurring conditions, population growth, and historical well-completion or abandonment practices. Many of the incidents that were reported (such as water loss and impacts on nearby flora and fauna from discharge of produced water) are beyond the authorities of EPA under SDWA and the scope of Phase I of this study.

ES-9 What Are EPA's Conclusions?

Based on the information collected and reviewed, EPA has determined that the injection of hydraulic fracturing fluids into CBM wells poses little or no threat to USDWs. Continued investigation under a Phase II study is not warranted at this time.

As proposed in the Final Study Design (April 2001), Phase I of the study was a limited–scope assessment in which EPA would:

- Gather existing information to review hydraulic fracturing processes, practices, and settings;

- Request public comment to identify incidents that have not been reported to EPA;

- Review reported incidents of groundwater contamination and any follow-up actions or investigations by other parties (state or local agencies, industry, academia, etc.); and,

- Make a determination regarding whether further investigation is needed, based on the analysis of information gathered through the Phase I effort.

EPA's approach for evaluating the potential threat to USDWs was an extensive information collection and review of empirical and theoretical data. EPA reviewed incidents of drinking water well contamination believed to be associated with hydraulic fracturing and found no confirmed cases that are linked to fracturing fluid injection into CBM wells or subsequent underground movement of fracturing fluids. Although thousands of CBM wells are fractured annually, EPA did not find confirmed evidence that drinking water wells have been contaminated by hydraulic fracturing fluid injection into CBM wells.

EPA also evaluated the theoretical potential for hydraulic fracturing to affect USDWs through one of two mechanisms:

1. Direct injection of fracturing fluids into a USDW in which the coal is located, or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. Creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

Regarding the question of injection of fracturing fluids directly into USDWs, EPA considered the nature of fracturing fluids and whether or not coal seams are co-located with USDWs. Potentially hazardous chemicals may be introduced into USDWs when fracturing fluids are used in operations targeting coal seams that lie within USDWs. In particular, diesel fuel contains BTEX compounds, which are regulated under SDWA. However, the threat posed to USDWs by the introduction of some fracturing fluid constituents is reduced significantly by the removal of large quantities of groundwater (and injected fracturing fluids) soon after a well has been hydraulically fractured. In fact, CBM production is dependent on the removal of large quantities of groundwater. EPA believes that this groundwater production, combined with the mitigating effects of dilution and dispersion, adsorption, and potentially biodegradation, minimize the possibility that chemicals included in the fracturing fluids would adversely affect USDWs.

Because of the potential for diesel fuel to be introduced into USDWs, EPA requested, and the three major service companies agreed to, the elimination of diesel fuel from hydraulic fracturing fluids that are injected directly into USDWs for CBM production (USEPA, 2003). Industry representatives estimate that these three companies perform approximately 95 percent of the hydraulic fracturing projects in the United States.

In evaluating the second mechanism, EPA considered the possibility that hydraulic fracturing could cause the creation of a hydraulic connection to an adjacent USDW. The low permeability of relatively unfractured shale may help to protect USDWs from being affected by hydraulic fracturing fluids in some basins. If sufficiently thick and relatively unfractured shale is present, it may act as a barrier not only to fracture height growth, but also to fluid movement. Shale's ability to act as a barrier to fracture height growth is primarily due to the stress contrast between the coalbed and the shale. Another factor controlling fracture height can be the highly cleated nature of some coalbeds. In some cases, when the fracturing fluid enters the coal seam, it is contained within the coal seam's dense system of cleats and the growth of the hydraulic fracture will be limited to the coal seam (see Appendix A).

Some studies that allow direct observation of fractures (i.e., mined-through studies) indicate many fractures that penetrate into, or sometimes through, one or more formations overlying coalbeds can be attributed to the existence of pre-existing natural fractures. However, given the concentrations and flowback of injected fluids, and the mitigating effects of dilution and dispersion, adsorption, and potentially biodegradation, EPA does not believe that possible hydraulic connections under these circumstances represent a significant potential threat to USDWs.

It is important to note that states with primary enforcement authority (primacy) for their UIC Programs implement and enforce their regulations, and have the authority under SDWA to place additional controls on any injection activities that may threaten USDWs. States may also have additional authorities by which they can regulate hydraulic fracturing. With the expected increase in CBM production, the Agency is committed to working with states to monitor this issue.

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List of Acronyms and Abbreviations

μg/g	*Micrograms per gram*
μg/L	*Micrograms per liter*
ADEM	*Alabama Department of Environmental Management*
Bbl/min	*Barrel per minute*
Bcf	*Billion cubic feet*
Bgs	*Below ground surface*
BHP	*Bottom hole pressure*
BLM	*Bureau of Land Management*
BTEX	*Benzene, toluene, ethylbenzene, xylenes*
Btu	*British thermal unit*
CBM	*Coalbed methane*
CDH	*Colorado Department of Health*
CCL	*Contaminant Candidate List*
CDWR	*Colorado Division of Water Resources*
CFR	*Code of Federal Regulations*
CMHPG	*Carboxymethylhydroxypropylguar*
COGCC	*Colorado Oil and Gas Conservation Commission*
DASC	*Data Access and Support Center*
DNR	*Department of Natural Resources*
DOE	*Department of Energy*
EPA	*Environmental Protection Agency*
g	*Gram*
g/mL	*Grams per milliliter*
GRI	*Gas Research Institute*
GTI	*Gas Technology Institute*
GSA	*Geological Survey of Alabama*
HCl	*Hydrochloric acid*

HEC	*Hydroxyethylcellulose*
HPG	*Hydroxypropylguar*
KCl	*Potassium chloride*
L	Liter
LEAF	*Legal Environmental Assistance Foundation*
Mcf	*Million cubic feet*
MCL	*Maximum contaminant level*
md	*Millidarcy*
mg/L	*Milligrams per liter*
mL	*Milliliter*
MOA	*Memorandum of Agreement*
MSDS	*Material Safety Data Sheet*
MTBE	*Methyl tert butyl ether*
NMOCD	*New Mexico Oil Conservation Division*
NPDEA	*National Pollution Discharge Elimination System*
OGB	*Oil and Gas Board*
OGWDW	*Office of Ground Water and Drinking Water*
P3D	*Pseudo 3 Dimensional*
PAH	*Polynuclear aromatic hydrocarbons*
POM	*Polycyclic organic matter*
ppm	*Parts per million*
PRBRC	*Powder River Basin Resource Council*
PRCMIC	*Powder River Coalbed Methane Information Council*
psi	*Pounds per square inch*
SDWA	*Safe Drinking Water Act*
SEO	*State Engineer's Office*
SJRA	*San Juan Regional Authority*
TBEG	*Texas Bureau of Economic Geology*

Tcf	*Trillion cubic feet*
TDS	*Total dissolved solids*
TGD	*Tennessee Geology Division*
UIC	*Underground Injection Control*
USBM	*United States Bureau of Mines*
USDW	*Underground Source of Drinking Water*
USGS	*United States Geological Survey*
VDMME	*Virginia Division of Oil and Gas, within the Department of Mines, Minerals and Energy*
wt.	*Weight*

Glossary

Adsorption	*Adhesion of gas molecules, ions or molecules in solution to the surface of solid bodies with which they are in contact.*
Alluvial aquifer	*A water-bearing deposit of unconsolidated material (e.g., sand and gravel) left behind by a river or other flowing water.*
Amphoteric	*Having both basic and acidic properties.*
Anaerobic Bacteria	*Bacteria that thrive in oxygen-poor environments.*
Anisotropic	*Having some physical property that varies with direction from a given location.*
Annulus	*The space between the casing (the material that is used to keep the well stable; typically this material is steel) in a well and the wall of the hole, or between two concentric strings of casing, or between casing and tubing.*
Anticline	*A fold of layered, sedimentary rocks whose core contains stratigraphically older rocks, the shape of the fold is generally convex upward.*
Aureole	*A ring surrounding a volcanic intrusion where the surrounding rock has been altered.*
Azimuth	*The direction of a horizontal line as measured on an imaginary horizontal circle.*
Bedrock aquifer	*An aquifer located in the solid rock underlying unconsolidated surface materials (i.e., sediment). Solid rock can bear water when it is fractured.*
Billion cubic feet	*A unit typically used to define gas production volumes in the coalbed methane industry; 1 Bcf is roughly equivalent to the volume of gas required to heat approximately 12,000 households for one year (based on the Department of Energy's average household energy consumption statistic, 2001).*
Biogenic	*A direct product of the physiological activities of organisms.*
Bituminous	*From the base word bitumen, referring to a general term for various solid and semi-solid hydrocarbons that are able to join together and are soluble in carbon bisulfide (e.g., asphalts).*
Breaker	*A fracturing fluid additive that is added to break down the viscosity of the fluid.*
Breccia	*A coarse-grained clastic rock composed of angular broken rock fragments held together by a mineral cement or a fine-grained matrix.*
Brecciated	*Consisting of angular fragments cemented together.*
Btu	*British thermal unit; a unit of measure used to define energy.*
Butt Cleat	*The coal cleat set that abuts into face cleats.*
Capture Zone	*The portion of an aquifer that contributes water to a particular pumping well.*

Cavitation Cycling	*Also known as cavity completion, an alternative completion technique to hydraulic fracturing, in which a cavity is generated by alternately pumping in nitrogen and blowing down pressure.*
Cleats	*Natural fractures in coal that often occur in systematic sets, through which gas and water can flow.*
CMHPG	*Carboxymethyl hydroxypropylguar; a form of guar gel.*
Craton	*A part of the earth's crust that has attained stability and has been relatively undeformed for a long time; the term is restricted to continents, and includes both shield and platform.*
Crosslinker	*An additive that when added to a linear gel, will create a complex, high viscosity, pseduoplastic fracturing fluid.*
Crosslinked Gel	*A gel to which a crosslinker has been added (see crosslinker).*
Darcy	*A measure of the permeability of rock or sediment.*
Desorption	*Liberation of tightly held methane gas molecules previously bound to the solid surface of the coal.*
Epiclastic	*Formed from the fragments or particles broken away (by weathering and erosion) from pre-existing rocks to form an altogether new rock in a new place.*
Evapotranspiration	*The portion of precipitation returned to the air through evaporation and transpiration.*
Face Cleat	*A coal cleat set that is through-going and continuous.*
Flowback	*The process of causing fluid to flow back to the well out of a fracture after a hydraulic fracturing event is completed.*
Fracture Conductivity	*The capability of the fracture to conduct fluids under a given hydraulic head difference.*
Geophone	*A seismic detector, placed on or in the ground, that responds to ground motion at its point of location.*
Graben	*An elongate, down-dropped block that is bounded by nearly parallel faults on both sides.*
Guar	*Organic powder thickener, typically used to make viscous fracturing fluids, completely soluble in hot and cold water, insoluble in oils, grease and hydrocarbons.*
HCl	*Molecular formula for hydrochloric acid; can be used in diluted form in the hydraulic fracturing process to fracture limestone formations and to clean up perforations in coalbed methane fracturing treatments.*
HEC	*Hydroxyethylcellulose; a form of guar gel.*
Hydraulic Conductivity	*(see permeability)*
Injectate	*In relation to the coalbed methane industry, this is the fracturing fluid injected into a coalbed methane well.*
Isopach	*A line drawn on a map through points of equal true thickness of a designated stratigraphic unit or group of stratigraphic units.*

Isotopic *Rocks formed in the same environment, i.e. in the same sedimentary basin or geologic province.*

Isotropic *A medium, such as unconsolidated sediments or a rock formation, whose properties are the same in all directions.*

KCl *Molecular formula for potassium chloride.*

Lacustrine *Pertaining to, produced by, or formed in a lake or lakes.*

Laminar Flow *Water flow in which the stream lines remain distinct and the flow direction at every point remains unchanged with time; non-turbulent flow.*

Leakoff *The magnitude of pressure exerted on a formation that causes fluid to be forced into the formation. In common usage, leakoff is often considered the movement of fluid out of primary fractures and into a geologic formation, either through small existing permeable paths (connected pores and natural fracture networks) or through small pathways created or enlarged in the rock through the fracturing process.*

Lenticular *Pertaining to a discontinuous, lens-shaped (saucer-shaped) stratigraphic body.*

Linear Gel *A simple guar-based fracturing fluid usually formulated using guar and water with additives or guar with diesel fuel.*

Lithology *The description of rocks based on mineralogic composition and texture.*

Millidarcy *The customary unit of measurement of fluid permeability; equivalent to 0.001 Darcy.*

Mcf *Million cubic feet; a unit typically used to define gas production volumes in the coalbed methane industry; 1 Mcf is roughly equivalent to the volume of gas required to heat approximately 12 households for one year (based on the Department of Energy's average household energy consumption statistic, 2001); Mcf can sometimes represent 1,000 cubic feet.*

mg/L *Milligrams per liter; typically used to define concentrations of a dissolved compound in a fluid.*

Mined-through studies *Mined-through studies are projects in which coalbeds have been actually mined through (i.e., the coal has been removed) so that remaining coal and surrounding rock can be inspected, after the coalbeds have been hydraulically fractured. These studies provide unique subsurface access to investigate coalbeds and surrounding rock after hydraulic fracturing.*

Moduli *Plural of modulus (often referred to as bulk modulus), the ratio of stress to strain, abbreviated as "k". The bulk modulus is an elastic constant equal to the applied stress divided by the ratio of the change in volume to the original volume of a body.*

Overthrust *A low-angle thrust fault of large scale, with total displacement (lateral or vertical) generally measured in kilometers.*

Pad *An initial volume of fluid that is used to initiate and propagate a fracture before a proppant is placed.*

Paleochannels *Old or ancient river channels preserved in the subsurface as lenticular sandstones.*

Permeability	*The capacity of a porous rock, sediment, or soil to transmit a fluid; it is a measure of the relative ease of fluid flow under equal pressure and from equal elevations.*
Physiographic	*A region of which all parts are similar in geologic structure and climate and which has had a unified geomorphic history; its relief features differ significantly from those of adjacent regions.*
Play	*A productive coalbed methane formation, or a productive oil or gas deposit.*
Potentiometric	*The total head of ground water, defined by the level to which water will rise in a well.*
ppm	*Parts per million; typically used to define concentrations of a dissolved compound in a fluid; equivalent to 1 mg/L.*
Primacy	*The right to self-establish, self-enforce and self-regulate environmental standards; this enforcement responsibility is granted by EPA to States and Indian Tribes.*
Primary porosity	*The porosity preserved from some time between sediment deposition and the final rock-forming process; (e.g., the spaces between grains of sediment).*
Proppant	*Granules of sand, ceramic or other minerals that are wedged within the fracture and act to "prop" it open after the fluid pressure from fracture injection has dissipated.*
psi	*Pounds per square inch; a unit of pressure.*
Rank	*The degree of metamorphism in coal; the basis of coal classification into a natural series from lignite to anthracite.*
Screen-out	*Term used to describe a fracturing job where proppant placement has failed.*
Secondary porosity	*The porosity created through alteration of rock, commonly by processes such as, dissolution and fracturing.*
Semianthracite	*Term used to identify coal rank; specifically representing coal that possesses a fixed-carbon content of 86% to 92%.*
Stratigraphy	*The study of rock strata; concerning all characteristics and attributes of rocks and their interpretation in terms of mode of origin and geologic history.*
Subbituminous	*A black coal, intermediate in rank between lignite and bituminous.*
Subgraywacke	*A sedimentary rock (sandstone) that contains less feldspar, and more and better-rounded quartz grains than graywacke; intermediate in composition between graywacke and orthoquartzite; it is lighter-colored and better-sorted, and has less matrix than greywacke.*
Surficial	*Pertaining to or lying in or on a surface; specific to the surface of the earth.*
Syncline	*A fold of layered, sedimentary rocks whose core contains stratigraphically younger rocks; shape of fold is generally concave upward.*
Tcf	*Trillion cubic feet; a unit typically used to define gas production volumes in the coalbed methane industry; 1 Tcf is roughly equivalent to the volume of gas required to heat approximately 12 million households for one year (based on the Department of Energy's average household energy consumption statistic, 2001).*

Thermogenic *A direct product of high temperatures, (e.g. Thermogenic methane).*

Toughness *The point at which enough stress intensity has been applied to a rock formation, so that a fracture initiates and propagates.*

Transmissivity *A measure of the amount of water that can be transmitted horizontally through a unit width by the full saturated thickness of the aquifer under a hydraulic gradient of one.*

Up-warp *The uplift of a region; usually a result of the release of isostatic pressure, e.g. the melting of an ice sheet.*

Viscosity *The property of a substance to offer internal resistance to flow; internal friction.*

Volcaniclastic *Composed of fragments or particles, and related to volcanic processes either by forming as the result of explosive processes or due to the weathering and erosion of volcanic rocks.*

Chapter 1
Introduction

Section 1421 of SDWA tasks EPA with protecting USDWs for all current and future drinking water supplies across the country (see section 1.3 for the complete definition of a USDW). EPA's UIC Program is responsible for ensuring that fluids injected into the ground (for purposes including waste disposal, oil field brine disposal, enhanced recovery of oil and gas, mining, and emplacement of other fluids) do not endanger USDWs.

EPA, through its UIC Program, conducted a fact-finding effort based primarily on existing literature. The goal of this study was to assess the potential for contamination of USDWs due to the injection of hydraulic fracturing fluids into coalbed methane wells and to determine, based on these findings, whether further study is warranted. For the purposes of this study, EPA assessed USDW impacts by the presence or absence of documented drinking water well contamination cases caused by coalbed methane hydraulic fracturing, clear and immediate contamination threats to drinking water wells from coalbed methane hydraulic fracturing, and the potential for coalbed methane hydraulic fracturing to result in USDW contamination based on two possible mechanisms as follows:

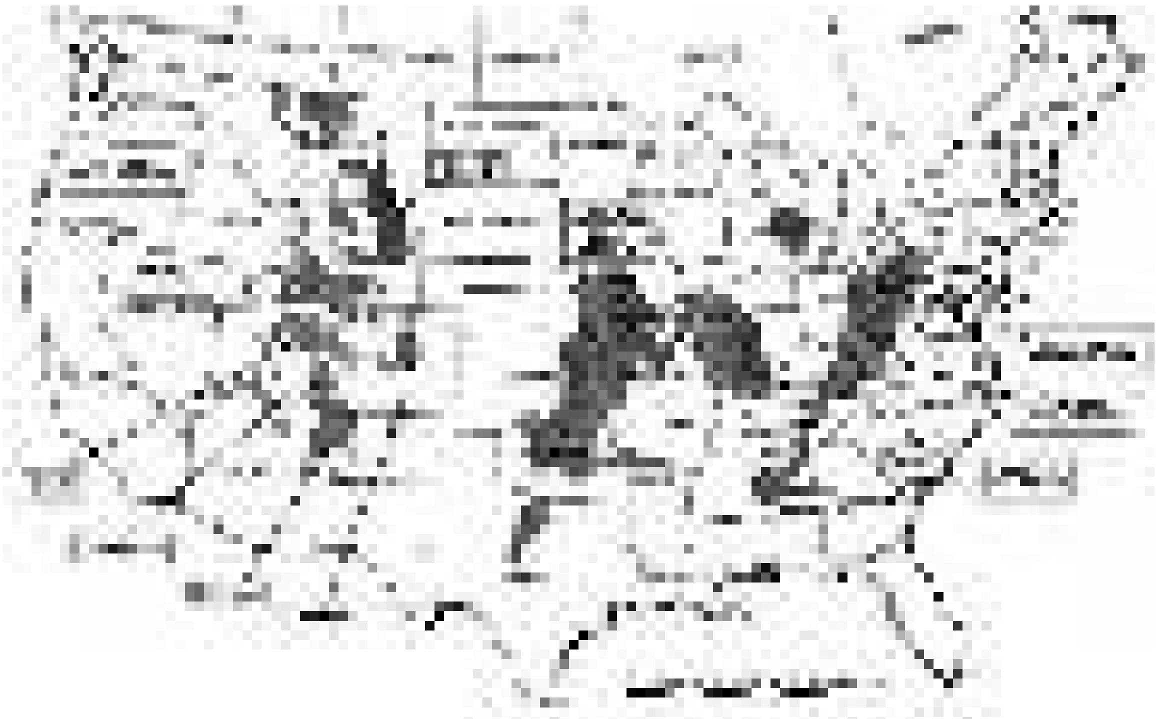
1. Direct injection of fracturing fluids into a USDW in which the coal is located, or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. Creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

EPA obtained information for this study from literature searches, field visits, a review of reported groundwater contamination incidents in areas where coalbed methane is produced, and solicitation of information from the public on any impacts to groundwater believed to be associated with hydraulic fracturing.

EPA also reviewed 11 major coal basins throughout the United States to determine if coalbeds are co-located with USDWs and to understand the coalbed methane activity in the area (Figure 1-1). The basins shown in red have the highest coalbed methane production volumes. They are the Powder River Basin in Wyoming and Montana, the San Juan Basin in Colorado and New Mexico, and the Black Warrior Basin in Alabama. Hydraulic fracturing is or has been used to stimulate coalbed methane wells in all basins, although it has not frequently been used in the Powder River, Sand Wash, or Pacific Coal Basins.

Figure 1-1. Locus Map of Major United States Coal Basins



1.1 EPA's Rationale for Conducting This Study

Although coalbed methane has many environmental advantages over traditional energy sources, concerns have been raised regarding the environmental impacts of coalbed methane production. Coalbed methane production in certain areas has led to groundwater depletion and production water discharge issues (i.e., issues that are not associated with the quality of USDWs). Citizens, state agencies, producers, and the regional EPA offices in those areas are working in concert to better understand and mitigate these potential problems.

This study examines the potential for hydraulic fracturing fluid injection into coalbed methane wells to contaminate USDWs. EPA conducted this study in response to allegations that hydraulic fracturing of coalbed methane wells has affected the quality of groundwater (i.e., issues that are associated with the mandates of the UIC Program). State oil and gas agencies receiving such complaints have indicated that, based on their investigations, hydraulic fracturing of coalbed methane wells has not contributed to water quality degradation in USDWs.

In response to an Eleventh Circuit Court of Appeals (hereafter, "the Court") decision [*LEAF v. EPA*, 118F.3d 1467 (11th Cir, 1997)], the State of Alabama recently supplemented its rules governing the hydraulic fracturing of wells to include additional requirements to protect USDWs during the hydraulic fracturing of coalbeds for methane production. Prior to the Court's decision, EPA had not considered hydraulic fracturing as an underground injection activity, because the Agency did not consider production well stimulation as an activity subject to UIC regulations. Nevertheless, the Court held that the injection of fluids for the purpose of hydraulic fracturing constitutes underground injection as defined under SDWA, that all underground injection must be regulated, and that hydraulic fracturing of coalbed methane wells in Alabama must be regulated under Alabama's UIC program.

In the wake of the Eleventh Circuit Court decision, EPA decided to assess the potential for hydraulic fracturing fluid injection into coalbed methane wells to contaminate USDWs. EPA's decision to conduct this study was also based on concerns voiced by individuals who may be affected by coalbed methane development, Congressional interest, and the need for additional information before EPA could make any further regulatory or policy decisions regarding hydraulic fracturing.

1.2 Overview of Hydraulic Fracturing

Hydraulic fracturing is a technique used by the oil and gas industry to improve the production efficiency of oil and coalbed methane wells. The hydraulic fracturing process uses high hydraulic pressures to initiate a fracture. A hydraulically induced fracture acts as a conduit in the rock or coal formation that allows the oil or coalbed methane to travel more freely from the rock pores to the production well that can bring it to the surface.

In the case of coalbed methane gas production, the gas is not structurally "trapped" under pressure. Rather, most of the coalbed methane is adsorbed within small pores in the "micro-porous matrix" of the coal (Koenig, 1989; Winston, 1990; Close, 1993). When coalbed methane production begins, water is first pumped out (or "produced" in the industry terminology) from the fractures, joints, and cleats (i.e., tiny, disconnected clusters of fractures) in the coal until the pressure declines to the point that methane begins to desorb from the coal matrix itself (Gray, 1987).

To extract the coalbed methane, a production well is drilled through rock layers to intersect the coal seam that contains the coalbed methane. Next, a fracture is created or enlarged in the coal seam to connect the well bore to the coalbed joint/cleat system. To create such a fracture, a thick, water-based fluid is pumped into the coal seam at a gradually increasing rate. At a certain point, the coal seam will not be able to accommodate the fracturing fluid as quickly as it is being injected. When this occurs, the pressure is high enough that the coal gives way, and a fracture is created or an existing fracture is enlarged. To hold the fracture open, a propping agent, usually sand (commonly known as "proppant"), is pumped into the fracture so that when the pumping pressure is released, the fracture does not close completely because the proppant is "propping" it open. The resulting fracture filled with proppant becomes a conduit through which water can flow to the production well, thus depressurizing the coal matrix, allowing for the desorption of methane and its flow towards the production well.

The extent of the fracture in a coalbed is controlled by the characteristics of the geologic formation (including the presence of natural fractures), the fracturing fluid used, the pumping pressure, and the depth at which the fracturing is performed. Whether the fracture grows taller or longer is determined by the properties of the surrounding rock. A hydraulically created fracture will always take the path of least resistance through the coal seam and surrounding formations.

A more comprehensive discussion of the fracturing process and the fracturing fluids/additives used in hydraulic fracturing of coalbed methane wells is presented in Chapters 3 and 4, respectively.

1.3 EPA's Authority to Protect Underground Sources of Drinking Water

SDWA requires EPA and EPA-authorized states to have effective programs to prevent underground injection of fluids from endangering USDWs (42 U.S.C. 300h et seq.). Underground injection is the subsurface emplacement of fluids through a well bore (42 U.S.C. 300h(d)(1)). Underground injection endangers drinking water sources if it may result in the presence of any contaminant in underground water which supplies or can reasonably be expected to supply any public water system, and if the presence of such contaminant may result in such system's noncompliance with any national primary drinking water regulation (i.e., maximum contaminant levels) or may otherwise adversely affect the health of persons (42 U.S.C. 300h(d)(2)). SDWA's regulatory authority extends to underground injection practices; SDWA does not provide a general grant of authority for EPA to regulate oil and gas production.

A USDW is defined in the UIC regulations at 40 CFR 144.3 as an aquifer or a portion of an aquifer that:

"A. 1. Supplies any public water system; or
2. Contains sufficient quantity of groundwater to supply a public water system; and
   i. currently supplies drinking water for human consumption; or
   ii. contains fewer than 10,000 milligrams per liter (mg/L) total dissolved solids (TDS); and
B. Is not an exempted aquifer."

The water quality standard for USDWs is more stringent than EPA's National Secondary Drinking Water Standards for potable water, which cover aesthetic concerns such as taste and odor. These secondary standards recommend a TDS limit of 500 mg/L (40 CFR 143.3).

An accurate understanding of the definition of USDW requires understanding of two other terms: public water system and aquifer exemption.

A public water system is defined at 40 CFR 141.2 as:

> "A system for the provision to the public of water for human consumption through pipes or, after August 5, 1998, other constructed conveyances, if such a system has at least 15 service connections or regularly serves an average of at least twenty-five individuals daily at least 60 days out of the year."

To better quantify the definition of USDW, EPA determined that any aquifer yielding more than 1 gallon per minute can be expected to provide sufficient quantity of water to serve a public water system and therefore falls under the definition of a USDW (U.S. EPA Memorandum, 1993). EPA also assumes that all aquifers contain sufficient quantity of groundwater to supply a public water system, unless proven otherwise through empirical data.

An aquifer exemption may be granted under certain circumstances. According to 40 CFR 144.3, an exempted aquifer meets the definition of a USDW, but has been exempted according to the procedures in 40 CFR 144.7. An aquifer, or portion thereof, can be designated as an exempted aquifer, if it meets the following criteria (40 CFR 146.4):

1. It does not currently serve as a source of drinking water; and,

2. It cannot now and will not in the future serve as a source of drinking water because it is:

   - Mineral, hydrocarbon, or geothermal energy producing, or can be demonstrated to be commercially producible; or
   - Situated at a depth or location which makes recovery of water for drinking water purposes economically or technologically impractical; or
   - So contaminated that it would be economically or technologically impractical to render that water fit for human consumption; or
   - Located over a Class III well mining area subject to subsidence or catastrophic collapse; or,

3. The TDS content of the groundwater is more than 3,000 and less than 10,000 mg/L and, it is not reasonably expected to supply a public water system.

All requests for aquifer exemptions must be approved by the EPA Administrator or an authorized representative. A list of exempted aquifers, for states where such exemptions exist, is maintained by the state agency managing the UIC program or the regional EPA office. A comprehensive list or map identifying all USDWs in every state does not exist. Identification of USDWs is an ongoing effort, as is EPA's consideration of aquifer exemptions. For example, coalbed methane production wells using hydraulic fracturing to stimulate production may be located in areas that coincide with existing aquifer exemptions.

Currently, injection associated with hydraulic fracturing of coalbed methane production wells is regulated only in Alabama under the state UIC program, and that injection activity falls under the category of Class II wells (Alabama Oil and Gas Board, Administrative Code, Oil and Gas Report 1, 400-3). Class II wells include the injection of brines and other fluids that are associated with oil and gas production.

1.4 Potential Effects of Hydraulic Fracturing of Coalbed Methane Wells on USDWs

EPA identified two possible mechanisms by which hydraulic fracturing fluid injection into coalbed methane wells might affect the quality of USDWs:

1. The direct injection of fracturing fluids into a USDW in which the coal is located (Figure 1-2), or injection of fracturing fluids into a coal seam which is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. The creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

Fracturing fluids can be directly or indirectly injected into a USDW, depending on the location of the coalbed relative to a USDW. In many coalbed methane-producing regions, the target coalbeds occur *within* USDWs, and the fracturing process injects stimulation fluids directly into the USDWs (Figure 1-2 at the end of the chapter). In other production regions, target coalbeds are adjacent to the USDWs, which are either higher or lower in the geologic section. EPA investigated the potential for fractures to extend through stratigraphic layers that separate coalbeds and USDWs and the potential for stimulation fluids to indirectly enter a USDW during the fracturing process (Figure 1-3 at the end of the chapter).

Local geologic conditions may interfere with the complete recovery of fracturing fluids injected into a formation. As a result, some of the fracturing fluids may be "stranded" in the USDW (Figures 1-2 and 1-3). Any hazardous constituents in the stimulation fluids could potentially contaminate groundwater in a USDW and any drinking water supplies that rely on the USDW.

1.5 Study Approach

Given the enormous variation in geology among and within coalbed basins in the United States, any initial evaluation of potential impacts by hydraulic fracturing of coalbeds on USDWs at a national level would necessarily be broadly focused. Based on public input, EPA decided to carry out this study in discrete phases to better define its scope and to determine if additional study is needed after assessing the results of the preliminary phase(s). EPA designed the study to have three possible phases, changing the focus from general to more specific as findings warrant.

Phase I of the study is a fact-finding effort based primarily on existing literature to identify and assess the potential threat to USDWs posed by hydraulic fracturing fluid injection into coalbed methane wells. It is designed to determine if site-specific detailed studies, including collection of new data, are needed. An overview of the methodology used for Phase I is provided below; a detailed discussion of this methodology is provided in Chapter 2.

In Phase I, EPA:

- Conducted a literature review for information on hydraulic fracturing processes, hydraulic fracturing fluids and additives, the geologic settings of and the hydraulic fracturing practices used in the 11 major coal basins (Figure 1-1), and the identification of coal seams that are co-located with USDWs.

- Published a request in the *Federal Register* (66 FR 39396 (U.S. EPA, 2001)) for information from the public, as well as governmental and regulatory agencies, regarding incidents of groundwater contamination believed to be associated with hydraulic fracturing of coalbed methane wells.

- Reviewed reported incidents of groundwater contamination and any follow-up actions or investigations by other parties such as state or local agencies, industry, and academia.

- Conducted field visits in three states.

In addition, EPA collaborated with the Department of Energy (DOE) to produce a document that details the technical aspects of hydraulic fracturing in the oil and gas industry. This document is included as Appendix A to this report.

EPA also provided support for a site-specific study, which was conducted by the Geological Survey of Alabama (GSA). This study attempts to address a concern that is central to USDW contamination and drawdown issues: the degree to which flow is confined within coalbeds in coalbed methane fields. Information on the GSA study is available at http://www.gsa.state.al.us/gsa/3DFracpage/3Dfracstudy.htm.

1.6 Stakeholder Involvement

EPA took several steps to fully involve the public and all stakeholders during the study. These steps included:

- Publishing *Federal Register* notices:
    - requesting comments on the study plan (65 FR 45774 (USEPA, 2000));
    - requesting information from the public on any impacts to groundwater believed to be associated with hydraulic fracturing of coalbed methane wells (66 FR 39396 (USEPA, 2001));
    - Requesting comments on the August 2002 draft of the study (67 FR 55249 (USEPA, 2002)).
- Holding a public meeting to obtain additional stakeholder input on the proposed study plan published in the July 2000 *Federal Register* notice (65 FR 45774 (USEPA, 2000))
- Providing periodic updates for stakeholders in the form of written communication.
- Maintaining a Web site where stakeholders can view the project documents and provide information to EPA.

EPA also received and reviewed comments from 105 commenters submitted in response to the August 2002 *Federal Register* notice (67 FR 55249 (USEPA, 2002)), which announced the availability of the August 2002 version of this Phase I study report. EPA incorporated many of these comments into this final Phase I report. A summary of the public comments and EPA's responses is provided in, "Public Comment and Response Summary for the Study on the Potential Impacts of Hydraulic Fracturing of Coalbed Methane Wells on Underground Sources of Drinking Water" (EPA 816-R-04-004), available on EPA's electronic docket.

1.7 Information Contained within This Report

This Phase I report is composed of an executive summary, 7 chapters, 11 attachments, and 2 appendices. The main chapters address the following topics:

- Chapter 2, Study Methodology, discusses in detail EPA's method for collecting information under Phase I of the study.

- Chapter 3, Characteristics of Coalbed Methane Production and Associated Hydraulic Fracturing Practices, discusses the hydraulic fracturing process as it applies to coalbed methane production.

- Chapter 4, Hydraulic Fracturing Fluids, describes the use and nature of hydraulic fracturing fluids and their additives. It also discusses EPA's evaluation of the fate and transport of fracturing fluids that are injected into targeted coal layers during the hydraulic fracturing process.

- Chapter 5, Summary of Coalbed Methane Basin Descriptions, briefly describes each of the 11 major coal basins in the United States and discusses the potential for impacts to USDWs in these basins.

- Chapter 6, Water Quality Incidents, in response to stakeholders' recommendations, summarizes water quality and quantity complaints received from citizens pertaining to hydraulic fracturing, coalbed methane production, and well stimulation.

- Chapter 7, Summary of Findings, summarizes the major findings presented in Chapters 3 through 6.

In addition, Chapters 3 through 6 contain numerous figures and tables to help readers visualize the hydraulic fracturing process and to help summarize some of the key information in the report.

The attachments to the report are a collection of in-depth hydrologic investigations of the 11 coal basins, focusing primarily on the coalbed methane production activities and the relationship between coalbed and USDW locations within these 11 basins. The attachments expand the discussions of Chapter 5 with greater details on the specific geology and gas production activities for the 11 basins.

Appendix A, Hydraulic Fracturing, contains DOE's technical report on hydraulic fracturing. Appendix B, Quality Assurance Protocol, explains the quality assurance and quality control measures EPA used to conduct this study.

Figure 1-2. Hypothetical Mechanism - Direct Fluid Injection Into a USDW (Coal within USDW)

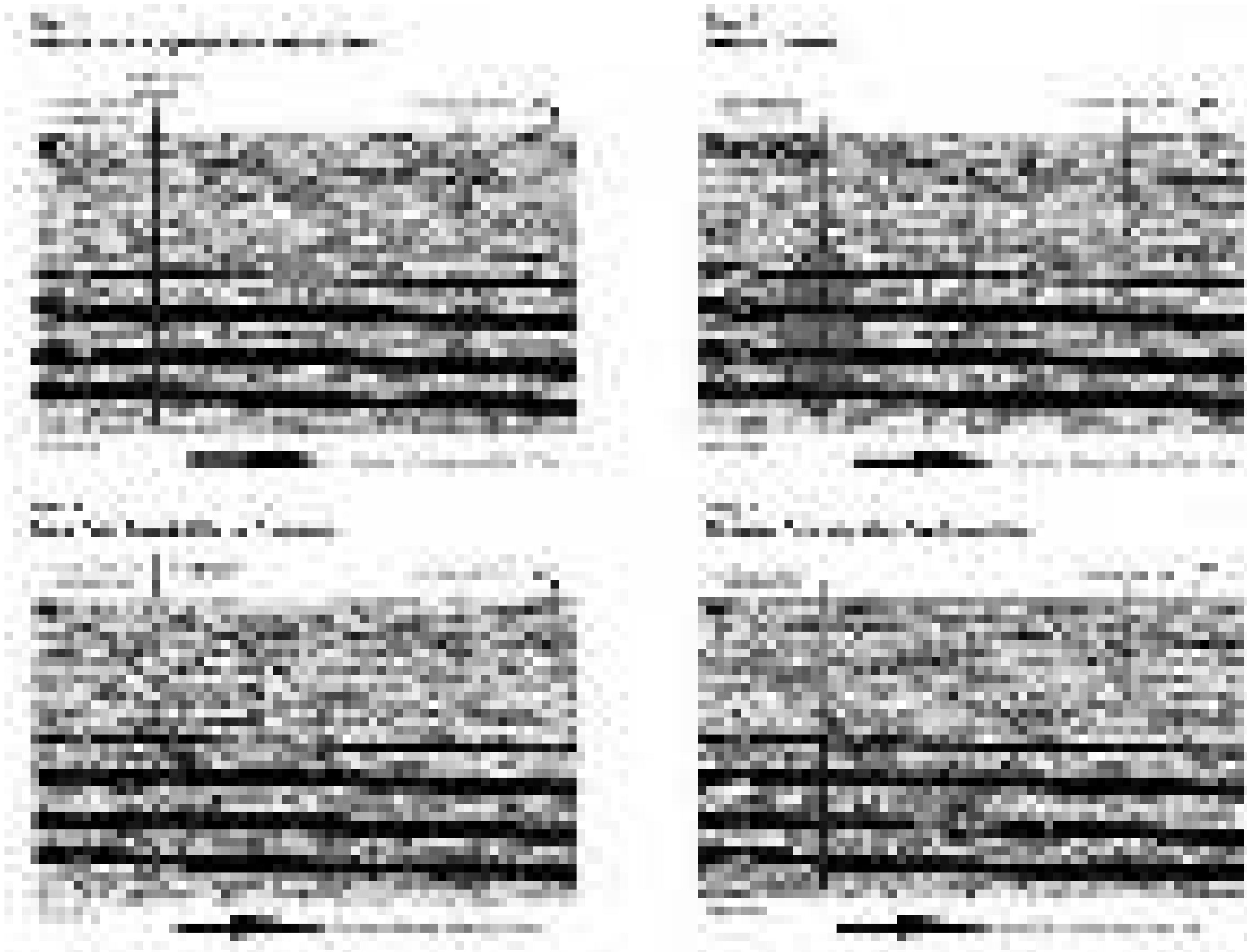
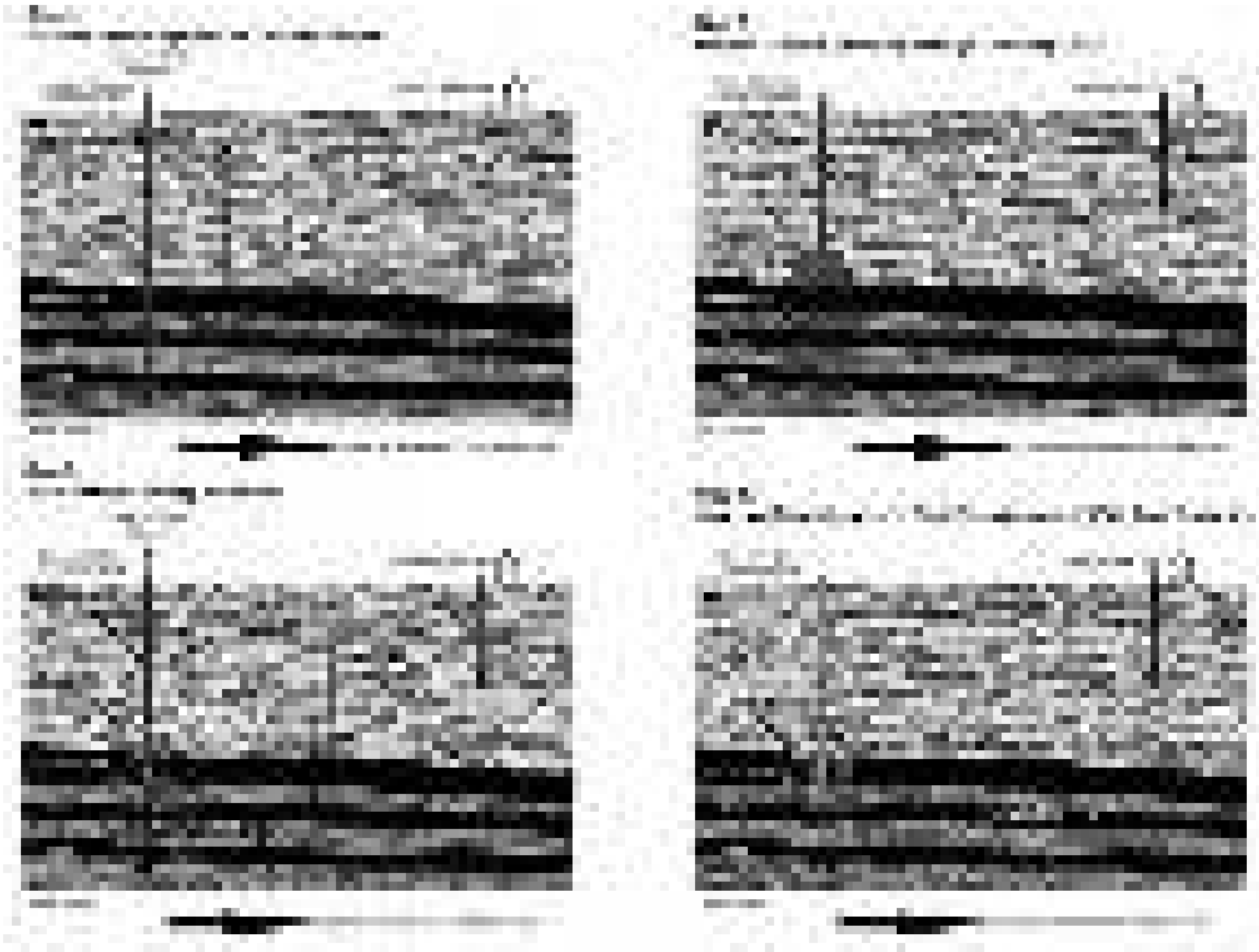


Figure 1-3. Hypothetical Mechanism - Fracture Creates Connection to USDW



Chapter 2
Study Methodology

This chapter outlines EPA's approach for completing Phase I of the study. This chapter describes the development of the study, the information collection and review process that EPA used, and the internal and external review process for the report.

2.1 Overview of the Study Methods

EPA developed the Phase I study methodology to assess the potential for contamination of USDWs due to the injection of hydraulic fracturing fluids into coalbed methane wells, and to determine, based on these findings, whether further study is warranted.

On July 25, 2000, EPA published a *Federal Register* notice (65 FR 45774 (USEPA, 2000)) requesting comment on a conceptual study design in order to receive stakeholder input on how an EPA study should be structured. On August 24, 2000, EPA held a public meeting to obtain additional stakeholder input on the proposed study design. EPA received more than 80 sets of comments from industry, state oil and gas agencies, environmental groups, and individual citizens in response to the *Federal Register* notice and public meeting. A summary of these comments can be viewed on EPA's Web site at www.epa.gov/safewater/uic/cbmstudy.

EPA revised its study approach in response to the comments it received on the conceptual study design. The final study design, "Study Design for Evaluating the Impacts to Underground Sources of Drinking Water by Hydraulic Fracturing of Coalbed Methane Reservoirs", was released in April 2001 and is available on the website referenced above. One significant change in the final study design was EPA's decision to complete the study in a phased approach to efficiently address the stated project objectives. This phased approach, similar in design to that used in other complex studies, would allow EPA to use information gained in one phase to focus on the need for, and direction of, subsequent phases.

Phase I of the study was intended as a limited-scope assessment that would enable the Agency to determine if hydraulic fracturing of coalbed methane wells clearly poses little or no threat to USDWs, or if the practice may pose a threat. In Phase I, EPA:

- Gathered existing information to review hydraulic fracturing processes, practices and settings;
- Requested public comment to identify incidents that had not been reported to EPA; and
- Reviewed reported incidents of groundwater contamination and any follow-up

actions or investigations by other parties (state or local agencies, industry, academia, etc.).

In addition, as recommended by commenters, EPA decided to compile accounts of personal experiences with coalbed methane impacts on drinking water wells. These experiences are summarized in Chapter 6.

In its final study design, EPA indicated that the Agency would make a determination regarding whether further investigation was needed after analyzing the Phase I information. Specifically, EPA determined that it would not continue into Phase II of the study if the investigation found that no hazardous constituents were used in fracturing fluids, hydraulic fracturing did not increase the hydraulic connection between previously isolated formations, *and* reported incidents of water quality degradation could be attributed to other, more plausible causes.

EPA identified two possible mechanisms by which hydraulic fracturing fluid injection into coalbed methane wells could potentially affect the quality of USDWs:

1. Direct injection of fracturing fluids into a USDW in which the coal is located, or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. Creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

To determine if contamination might occur through either mechanism, EPA collected information on:

1. Hydraulic fracturing practices.

2. Hydraulic fracturing fluids and additives to determine whether these substances contain hazardous constituents.

3. The hydrogeology of the coalbed methane basins, including the identification of coal seams that are located in USDWs.

4. Water quality incidents potentially associated with hydraulic fracturing.

EPA anticipated that sufficient information would be available to evaluate the impacts of direct injection into USDWs because the main considerations are the location of the coal formations relative to USDWs and the chemical constituents in hydraulic fracturing fluids. The Agency further anticipated that documenting USDW impacts via the creation of a hydraulic connection between the coalbed formation and adjacent USDW(s) would be more difficult. This is because more detailed, site-specific, geological information or data for specific fracturing events needed to definitively document such a hydraulic communication are not readily available. Site-specific data include:

- Water quantity and quality conditions in a USDW (or a well) both before and after a fracturing event;
- Location, dimensions, and conductivity of fractures created during the coalbed stimulation event;
- Measured changes in groundwater flow between the USDW and coalbeds or other aquifers; and
- Impacts of other, unrelated, hydrologic and water quality processes that could also be affecting the USDW.

2.2 Information Sources

EPA obtained available literature and information through:

- Literature reviews.
- Coordination with DOE.
- Interviews with companies that perform hydraulic fracturing and interviews with citizens, local and state authorities, the Bureau of Land Management and EPA Region 8 personnel.
- Field visits.
- Responses to EPA's *Federal Register* request (66 FR 39396 (U.S. EPA, 2001)) for information on incidents of groundwater contamination believed to be associated with hydraulic fracturing of coalbed methane wells.

EPA researched more than 200 peer-reviewed publications, interviewed approximately 50 employees from industry and state or local government agencies, and communicated with approximately 40 citizens and groups who are concerned that CBM production affected their drinking water wells.

The procedure that EPA used to obtain information from each of these sources is discussed in more detail below. A copy of the quality assurance protocol that EPA employed to verify all the sources of data used to write this report is provided as Appendix B.

2.2.1 Literature Reviews

EPA conducted a review of existing literature and information on hydraulic fracturing for coalbed methane production. The focus of the literature review was to obtain information on topics 1 through 3 listed in Section 2.1, above.

The degree to which information was available for each of the 11 coalbed basins in the report was variable. The amount of information available depended on the extent of exploration and production in each basin.

EPA conducted an extensive literature search, using the Engineering Index and GeoRef on-line reference databases, for abstracts from technical articles, books, and proceedings. EPA also conducted Internet-based searches to locate additional information using relevant Web sites located using various search engines, including Google™, Yahoo®, and Alta Vista®. EPA used specialized search engines, such as those provided on state geological survey Web sites and by the Gas Technology Institute (GTI) for specific queries. All relevant Web sites were logged in project books and referenced in this report when cited.

EPA conducted these literature searches by subject topics and using the following key words, either separately or in combination: coal basin, coalbed methane, cross-linked gel, fracturing fluid additives, fracturing fluid technology, fracturing fluid performance, fracturing fluids, ground water, hydraulic fracturing, hydraulic fracture dimension, hydraulic fracture growth, hydrology, linear gel, methane gas production, nitrogen foam, underground sources of drinking water, and USDWs. EPA printed, catalogued, and surveyed all results of searches for pertinent journal articles, books, and conference proceedings containing information that might meet the specific data needs of this report.

EPA acquired most of the pertinent articles, which were identified from the Engineering Index and GeoRef on-line reference databases, from the University of Texas Library in Austin because this library's holdings include an extensive collection of publications related to oil and gas production. EPA researched references from the University of Texas documents and acquired those documents that were relevant to the study. Only a small fraction of the pertinent articles, specifically proprietary articles and articles published for overseas conferences were unavailable. EPA also acquired articles from GTI. EPA has archived, by topic, all papers collected for the study.

To verify key information extracted from the literature, EPA contacted by phone relevant organizations such as state regulatory agencies, state geological surveys, natural gas companies, GTI, and service companies. The Agency used telephone logs to document all communications. Personal conversations with the employees of the various organizations yielded additional information in the form of reports, figures, and maps, as well as statements based on best professional judgment and experience. These were collected, documented, and referenced in conjunction with the literature identified in the literature searches. The majority of the literature pertaining to coalbed methane basins and hydraulic fracturing was written in the early to mid 1990s. According to the Texas Bureau of Economic Geology (TBEG) (personal communication, TBEG Staff, 2000), this period of intense activity was a result of the emphasis placed on gas exploration by the Section 29 Tax Credit of the Crude Oil Windfall Profit Tax Act of 1980 and research grants to industry, academia, and government agencies. The Section 29 credit does not, however, apply to coal gas wells drilled after December 31, 1992.

2.2.2 Department of Energy

EPA reviewed information from DOE's "White Paper" on hydraulic fracturing practices. This paper addresses the following topics:

- Objectives of hydraulic fracturing.
- How candidate wells are selected for hydraulic fracturing.
- How fracture treatments are designed.
- Field operation considerations.
- Physics of fracture formation in coalbeds.
- Fracturing fluids.
- Stimulation techniques used for developing coalbeds.
- Instrumentation/methods for tracking fractures.

The complete DOE paper is included as Appendix A, and excerpts from this paper are included in Chapter 3, Characteristics of Coalbed Methane Production and Associated Hydraulic Fracturing Practices.

2.2.3 Interviews

EPA contacted hydraulic fracturing service companies including BJ Services Company, Halliburton Energy Services, Inc., and Schlumberger Technology Corporation, as well as a fracturing fluids producer, Hercules, Inc., to obtain information regarding the content of hydraulic fracturing fluids and additives they use or manufacture. Two companies, Halliburton and Schlumberger, provided EPA with material safety data sheets (MSDSs) for several hydraulic fracturing fluids and additives. The MSDSs were reviewed to determine the nature of the constituents in fracturing fluids used to stimulate coalbed methane production. These topics are discussed in Chapter 4, Hydraulic Fracturing Fluids.

EPA also evaluated reports from individuals and organizations that are concerned that their drinking water supplies were affected by hydraulic fracturing. These reported personal experiences came from Colorado, New Mexico, Wyoming, Alabama, and Virginia. In response to these reports, EPA conducted telephone interviews with citizens, local and state authorities, the Bureau of Land Management and EPA Region 8 personnel. EPA also evaluated state agency responses to any complaints received by EPA or state agencies. The Agency also evaluated the available data to determine whether there is sufficient information to reveal the source of the alleged water quality contamination.

2.2.4 Field Visits

EPA conducted field visits in Colorado, Kansas, and Virginia to better understand how local coalbed methane production activities may vary from basin to basin. In addition, during the field visits, EPA was able to meet with concerned local citizens and state agencies to discuss coalbed methane production issues. A summary of these field visits is outlined below.

In August 2000, EPA met with a group of concerned citizens, officials from the Colorado Oil and Gas Conservation Commission, and representatives of the La Plata County government. EPA witnessed a fracturing event, reviewed records including temperature logs of past fracturing events conducted on coalbed methane wells, and performed a reconnaissance of the area allegedly affected by coalbed methane production.

In August 2001, EPA met with the Virginia Department of Mines, Minerals and Energy, the agency that regulates the coalbed methane production industry in Virginia. The Department provided information about the state's investigation of water quality incidents potentially associated with coalbed methane production in the Central Appalachian Valley. The Department also submitted water quality incident reports for review by EPA. During this visit, EPA met with concerned citizens in Virginia. Citizens groups from Buchannan and surrounding counties were invited to meet with EPA and DOE staff to discuss water quality issues believed to be related to local hydraulic fracturing of coalbed methane wells. Notes from the meeting are referenced in Chapter 6.

EPA also organized a field visit with Consol Energy, Inc. and Halliburton to witness a hydraulic fracturing event. Halliburton performed a hydraulic fracture job on a coalbed methane well in western Virginia using equipment, fracturing fluids, and techniques, which are typical of those described in the literature. EPA was able to observe the fracturing process and collect information, including MSDSs from the service company and gas company engineers. The information from this field visit was used to supplement the data on hydraulic fracturing fluids in Chapter 4.

In November 2001, EPA witnessed a fracturing event in Wilson County, Kansas, to gain a better understanding of the regional geology and hydraulic fracturing practices in the area. In attendance were Colt Energy (the well operator); Consolidated Industrial Services, Inc. (the service company conducting the fracture job); and two state agencies, the Kansas Corporation Commission, and the Missouri Department of Natural Resources. MSDSs for fracturing fluids typically used in the area were also provided to EPA by the Kansas Corporation Commission.

2.2.5 Federal Register Notice to Identify Reported Incidents

EPA provided an opportunity for the public to submit information on any impacts to groundwater believed to be associated with hydraulic fracturing through a request for public comment (66 FR 39396 (USEPA, 2001)). EPA also sent copies of the *Federal Register* notice with a cover letter to county-level public health and/or environmental officials in counties that may be producing coalbed methane. In addition, letters were sent to stakeholders informing them that the *Federal Register* notice had been published. Responses to the *Federal Register* notice are available at EPA's water docket (docket number W-01-09; Water Docket (MC 4101); Rm EB 57; U.S. Environmental Protection Agency; 1200 Pennsylvania Avenue, NW; Washington, DC 20460; phone number: (202) 566-2426). A summary of the comments is provided in Chapter 6.

2.3 Review Process

This report has benefited from a series of internal and external technical reviews. EPA verified information through telephone interviews with state and local officials, as well as through the Agency's internal quality assurance process. EPA conducted a quality assurance review of the data collection procedures as well as a review of the individual literature sources cited in the report. In addition, more than nine EPA offices reviewed and commented on the draft report. Other federal agencies that reviewed the draft report included DOE and the U.S. Geological Survey (USGS).

In 2001, EPA also submitted the draft report to a scientific peer-review panel consisting of experts from industry, academia, and government agencies. The panel's task was to review the draft report and provide comments on the descriptions and conclusions developed by EPA. The panel also provided information about additional data sources to supplement those used in the report. Following receipt of comments on the draft report, EPA made the appropriate changes to the document prior to its publication and release.

EPA made the report available for public comment by an announcement in the *Federal Register* on August 28, 2002 (67 FR 55249 (USEPA, 2002)). The 60-day public comment period officially ended on October 28, 2002. The Agency received and reviewed comments from 105 commenters and incorporated many of these comments into this final Phase I report. A summary of the public comments and EPA's responses is provided in, "Public Comment and Response Summary for the Study on the Potential Impacts of Hydraulic Fracturing of Coalbed Methane Wells on Underground Sources of Drinking Water" (EPA 816-R-04-004), available on EPA's electronic docket.

Chapter 3
Characteristics of Coalbed Methane Production and Associated Hydraulic Fracturing Practices

Understanding the practice of hydraulic fracturing as it pertains to coalbed methane production is an important first step in evaluating its potential impacts on the quality of USDWs. This chapter presents the following: an overview of the geologic processes leading to coal formation, an overview of coalbed methane production practices, a discussion of fracture behavior, a review of the literature on the use and recovery of fracturing fluids, a discussion of mechanisms affecting fluid recovery, and a summary of the methods used for measuring and predicting fracture dimensions and fracturing fluid movement. In addition, several diagrams have been included at the end of this chapter to help illustrate many of these topics. Specifically, Figures 3-1 through 3-8 show the location of the coal basins, the geography of a peat-forming system, the geometry of natural cleats and hydraulically induced fractures, an overview of the hydraulic fracturing process, the relationship between water and gas production rates, and side and plan views of vertical hydraulic fractures.

3.1 Introduction

Coalbed methane is a gas formed as part of the geological process of coal generation, and is contained in varying quantities within all coal. Coalbed methane is exceptionally pure compared to conventional natural gas, containing only very small proportions of "wet" compounds (e.g., heavier hydrocarbons such as ethane and butane) and other gases (e.g., hydrogen sulfide and carbon dioxide). Coalbed gas is over 90 percent methane, and is suitable for introduction into a commercial pipeline with little or no treatment (Rice, 1993; Levine, 1993).

From the earliest days of coal mining, the flammable and explosive gas in coalbeds has been one of mining's paramount safety problems. Over the centuries, miners have developed several methods to extract the coalbed methane from coal and mine workings. Coalbed methane well production began in 1971 and was originally intended as a safety measure in underground coalmines to reduce the explosion hazard posed by methane (Elder and Deul, 1974).

In 1980, the United States Congress enacted a tax credit for "Non-conventional energy production." In 1984, there were only several hundred coalbed methane wells in the United States and most were used for mine de-methanization. By 1990, the anticipated expiration of the tax credit contributed to a dramatic increase in the number of coalbed methane wells nationwide. In addition, DOE and GTI supported extensive research into coalbed exploration and production methods. Federal tax credits and State Severance Tax exemptions served to subsidize the development of coalbed methane resources (Soot, 1991; Pashin and Hinkle, 1997). The federal tax credits and incentives expired at the end of 1992, but coalbed methane exploration, development, and reserves have remained stable or increased (Stevens et al., 1996). At the end of 2000, coalbed methane production from 13 states totaled 1.353 trillion cubic feet, an increase of 156 percent from 1992. During 2000, a total of 13,973 coalbed methane wells were in production (GTI; EPA Regional Offices, 2001). By the end of 2000, coalbed methane production accounted for about 7 percent of the total United States dry gas production and 9 percent of proven dry gas reserves (EIA, 2001).

Coal is defined as a rock that contains at least 50 percent organic matter by weight. The precursor of coal is peat, plant matter deposited over time in fresh-water swamps associated with coastal deltaic rivers. The coal resources from which coalbed methane is derived have similar geologic origins. In the United States, they are usually found in geologic formations that are approximately 65-325 million years old. Coal formation occurred during a time of moderate climate and broad inland oceans. Sea level rose and fell in conjunction with tectonic forces (i.e., subsidence and uplift of land masses) and melting/freezing cycles of decreases and increases in the polar ice masses. As a result, coastal environments such as coastal deltas and peat swamps migrated landward when sea levels rose and moved seaward when sea levels fell, marked by cycles of submergence and emergence. With these cycles of rising and falling sea levels, what was a peat swamp at one time would later be under 100 feet of water. The cycle of sea level rising and falling is marked in the geologic record as cycles of inter-layered deep and shallow water sediments.

The type of sediments deposited at a given location varied with the depth of submergence (Figure 3-2). Generally, carbon-rich organic plant matter was deposited in shallow peat swamps, sand was deposited along beaches and other near-shore, shallow marine environments, and silts and clays and calcium-rich muds were deposited further off-shore in deeper marine environments. Subjected to high pressure over considerable time (due to burial under subsequent sediments), the peats were transformed into coal, the sand into sandstone, the silts and clays into shales, siltstones, and mudstones, and the calcium-rich muds were transformed into limestones. These coal-bearing inter-layered sedimentary sequences are sometimes referred to as “coal cycles.” The idealized coal cycle consists of repeated sequences of very fine-grained sediments (shales and limestones) overlain by coarser sediments (siltstones and sandstones), and then capped by coal. The sequence repeats with shales and limestones over the coal, followed by siltstones and sandstones, then more coal, and so on. Sometimes certain formations are missing from the sequences, so coal is often, though not always, overlain by shales and limestones.

The sedimentation patterns in these fluctuating coastal environments over geologic time scale determined the presence, thickness, and geometry of present-day coalbeds. The number of coal cycles determines the number of resulting coalbeds. For example, the Black Warrior Basin of Alabama includes up to 10 cycles, whereas the San Juan Basin (New Mexico and Colorado) contains as few as 3. The short, rising and falling sea level cycles reflected in the Black Warrior Basin geology produced many thin coalbeds, ranging from less than 1 inch to as much as 4 feet thick (Carrol et al., 1993; Pashin, 1994a and 1994b), whereas the stable, long-term cycles of the San Juan Basin produced fewer, but thicker coalbeds, with single coalbeds up to 70 feet thick (Kaiser and Ayers, 1994).

Peat is transformed into coal when it is buried by accumulating sediment and heated in the subsurface over geologic time. The "rank" of coal describes the amount of energy (measured in British thermal units or Btus) it contains, and is a function of the proportion and type of organic matter, the length and temperature of burial, and the influences of subsequent hydrogeologic and tectonic processes (Carrol et al., 1993; Levine, 1993; Rice, 1993). Methane is generated as part of the process whereby peat is transformed into coal. The origin of methane in coal of low rank, such as bituminous coal, is primarily biogenic (i.e., the result of bacterial action on organic matter) (Levine 1993, as cited by the Alabama Oil and Gas Board, 2002). Low rank coals tend to have lower gas content than high rank coals such as anthracite. Anthracite can have extremely high gas content, but the gas tends to desorb so slowly that anthracite is an insignificant source of coalbed methane (Levine, 1993, as cited by the Alabama Oil and Gas Board, 2002). Commercial coalbed methane production takes place in coals of mid-rank, usually low- to high-volatile bituminous coals (Levine, 1993; Rice, 1993).

A network of fractures, joints, and a sub-network of small joints called cleats commonly characterize the physical structure of coalbeds. Joints are larger, systematic, near-vertical fractures within the coal, generally spaced from several feet to several dozen feet apart (Close, 1993; Levine, 1993). There are two types of cleats: the primary, more continuous cleats are called *face cleats*, while the abutting cleats are called *butt cleats* (Laubach and Tremain 1991; Close, 1993; Levine, 1993) (Figure 3-3). The butt cleats appear as the rungs on a ladder that are bounded on each side by the face cleats. The spacing between cleats is often roughly proportional to the thickness of coal cut by the cleats; thin coals have more closely spaced cleats and thick coals more widely spaced cleats (Laubach et al., 1998, as cited by Olson, 2001).

The primary (or natural) permeability of coal is very low, typically ranging from 0.1 to 30 millidarcies (md) (McKee et al., 1989). According to Warpinski (2001), because coal is a very weak (low modulus) material and cannot take much stress without fracturing, coal is almost always highly fractured and cleated. The resulting network of fractures commonly gives coalbeds a high secondary permeability (despite coal's typically low primary permeability). Groundwater, hydraulic fracturing fluids, and methane gas can more easily flow through the network of fractures. Because hydraulic fracturing generally enlarges pre-existing fractures and rarely creates new fractures (Steidl, 1993; Diamond, 1987a and b; Diamond and Oyler, 1987), this network of natural fractures is very important to the extraction of methane from the coal.

3.2 Hydraulic Fracturing

This section provides an overview of the hydraulic fracturing process, and the factors that affect fracture behavior and fracture orientation. Figure 3-4 provides a simplified graphical representation of a hypothetical hydraulic fracturing event in a coalbed methane well. This diagram shows the fracture initiation and propagation stages, as well as the proppant placement and fracturing fluid recovery stages. Only horizontal fractures are shown in this diagram, although hydraulically induced fractures are often vertically oriented.

3.2.1 The Hydraulic Fracturing Process

Hydraulic fracturing is a technique used by the oil and gas industry to improve the production efficiency of oil and coalbed methane wells. The extraction of coalbed methane is enhanced by hydraulically enlarging and/or creating fractures in the coal zones. The resulting fracture system facilitates pumping of groundwater from the coal zone, thereby reducing pressure and enabling the methane to be released from the coal and more easily pumped through the fracture system back to the well (and then through the well to the surface). To initiate the process, a production well is drilled into the targeted coalbeds. Fracturing fluids containing proppants are then injected under high pressure into the well and specifically into the targeted coalbeds in the subsurface.

The fracturing fluids are injected into the subsurface at a rate and pressure that are too high for the targeted coal zone to accept. As the resistance to the injected fluids increases, the pressure in the injecting well increases to a level that exceeds the breakdown pressure of the rocks in the targeted coal zone, and the rocks "breakdown" (Olson, 2001). In this way, the hydraulic fracturing process "fractures" the coalbeds (and sometimes other geologic strata within or around the targeted coal zones). This process sometimes can create new fractures, but most often opportunistically enlarges existing fractures, increasing the connections of the natural fracture networks in and around the coalbeds (Steidl 1993; Diamond 1987a and b; Diamond and Oyler, 1987). The pressure-induced fracturing serves to connect the network of fractures in the coalbeds to the hydraulic fracturing well (which subsequently will serve as the methane extraction or production well). The fracturing fluids pumped into the subsurface under high pressure also deliver and emplace the "proppant." The most common proppant is fine sand; under pressure, the sand is forced into the natural and/or enlarged fractures and acts to "prop" open the fractures even after the fracturing pressure is reduced. The increased permeability due to fracturing and proppant emplacement facilitates the flow and extraction of methane from coalbeds.

Methane within coalbeds is not structurally "trapped" by overlying geologic strata, as in the geologic environments typical of conventional gas deposits. Only about 5 to 9 percent of the coalbed methane is present as "free" gas within the joints and cleats of coalbeds. Most of the coalbed methane is contained within the coal itself (adsorbed to the sides of the small pores in the coal) (Koenig, 1989; Winston, 1990; Close, 1993).

Before coalbed methane production begins, groundwater and injected fracturing fluids are first pumped out (or "produced" in industry terminology) from the network of fractures in and around the coal zone. The fluids are pumped until the pressure declines to the point that methane begins to desorb from the coal (Gray, 1987).

Coalbed methane production initially requires pumping and removing significant amounts of water to sufficiently reduce the hydrostatic pressure in the subsurface so that methane can desorb from the coal before methane extraction can begin. Coalbed methane is produced at close to atmospheric pressure (Ely et al., 1990; Schraufnagel, 1993). The proportion of water to methane pumped is initially high and declines with increasing coalbed methane production (Figure 3-5). In contrast, in the production of conventional petroleum-based gas, the production of gas is initially high, and as gas production continues over time and the gas resources are progressively depleted, gas production decreases and the amount of water pumped increases.

Almost every coalbed targeted for methane production must be hydraulically fractured to connect the production well bore to the coalbed fracture network (Holditch et al., 1988). Although the general hydraulic fracturing process (described above) is generally similar across the country, the details of the process can differ significantly from location to location depending on the site-specific geologic conditions. For example, although most hydraulic fracturing wells are completely cased except for openings at the targeted coal zone, many wells in the San Juan Basin are fractured by creating a cavity in the open-hole section. Also, in contrast to the typical fracturing job, many wells in the Black Warrior Basin are stimulated more than once. Here, when wells are open to multiple coal seams, the hydraulic fracturing process may involve several or multiple fracturing events, using from 2 to 5 hydraulic fracture treatments per well (depending on number of seams and spacing between seams). Many coalbed methane wells are re-fractured at some time after the initial treatment in an effort to re-connect the wellbore to the production zones to overcome plugging or other well problems (Holditch, 1990; Saulsberry et al., 1990; Palmer et al., 1991a and 1991b; Holditch, 1993). Also, in response to site-specific coal geology and the economics of coalbed methane production where coal seams are thin and vertically separated by up to hundreds of feet of intervening rock) operators might design fracture treatments to enhance the vertical dimension and perform several fracture treatments within a single well to produce methane in an economically viable fashion, (Ely, et al., 1990; Holditch, 1990; Saulsberry et al., 1990; Spafford, 1991; Holditch, 1993).

3.2.2 Factors Affecting Fracture Behavior

Fracture behavior is of interest because it contributes to an understanding of the potential impact of fracturing fluid injection on USDWs; the opportunities for fracture connections within or into a USDW are affected by the extent to which a hydraulically induced fracture grows. Specifically, when hydraulic fracturing fluids are injected into formations that are not themselves USDWs, the following scenarios are of potential concern:

- The hydraulically induced fracture may extend from the target formation into a USDW.

- The hydraulically induced fracture may connect with natural (existing) fracture systems and/or porous and permeable formations, which may facilitate the movement of fracturing fluids into a USDW.

Fracture behavior through coal and other geologic formations commonly present above and below coalbeds depends on site-specific factors such as the following:

1. Physical properties, types, thicknesses, and depths of the targeted coalbeds as well as those of the surrounding geologic formations.

2. Presence of existing natural fracture systems and their orientation in the coalbeds and surrounding formations.

3. Amount and distribution of stress (i.e., in-situ stress), and the stress contrasts between the targeted coalbeds and surrounding formations.

4. Hydraulic fracture stimulation design including volume of fracturing fluid injected into the subsurface as well as the fluid injection rate and fluid viscosity.

Many of these factors are interrelated and together will influence whether and how far hydraulic fractures will propagate into or beyond coalbeds targeted for fracturing. These factors are discussed below.

Properties of Coalbeds and Surrounding Formations

Coalbed depth and rock types in the coal zone have important fundamental influences on fracture dimensions and orientations. According to Nielsen and Hansen (1987, as cited in Appendix A: DOE, Hydraulic Fracturing), generally, at depths of less than 1,000 feet, the direction of least principal stress tends to be vertical and, therefore, at these relatively shallow depths fractures typically have more of a horizontal than a vertical component. Here, horizontal fractures tend to be created because the hydraulically induced pressure forces the walls of the fracture to open in the direction of least stress (which is vertical), creating a horizontal fracture. At these shallower depths, the horizontal fractures result from the low vertical stress due to the relatively low weight of overlying geologic material (due to the shallow depth). Shallow vertical fractures are most likely due to the presence of natural (existing) vertical fractures, from which hydraulically induced vertical fractures can initiate. Generally, in locations deeper than 1,000 feet, the least principal stress tends to be horizontal so vertical fractures tend to form. Vertical fractures created in these greater depths can propagate vertically to shallower depths and develop a horizontal component (Nielsen and Hansen, 1987 as cited in Appendix A: DOE, Hydraulic Fracturing). In the formation of these "T-fractures," the fracture tip may fill with coal fines or intercept a zone of stress contrast, causing the fracture to turn and develop horizontally, sometimes at the contact of the coalbed and an overlying formation.

In many coalbed methane basins, the depths, lithologic properties, and stress fields of the coal zones result in hydraulic fractures that are higher than they are long ("length" refers to horizontal distance from the well bore) (Diamond, 1987a; Morales et al., 1990; Zuber et al., 1990; Holditch et al., 1989; Palmer and Sparks, 1990; Jones and Schraufnagel, 1991; Steidl, 1991; Wright, 1992; Palmer et al., 1991a and 1993a). Almost all of the sites studied by Diamond (1987a and b) had vertical fractures, and about half had horizontal fractures.

Naceur and Touboul (1990) state that the primary mechanisms controlling fracture height are contrasts in the physical properties of the rock strata within and surrounding the coal zone being fractured. Contrasts in strata stresses, moduli, leakoff, and toughness affect fracture growth, with stress contrasts being the most important mechanism controlling fracture height (Naceur and Touboul 1990). (Stress is discussed in more detail later in this section.) Moduli are the ratios of stress to strain in various formations. Leakoff is the magnitude of pressure exerted on a formation that causes fluid to be forced into the formation. The fluid may be flowing into the pore spaces of the rock or into cracks opened and propagated into the formation by the fluid pressure. Toughness can be defined as the point at which enough stress intensity has been applied to a rock formation, so that a fracture initiates and propagates. Coal is generally very weak (with low modulus) and easily fractures. Siltstones, sandstones, and mudstones (other rock types commonly occurring in coal zones) tend to have higher moduli, greater toughness and fracture less easily (Warpinski, 2001). Thick shales, which commonly overlie coalbeds, often act as a barrier to fracture growth (see Appendix A).

Another factor controlling fracture height can be the highly cleated nature of some coalbeds. In some cases, highly cleated coal seams will prevent fractures from growing vertically. When the fracturing fluid enters the coal seam, it is contained within the coal seam's dense system of cleats and the growth of the hydraulic fracture will be limited to the coal seam (see Appendix A).

The low permeability of relatively unfractured shale may help to protect USDWs from being affected by hydraulic fracturing fluids in some basins. If sufficiently thick and relatively unfractured shales are present, they may act as a barrier not only to fracture height growth, but also to fluid movement. The degree to which any formation overlying targeted coalbeds will act as a hydraulic barrier will depend on site-specific factors.

The lithology of coalbeds and surrounding formations is variable in the basins where coalbed methane is produced. Although common, the idealized coal cycle (with shales overlying coalbeds) is not always present in all coal basins or necessarily in all parts of any basin. Although Holditch (1993) states that fracture heights can grow where the coal seam is bounded above or below by sandstone, Warpinski (2001) states that highly layered formations or very permeable strata, such as some sandstones, can act to inhibit fracture growth. Some of the coal seams of the San Juan Basin are bounded below by sandstone. In some locations in each of the other basins, coalbeds are underlain by, overlain by, or interbedded with sandstones. Additional detail on the stratigraphy within each basin is provided in the attachments to this study.

Differences in fracture behavior may also be due in part to very small (but influential) layers or irregularities that exist in the rocks as part of the sedimentation process that created them. Therefore, a valid measurement of rock properties relevant to fracture behavior at one location may not adequately represent the properties of similar rock at another location (Hanson et al., 1987; Jones et al., 1987a and 1987b; Palmer et al., 1989; Morales et al., 1990; Naceur and Touboul, 1990; Jones and Schraufnagel, 1991; Palmer et al., 1993b; Elbel, 1994). For example, the presence of a shallow clay layer as thin as 10 millimeters at the upper contact of a coal seam can cause a vertically propagating, shallow hydraulic fracture to "turn" horizontal and fail to penetrate the next overlying coal seam (Jones et al., 1987a; Palmer et al., 1989; Morales et al., 1990; Palmer et al., 1991b and 1993b). In other cases, hydraulic fractures may penetrate into or even, as shown in the case of some thin shales, completely through overlying shale layers (Diamond, 1987a and b; Diamond and Oyler, 1987). Warpinski et al. (1982) found that even microscopically-thin ash beds can influence hydraulic fracture propagation. In other words, the site-specific geology can play a key role in influencing fracture behavior. In addition to the effects of the rock type and sometimes even thin layers within strata, natural fractures also play a role in fracture behavior and fracture propagation.

Natural Fracture Systems

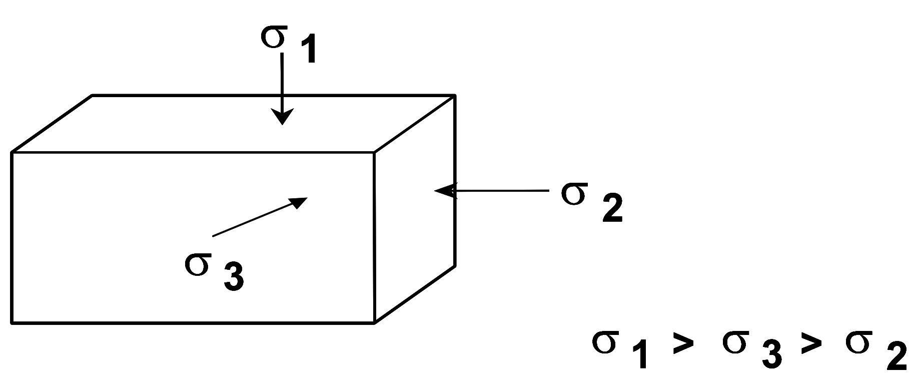
Steidl (1993), based on his "mined-through" studies, concluded that high coalbed methane production depends greatly on the presence of pre-existing natural fracture systems. Hydraulic fracturing tends to widen naturally occurring planes of weakness and rarely creates new fractures, as based on observations by Diamond (1987a and b) and Diamond and Oyler (1987) in their mined-through studies. ("Mined-through" studies provide unique subsurface access to investigate coalbeds and surrounding rock after hydraulic fracturing. Mined-through studies are reviewed in more detail in section 3.4.1.) Diamond and Oyler (1987) also noted that this opportunistic enlarging of preexisting fractures appears to account for those cases where hydraulic fractures propagate from the targeted coalbeds into overlying rock, and their studies found penetration into overlying layers in nearly half of the fractures intercepted by underground mines.

Importantly, in several locations in the Diamond (1987a and b) study sites, fluorescent paint was injected along with the hydraulic fracturing fluids and the paint was found in natural fractures from 200 to slightly more than 600 feet beyond the sand-filled ("propped") portions of hydraulically induced or enlarged fractures. This suggests that the induced/enlarged fractures link into the existing fracture network system and that hydraulic fracturing fluids can move beyond, and sometimes significantly beyond, the propped, sand-filled portions of hydraulically induced fractures (Steidl 1993; Diamond 1987a and b; Diamond and Oyler, 1987). The mined-through studies did not conduct systematic assessments of the extent of the fractures into or through the roof rock shales that were immediately above the mined coal (the rock strata immediately above a mined coal layer is referred to as the "roof rock").

In-Situ Stress and Stress Contrasts

In-situ stress and the relative stress of neighboring geologic strata are important influences on fracture behavior. A discussion of in-situ stress is provided in DOE's paper "Hydraulic Fracturing" (provided as Appendix A). In-situ stress is described as:

> "Underground formations are confined and under stress… [The graphic below] illustrates the local stress state at depth for an element of formation. The stresses can be divided into 3 principal stresses… [In the graphic below,] σ_1 is the vertical stress, σ_2 is the maximum horizontal stress, while σ_3 is the minimum horizontal stress, where $\sigma_1>\sigma_2>\sigma_3$. This is a typical configuration for coalbed methane reservoirs. However, depending on geologic conditions, the vertical stress could also be the intermediate (σ_2) or minimum stress (σ_3). These stresses are normally compressive and vary in magnitude throughout the reservoir, particularly in the vertical direction (from layer to layer). The magnitude and direction of the principal stresses are important because they control the pressure required to create and propagate a fracture, the shape and vertical extent of the fracture, the direction of the fracture, and the stresses trying to crush and/or embed the propping agent during production."



Local in-situ stress at depth.

According to (Naceur and Touboul 1990), the contrast in stress between adjacent rock strata within and surrounding the targeted coal zone is the most important mechanism controlling fracture height. Stress contrast is important in determining whether a fracture will continue to propagate in the same direction when it hits a geologic contact between two different rock types. Often, a high stress contrast results in a barrier to fracture propagation. An example of this would be where there is a geologic contact between a coalbed and an overlying, thick, higher-stress shale.

Hydraulic Fracture Stimulation Design

The procedures and fracturing fluids used to stimulate coalbed methane wells can differ from operator to operator in a single basin due to local characteristics of geology and depth and to perceived advantages of cost, effectiveness, production characteristics, or other factors. On a larger scale, although fracture stimulations in coalbed methane projects in different basins may share common rock types and characteristics, fracture behavior can differ significantly. Discussions on hydraulic fracturing practices in 11 individual coal basins are included in Chapter 5 and in Attachments 1 through 11.

Aspects of fracture behavior, such as fracture dimensions (height, length, and width), are affected by the different fracturing approaches taken by the operator during a hydraulic fracturing event. Generally, the larger the volume of fracturing fluids injected, the larger the potential fracture dimensions. Fluid injection rates and viscosity can also affect fracture dimensions (Olson, 2001; Diamond and Oyler, 1987). Large injection volumes also often result in extensive networks of induced fractures. Gelled water treatments may result in the widest and longest fractures, but this occurrence cannot be concluded with certainty from the mined-through studies (Diamond and Oyler, 1987; Diamond 1987a and b).

The effects of these operator-controlled actions interact with and are influenced by the physical properties, depths, and in-situ stress of the geologic formations being fractured (as listed above). For example, if a hydraulically induced fracture has a relatively constant height due to a geologic layer acting as a barrier to fracture propagation, and the fracture is forced to grow and increase in volume (through an increased volume of fracturing fluid), the fracture will mainly grow in length. Also, increasing fluid viscosity can increase the pressure due to injection, resulting in greater fracture width, and thus often shorter fractures (Olson, 2001).

3.3 Fracturing Fluids

The fluids used for fracture development are pumped at high pressure into the well. They may be “clear” (most commonly water, but may include acid, oil, or water with friction-reducer additives) or “gelled” (viscosity-modified water, using guar or other gelling agents). Some literature indicates that coalbed fracture treatments use from 50,000 to 350,000 gallons of various stimulation and fracturing fluids, and from 75,000 to 320,000 pounds of sand as proppant (Holditch et al., 1988 and 1989; Jeu et al., 1988; Hinkel et al., 1991; Holditch, 1993; Palmer et al., 1991b, 1993a, and 1993b). More typical injection volumes, based on average injection volume data provided by Halliburton for six coalbed methane locations, indicate a maximum average injection volume of 150,000 gal/well and a median average injection volume of 57,500 gal/well (Halliburton, Inc., 2003).

Depending on the basin and treatment design, the composition of these fluids varies significantly, from simple water and sand to complex polymeric substances with a multitude of additives. Types of fracturing fluids are discussed in greater detail in Chapter 4.

3.3.1 Quantifying Fluid Recovery

Several studies have evaluated the recovery rates of hydraulic fracturing treatment fluids in coal and non-coal formations as discussed in more detail below. Non-coal formations were evaluated to augment the available flowback data.

Coal Formation

Palmer et al. (1991a) measured flowback rates in 13 hydraulic fracturing wells to compare the gas production resulting from the use of water versus gel-based fracturing fluids. This study was conducted in a coal seam with permeabilities from 5 to 20 md. Ten samples collected over a 19-day flowback period indicated a recovery rate of 61 percent. Palmer et al. (1991a) predicted total recovery to be from 68 percent to as much as 82 percent.

Non-Coal Formations

Willberg et al. (1997) conducted a flowback analysis in 10 wells in a heterogeneous sandstone and shale environment that was highly impermeable (i.e., with a permeability of 0.01 md). The fluids used in this study were recovered at an average efficiency of 35 percent during the 4 to 5 day flowback period. Three wells were then sampled every 4 to 8 hours during the subsequent gas production phase to assess long-term polymer recovery, which was found to be minimal (3 percent). Sampling of injected fluid and chloride concentrations indicated that as the flowback and gas production periods progressed, decreasing proportions of the extracted water consisted of the injected fluid, while increasing proportions were natural formation water. In other words, natural formation water was able to bypass viscous gel trapped in the formation and flow into the production wells.

The authors further cited laboratory studies indicating that water may flow past the gel in sand such as that used as proppant in these studies (Willberg et al., 1997). Because the gel is more viscous than water, it is easier for water to respond to pumping and flow through the formation towards the production well. As Willberg et al. (1997) writes, "Production of formation water effectively competes and eventually supersedes residual fracturing fluid recovery, thereby limiting the overall cleanup efficiency." Given that the environments in which coalbed methane is produced are also generally saturated with water, and similar sands are used as proppants, it is possible that gel recovery is impeded in much the same way in coalbed methane stimulations.

Willberg et al. (1998) conducted another flowback analysis and described the effect of flowback rate on cleanup efficiency in an initially dry, very low permeability (0.001 md) shale. Some wells in this study were pumped at low flowback rates (less than 3 barrels per minute (bbl/min). Other wells were pumped more aggressively at greater than 3 bbl/min. Thirty-one percent of the injected fluids were recovered when low flowback rates were applied over a 5-day period. Forty-six percent of the fluids were recovered when aggressive flowback rates were applied in other wells over a 2-day period. Additional fluid recovery (10 percent to 13 percent) was achieved during the subsequent gas production phase, resulting in a total recovery rate of 41 percent to 59 percent. Willberg speculated that the lower recovery rate in the 1997 study was due to the pumping of large amounts of formation water during the recovery process, compared to the 1998 study that was conducted in a relatively dry environment.

3.3.2 Mechanisms Affecting Fluid Recovery

A variety of site-specific factors will influence the recovery efficiency of fracturing fluids. These factors are summarized as follows:

- Fluids can "leakoff" (flow away) from the primary hydraulically induced fracture into smaller secondary fractures. The fluids then become trapped in the secondary fractures and/or pores of porous rock.

- Fluids can become entrapped due to the "check-valve effect," wherein fractures narrow again after the injection of fracturing fluid ceases, formation pressure decreases, and extraction of methane and groundwater begins.

- Some fluid constituents can become adsorbed to coal or react chemically with the formation.

- Some volume of the fluids, moving along the hydraulically induced fracture, may move beyond the capture zone of the pumping well, and thus cannot be recovered during fluid recovery. The capture zone of the production well is that portion of the aquifer that contributes water to the well.

- Some fluid constituents may not completely mix with groundwater and therefore would be difficult to recover during production pumping.

Each of these mechanisms is discussed in greater detail in this section.

Fluid Leakoff

Fluids can be "lost" (i.e., remain in the subsurface unrecovered) due to "leakoff" into connected fractures and the pores of porous rocks (Figure 3-7). Fracturing fluids injected into the primary hydraulically induced fracture can intersect and flow (leakoff) into preexisting smaller natural fractures. Some of the fluids lost in this way may occur very close to the well bore after traveling minimal distances in the hydraulically induced fracture before being diverted into other fractures and pores. The volume of fracturing fluids that may be lost in this way depends on the permeability of the rocks and the surface area of the fracture(s).

The high injection pressures of hydraulic fracturing can force the fracturing fluids to be transported deep into secondary fractures. The cleats in coal are presumed to play an important role in leakoff (Olson, 2001). Movement into smaller fractures and cleats can be to a point where flowback efforts will not recover them. The pressure reduction caused by pumping during subsequent production is not sufficient to recapture all the fluid that has leaked off into the formation. The loss of fluids due to leakoff into small fractures and pores is minimized by the use of cross-linked gels, discussed in more detail in Chapter 4.

Check-Valve Effect

A check-valve effect occurs when natural or propagating fractures open and allow fluids to flow through when fracturing pressure is high, but subsequently prevent the fluids from flowing back towards the production well as they close after fracturing pressure decreases (Warpinski et al., 1988; Palmer et al., 1991a). A long fracture can be pinched off at some distance from the well. This reduces the effective fracture length available to transport methane from various locations within the coalbed to the production well. Fluids trapped beyond the "pinch point" are unlikely to be recovered during flowback.

In most cases, when the fracturing pressure is released, the fracture closes in response to natural subsurface compressive stresses. Because the primary purpose of hydraulic fracturing is to increase the effective permeability of the target formation and connect new or widened fractures to the well, a closed fracture is of little use. Therefore, a component of coalbed methane production well development is to "prop" the fracture open, so that the enhanced permeability from the pressure-induced fracturing persists even after fracturing pressure is terminated. To this end, operators use a system of fluids and "proppants" to create and preserve a high-permeability fracture-channel from the well into the formation.

The check-valve effect takes place in locations beyond the zone where proppants have been emplaced (or in smaller secondary fractures that have not received any proppant). Because of the heterogeneous, stratified, and fractured nature of coal deposits, it is likely that some volume of stimulation fluid cannot be recovered due to its movement into zones that were not completely "propped."

Adsorption and Chemical Reactions

Adsorption and chemical reactions can prevent the fluid from being recovered. Adsorption is the process by which fluid constituents adhere to a solid surface (i.e., the coal, in this case) and are thereby unavailable to flow with groundwater. Adsorption to coal is likely; however, adsorption to other surrounding geologic material (e.g., shale, sandstone) is likely to be minimal. Another possible reaction affecting the recovery of fracturing fluid constituents is the neutralization of acids (in the fracturing fluids) by carbonates in the subsurface.

Movement of Fluids Outside the Capture Zone

Fracturing fluids injected into the target coal zone flow into fractures under very high pressure. The hydraulic gradients driving fluid flow away from the well during injection are much greater than the hydraulic gradients pulling fluid flow back towards the production well during flowback and production pumping. Some portion of the coalbed methane fracturing fluids could be forced along the hydraulically induced fracture to a point beyond the capture zone of the production well. The size of the capture zone will be affected by the regional groundwater gradients, as well as by the drawdown caused by the well. If fracturing fluids have been injected to a point outside of the well's capture zone, they will not be recovered through production pumping and, if mobile, may be available to migrate through an aquifer. Site-specific geologic, hydrogeologic, injection pressure, and production pumping details would provide the information needed to estimate the dimension of the production well capture zone and the extent to which the fracturing fluids might travel, disperse, and dilute.

Incomplete Mixing of Fracturing Fluids with Water

Steidl (1993) documented the occurrence of a gelling agent that did not dissolve completely and formed clumps at 15 times the injected concentration in the fracture induced by one well. Steidl (1993) also directly observed, in his mined-through studies, gel hanging in stringy clumps in many other fractures induced by that one well. As Willberg et al. (1997) noted, laboratory studies indicate that fingered flow of water past residual gel may impede fluid recovery. Therefore, some fracturing fluid gels appear not to flow with groundwater during production pumping and remain in the subsurface unrecovered. Such gels are unlikely to flow with groundwater during production, but may present a source of gel constituents to flowing groundwater during and after production.

3.4 Measuring and Predicting the Extent of Fluid Movement

Because fractures can possibly connect with or even extend into USDWs, fracture height is relevant to the issue of whether hydraulic fracturing fluids can affect USDWs. Current methods of measuring or predicting fracture growth, including mathematical models, are described. The models are effective in setting parameters for a given hydraulic fracture operation. Coalbed methane well operators have a financial incentive to keep the hydraulically induced fracture generally within the target coal zone so that expenditures on hydraulic horsepower, fracturing fluids, and proppants are minimized for commercial extraction of methane from the coal. In addition, a detailed review is included on "mined-through" studies that were conducted primarily by the U.S. Bureau of Mines. These studies provide unique information on the direct measurement of the dimensions and other characteristics of fractures created in coal seams and surrounding strata by hydraulic fracturing. Paint, injected with the fracturing fluids, was used as a tracer in some of these studies, enabling one of the most direct measurements of the extent of fluid movement due to hydraulic fracturing.

The particular stratigraphy of a fracturing site will determine what fracture heights are significant with respect to USDWs. That is, a given fracture height may be considered small at a particular site in one basin, but may be more significant in another basin where there is a smaller vertical separation between hydraulically fractured coalbeds and a USDW. The extent of fracturing is controlled by the characteristics of the geologic formations (including the presence of shales or natural fractures), the volume and type of fracturing fluid used, the pumping pressure, and the depth at which the fracturing is performed. Several methods are available to operators to measure or predict the extent to which fracture stimulation fluid moves and the related values of maximum induced fracture extension and "propped" fracture height. Propped height (i.e., height in the fracture to which proppant has been distributed) was found to be 60 percent to 75 percent of total vertical fracture height (Mavor et al., 1991; Rahim and Holditch, 1992; Nolte and Smith, 1981; Nolte and Economides, 1991; Zuber et al., 1991). Furthermore, in cases where proppant "screens out" or emplacement partially fails, proppant may exist in 20 percent or less of fracture height.

Both the current and some older methods for estimating fracture dimensions are discussed below. In general, these methods fall into three areas: direct measurements; indirect measurements; and model estimates. Terminology in the literature regarding fracture dimensions is sometimes inconsistent; some articles describe "measured" fracture dimensions when referring to indirect measures or even model estimates.

3.4.1 Direct Measurements

Direct measures include mined-through (or mineback) studies (where mining of subsurface coal seams that were previously hydraulically fractured allows direct access to fractures for measurement); dye tracing conducted in conjunction with mined-through studies; downhole cameras (used to visually inspect fractures in the borehole), including borehole image logging and downhole video logging; surface and downhole tiltmeters; and microseismic monitoring (or imaging). Fracture geometry is most dependably measured by microseismic monitoring or downhole tiltmeters (Warpinski, 2001), or by tracers (Diamond and Oyler, 1987). Downhole cameras can be used only in open bore holes (uncased wells), so fracture measurements using cameras do not reflect conventional coalbed methane fracturing that typically occurs in cased wells. Both downhole cameras and mined-through approaches to fracture measurements are limited to areas exposed by the wellbore and mining activities, respectively. Nonetheless, the mined-through studies provide the most direct approach for estimating fracture dimensions.

Mined-Through Studies

Twenty-two coalbeds were hydraulically fractured, subsequently mined-through, and investigated several months to several years later in Pennsylvania, Alabama, West Virginia, Illinois, Virginia, and Utah (Diamond 1987a and b; Diamond and Oyler 1987). Similar studies have been conducted by Jeffrey et al. (1993) in Queensland, Australia, and Steidl (1991a; 1991b; 1993) in the Black Warrior Basin, Alabama. The Diamond studies were designed to evaluate the effect of the hydraulic fracturing treatment on mining safety. All the mined-through studies enabled direct observation of induced fractures and surrounding material and evaluation of the movement of sand proppant and fracturing fluids through both induced and natural fractures. Eight of the treatments included fluorescent paint in the injected fluid to aid in mapping fluid movement (Diamond 1987a and b).

Steidl (1993) found that fracture widths were typically 0.1 inch, but could be as wide as 4 inches. Measured sand-filled (propped) fractures were 2 to 526 feet in length (Steidl 1993, Jeffrey et al., 1993), although Steidl found a sand-free extension of a sand-filled fracture 870 feet from the borehole. Diamond (1987a and b) found treatment fluids beyond the sand-filled portions of the fractures using paint injected with the fracturing fluids. In most of the wells where paint was injected, the paint was found 200 to 300 feet beyond the sand-filled portions of fractures. However in one borehole, paint extended out from the well bore for 630 feet, although the sand-filled portion of the fracture was only 95 feet in extent (Diamond, 1987a and b). These paint-coated fractures were produced using typical hydraulic fracturing processes in fairly typical coalbed methane geologic conditions.

Fluorescent paint was observed in locations that indicated fluids did not travel in a direct linear path from the induced fracture. Fluids often followed a stair-step pathway through the coalbed (Diamond and Oyler, 1987). The fluorescent paint was also useful for identifying small fractures penetrated by treatment fluids but not by sand proppant. Multiple small, parallel fractures were penetrated by treatment fluids at many of the locations studied. Given that treatment fluids have been documented to travel more than six times farther than sand proppant, studies looking at the dimensions of sand-filled fractures alone are unlikely to capture the extent of fluid movement within and beyond coalbed methane reservoirs (Diamond, 1987a and b).

About half of the sites studied by Diamond (1987a and b) and Diamond and Oyler (1987) had fractures penetrating beyond the coalbeds into the roof rock (the rock overlying the coal in the mined areas). Jeffrey et al. (1993) found that most of the proppant in three of their four treatments was found in the roof rock. Thus, mined-through studies in Australia and in six states in the United States found that hydraulic fracturing fluids penetrated into, and, when shales were very thin, through strata surrounding coalbeds in 50 percent of stimulations in the United States and 75 percent of the stimulations in Australia. The mined-through studies, however, generally cannot provide measures of how far the fractures actually extend, since mining did not extend beyond the coal and into the roof rock.

Other Direct Measurements

A discussion of other fracture diagnostic methods is provided in DOE's paper "Hydraulic Fracturing" (provided as Appendix A).

> "Fracture diagnostics involves analyzing the data before, during and after a hydraulic fracture treatment to determine the shape and dimensions of both the created and propped fracture. Fracture diagnostic techniques have been divided into several groups (Cipolla and Wright, 2000).
>
> **Direct far field techniques**
>
> Direct far field methods are comprised of tiltmeter fracture mapping and microseismic fracture mapping techniques. These techniques require delicate instrumentation that has to be emplaced in boreholes surrounding and near the well to be fracture treated. When a hydraulic fracture is created, the expansion of the fracture will cause the earth around the fracture to deform. Tiltmeters can be used to measure the deformation and to compute the approximate direction and size of the created fracture. Surface tiltmeters are placed in shallow holes surrounding the well to be fracture treated and are best for determining fracture orientation and approximate size. Downhole tiltmeters are placed in vertical wells at depths near the location of the zone to be fracture treated. As with surface tiltmeters, downhole tiltmeter data can be analyzed to determine the orientation and dimensions of the created fracture, but are most useful for determining fracture height. Tiltmeters have been used on an

experimental basis to map hydraulic fractures in coal seams (Nielson and Hanson, 1987).

Microseismic fracture mapping relies on using a downhole receiver array of accelerometers or geophones to locate microseisms or micro-earthquakes that are triggered by shear slippage in natural fractures surrounding the hydraulic fracture. … In essence, noise is created in a zone surrounding the hydraulic fracture. Using sensitive arrays of instruments, the noise can be monitored, recorded, analyzed and mapped.

…Microseismic monitoring has traditionally been too expensive to be used on anything but research wells, but its cost has dropped dramatically in the past few years, so although still expensive (on the order of $50,000 to $100,000), it is being used more commonly throughout the industry. … If the technology is used at the beginning of the development of a field, however, the data and knowledge gained are often used on subsequent wells, effectively spreading out the costs.

Direct near-wellbore techniques

Direct near-wellbore techniques are run in the well that is being fracture treated to locate or image the portion of fracture that is very near (inches) the wellbore. Direct near-wellbore techniques…[include] borehole image logging [and] downhole video logging, and caliper logging. If a hydraulic fracture intersects the wellbore, these direct near-wellbore techniques can be of some benefit in locating the hydraulic fracture.

However, these near-wellbore techniques are not unique and cannot supply information on the size or shape of the fracture once the fracture is 2-3 wellbore diameters in distance from the wellbore. In coal seams, where multiple fractures are likely to exist, the reliability of these direct near-wellbore techniques are even more speculative. As such, very few of these direct near-wellbore techniques are used on a routine basis to look for a hydraulic fracture."

3.4.2 Indirect Measurements

Indirect measures of fracture dimensions include pressure analyses (sometimes referred to as net, treating, or bottom hole pressure analyses that are sometimes analyzed in conjunction with proppant volume assessments) and radioactive tracing. (Radioactive tracing can be conducted on either fracturing fluids or proppants. It is sometimes referred to as a "tagged" study, and is typically measured through gamma ray logging.) Pressure analyses generally monitor bottom hole pressures (BHPs) over time to infer fracture propagation. For example, declining net pressure during water/gel pumping stages indicates rapid fracture height growth (Saulsberry, et al., 1990). Proppant volumes and historical fracturing and methane production data are used to improve estimates based on pressure analyses. Fracture heights and lengths that are inferred by pressure analyses are commonly described in the literature as "measured." Radioactive tracers provide only approximate estimates of fracture dimensions because they are measured in near-wellbore environments.

3.4.3 Model Estimates

The other main category of indirect measures of fracture dimensions is hydraulic fracture modeling. The basic elements of fracture modeling were developed between 1955 and 1961 (Nolte and Economides, 1991). Many modeling studies were conducted to aid in the design of fracture stimulation treatments (i.e., to determine the volume and pump rate of fluids and proppants that are required to achieve a desired fracture geometry).

Model estimates of fracture heights and lengths are common, including estimates using three-dimensional (and quasi-three dimensional) models. Modeling capabilities have advanced considerably in the last several 15 years, and the newest P3D (pseudo 3 dimensional) models simultaneously predict height, width, and length based on treatment input data and reservoir parameters (Olson, 2001). A discussion of indirect fracture modeling techniques is provided by DOE in the "Hydraulic Fracturing" paper (provided as Appendix A). An excerpt from that paper is provided below.

> "The indirect fracture techniques consist of hydraulic fracture modeling of net pressures, pressure transient test analyses, and production data analyses. Because the fracture treatment data and the post-fracture production data are normally available on every well, the indirect fracture diagnostic techniques are the most widely used methods to estimate the shape and dimensions of both the created and the propped hydraulic fracture.
>
> The fracture treatment data can be analyzed with a P3D fracture propagation model to determine the shape and dimensions of the created fracture. The P3D model is used to history match the fracturing data, such as injection rates and injection pressures. Input data, such as the in-situ stress and permeability in key layers of rock can be varied (within reason) to achieve a history match of the field data.
>
> Post-fracture production and pressure data can be analyzed using a 3D reservoir simulator to estimate the shape and dimensions of the propped fracture. Values of formation permeability, fracture length and fracture conductivity can be varied in the reservoir model to achieve a history match of the field data.
>
> The main limitation of these indirect techniques is that the solutions are not unique and require as much fixed data as possible. For example, if the engineer has determined the formation permeability from a well test or production test prior to the fracture treatment, so that the value of formation permeability is

> known and can be fixed in the models, the solution concerning values of fracture length become more unique. Most of the information in the literature concerning post-fracture analyses of hydraulic fractures has been derived from these indirect fracture diagnostic techniques."

There are several caveats regarding the use and interpretation of model estimates. In-situ stress values of the target coal seams and surrounding strata are important model inputs. Actual in-situ stress measurements are very difficult to obtain and are rarely conducted (Warpinski, 2001). Therefore, almost all modeling is conducted using inferred stress values (as estimated, for example, from the mechanical and lithological properties of rocks from, or similar to those in, the target coal zone). Given the geologic variability and site-specific influences on fracture behavior described above, the reliability of fracture height and length estimates obtained from various models is obviously influenced by the quality of the inferred model inputs regarding geologic factors.

Models also necessarily rely on simplifying assumptions to simulate fracture propagation and behavior through sometimes complex geologic zones. As with all modeling, the reliance on inferred input variables and some assumptions introduces some subjectivity to the modeling process. Dependable modeling requires knowledge of and allowance for the detailed stratigraphy of the geologic strata throughout the coal zone. (It was noted in section 3.2.2 that thin clay layers or ash beds can influence fracture behavior.) Simplified geologic models might represent the subsurface as 2 to 3 distinct geologic layers, to reduce computing and data requirements, when a 30- or 50-layer model may be necessary to accurately predict fracture height (Rahim et al., 1998). Nevertheless, models are necessary simplifications of fracture behavior in the geologic subsurface, and significant research has been conducted in the last several decades so that model estimates of fracture behavior in methane-producing coalbeds are now an invaluable tool for industry.

3.4.4 Limitations of Fracture Diagnostic Techniques

Warpinski (1996) discussed many of these same fracture diagnostic techniques. In general, the best fracture diagnostics techniques are expensive and used only in research wells. Fracture diagnostic techniques can provide important data when entering a new production area or a new formation. However, for coalbed methane wells, where costs must be minimized to maintain profitability, the best fracture diagnostic techniques are rarely used and are often considered to be prohibitively expensive.

Warpinksi (2001) further provided other general conclusions regarding estimates of fracture dimensions:

- Fracture heights inferred from pressure data are almost always greater than the corresponding heights measured with the more dependable microseismic monitoring or tiltmeters.

- Actual fracture lengths may be greater or less than the lengths estimated from models or inferred from pressure analyses, depending on many site-specific geologic factors.

- Fracture geometry can be accurately measured using microseismic monitoring and measured somewhat using downhole tiltmeters. These technologies have been found to be invaluable for determining how fractures actually behave.

Table 3-1 lists certain diagnostic techniques and their limitations.

Table 3-1. Limitations of Fracture Diagnostic Techniques (Appendix A: DOE, Hydraulic Fracturing)

Parameter	Technique	Limitation
Fracture Height	Tracer logs	Shallow depth of investigation; shows height only near the wellbore
Fracture Height	Temperature logs	Difficult to interpret; shallow depth of investigation; shows height only near wellbore
Fracture Height	Stress profiling	Does not measure fracture directly; must be calibrated with *in-situ* stress tests
Fracture Height	P3D models	Does not measure fracture directly; estimates vary depending on which model is used
Fracture Height	Microseismic	Optimally requires nearby offset well; difficult to interpret; expensive
Fracture Height	Tiltmeters	Difficult to interpret; expensive and difficult to conduct in the field
Fracture Length	P3D models	Length inferred, not measured; estimates vary greatly depending on which model is used
Fracture Length	Well testing	Large uncertainties depending upon assumptions and lack of prefracture well test data
Fracture Length	Microseismic	Optimally requires nearby offset well; difficult to interpret; expensive
Fracture Length	Tiltmeters	Difficult to interpret; expensive and difficult to conduct in the field
Fracture Azimuth	Core techniques	Expensive to cut core and run tests; multiple tests must be run to assure accuracy
Fracture Azimuth	Log techniques	Requires open hole logs to be run; does not work if natural fractures are not present
Fracture Azimuth	Microseismic	Analysis intensive; expensive for determination of azimuth
Fracture Azimuth	Tiltmeters	Useful only to a depth of 5,000 feet; requires access to large area; expensive
From: Appendix A, DOE, Hydraulic Fracturing		

3.5 Summary

Coalbed methane development began as a safety measure to extract methane, an explosion hazard, from coal prior to mining. Since 1980, coalbed methane production has grown rapidly, spurred by tax incentives to develop non-conventional energy production. At the end of 2000, coalbed methane production from 13 states totaled 1.353 trillion cubic feet, an increase of 156 percent from 1992. At year-end 2000, coalbed methane production accounted for about 7 percent of the total United States dry gas production and 9 percent of proven dry gas reserves (EIA, 2001).

Methane within coalbeds is not "trapped" under pressure as in conventional gas scenarios. Only about 5 to 9 percent of the methane is present as "free" gas within the natural fractures, joints, and cleats. Almost all coalbed methane is adsorbed within the micro-porous matrix of the coal (Koenig, 1989; Winston, 1990; Close, 1993).

Coalbed methane production starts with high-pressure injection of fracturing fluids and proppant into targeted coal zones. The resulting induced or enlarged fractures improve the connections of the production well to the fracture networks in and around the coal zone. When production begins, water is pumped from the fractures in the coal zone to reduce pressure in the formation. When pressures are adequately reduced, methane desorbs from the coal matrix, moves through the network of induced and natural fractures in the coal toward the production well, and is extracted through the well and to the surface.

Fractures that are created at shallow depths (less than approximately 1,000 feet) typically have more of a horizontal than a vertical component. Vertical fractures created at deeper depths can propagate vertically to shallower depths where they may develop a horizontal component. These "T-fractures" may involve the fracture "turning" and developing horizontally at a coalbed-mudstone interface.

Fracture behavior through coal, shale, and other geologic strata commonly present in coal zones depends on site-specific factors such as relative thicknesses and in-situ stress differences between the target coal seam(s) and the surrounding geologic strata, as well as the presence of pre-existing natural fractures. Often, a high stress contrast between adjacent geologic strata results in a barrier to fracture propagation. This occurs in coal zones where there is a geologic contact between a high-stress coal seam and an overlying, thick, relatively low-stress shale.

The fluids used for fracture development are injected at high pressure into the targeted coal zone in the subsurface. These fluids may be "clear" (primarily consisting of water, but may include acid, oil, or water with friction-reducer additives) or "gelled" (viscosity-modified water using guar or other gelling agents). Hydraulic fracturing in coalbed methane wells may require 50,000 to 350,000 gallons of fracturing fluids and 75,000 to 320,000 pounds of sand as proppant to prop or maintain the opening of fractures after the injection (fracturing) pressure is reduced (Holditch et al., 1988 and 1989; Jeu et al., 1988;

Hinkel et al., 1991; Holditch, 1993; Palmer et al., 1991b, 1993a, and 1993b). More typical injection volumes, based on average injection volume data provided by Halliburton for six coalbed methane locations, indicate a maximum average injection volume of 150,000 gal/well and a median average injection volume of 57,500 gal/well (Halliburton, Inc., 2003).

In any fracturing job, some fracturing fluids cannot be recovered and are said to be "lost" to the formation. Palmer (1991a) observed that for fracture stimulations in multi-layered coal formations, 61 percent of stimulation fluids were recovered during a 19-day production sampling of a coalbed methane well in the Black Warrior Basin. He further estimated that from 68 percent to possibly as much as 82 percent would eventually be recovered. A variety of site-specific factors, including leakoff into the coal seams and surrounding strata, the check-valve effect, adsorption and other geochemical processes, and flow through the hydraulic fracture beyond the well's capture zone will serve to reduce recovery of hydraulic fracturing fluids injected into subsurface coal zones to promote coalbed methane extraction.

The mined-through studies by the U.S. Bureau of Mines (see Diamond, 1987a and b) and others provide important directly-measured characteristics of hydraulic fracturing in coal seams and surrounding strata. Further, paint tracer studies conducted as part of Diamond's (1987a and b) mined-through studies can provide estimates on the extent of hydraulic fracturing fluid movement, which may be greater than the extent of sand-filled (propped) hydraulic fracture heights or lengths given fluid movement through natural fractures. These estimates of the extent of fluid movement are usually limited by the area exposed to mining.

A significant amount of diagnostic research has been conducted in the last decade enabling industry to develop a practical, applied understanding of general fracture behavior as it relates to methane production. Operators use a number of techniques to estimate fracture dimensions to design fracture stimulation treatments to minimize expenditures on hydraulic horsepower, fracturing fluids, and proppants. Modeling is increasingly more sophisticated, but still commonly depends on at least some inferred (and subjective) input data. Reliable fracture height and length can be measured accurately by microseismic monitoring and tiltmeters (Warpinksi, 2001).

Figure 3-1. Major United States Coal Basins

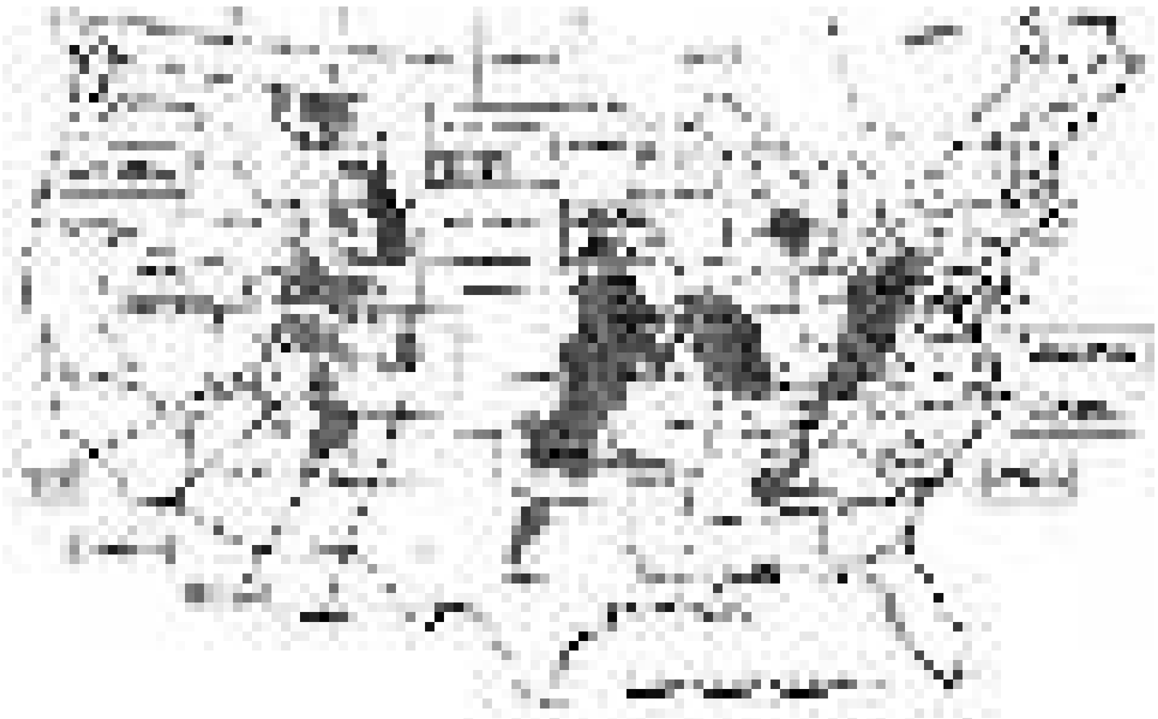


Figure 3-2. Geography of an Ancient Peat-Forming System

Figure 3-3. Schematic Representation of "Face Cleat" (F) and "Butt Cleat" (B)

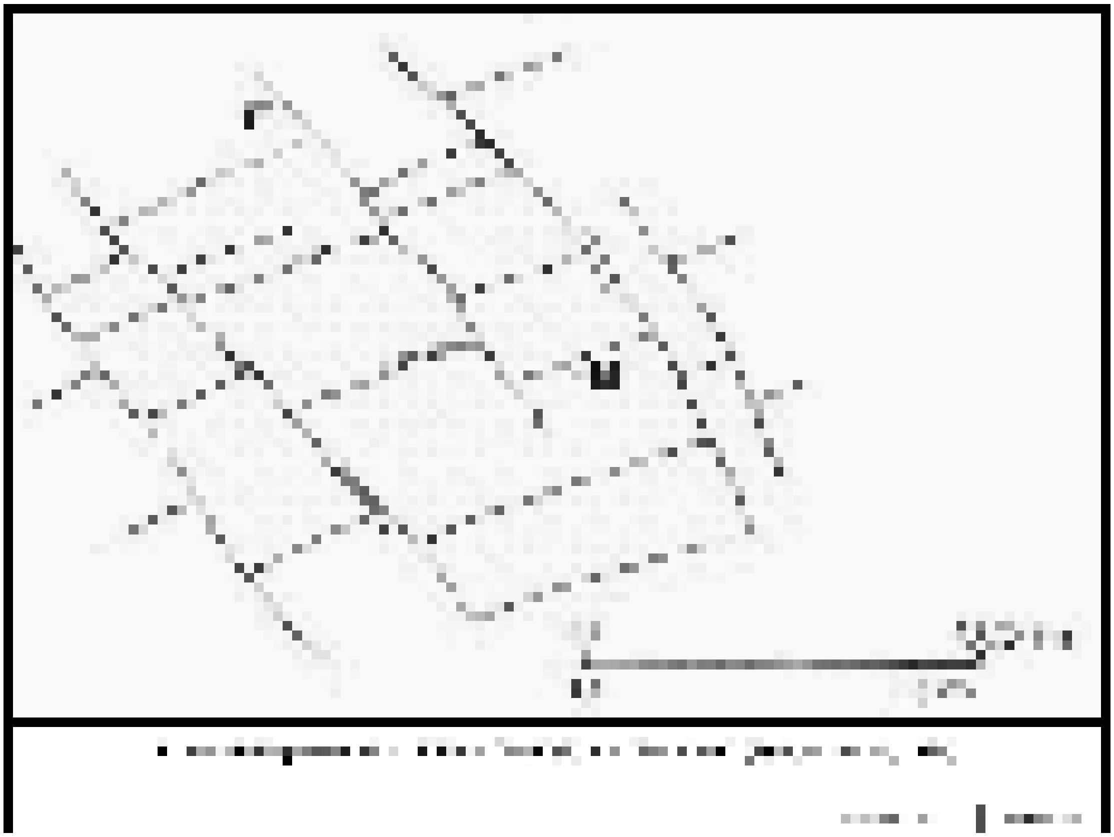


Figure 3-4. A Graphical Representation of the Hydraulic Fracturing Process in Coalbed Methane Wells

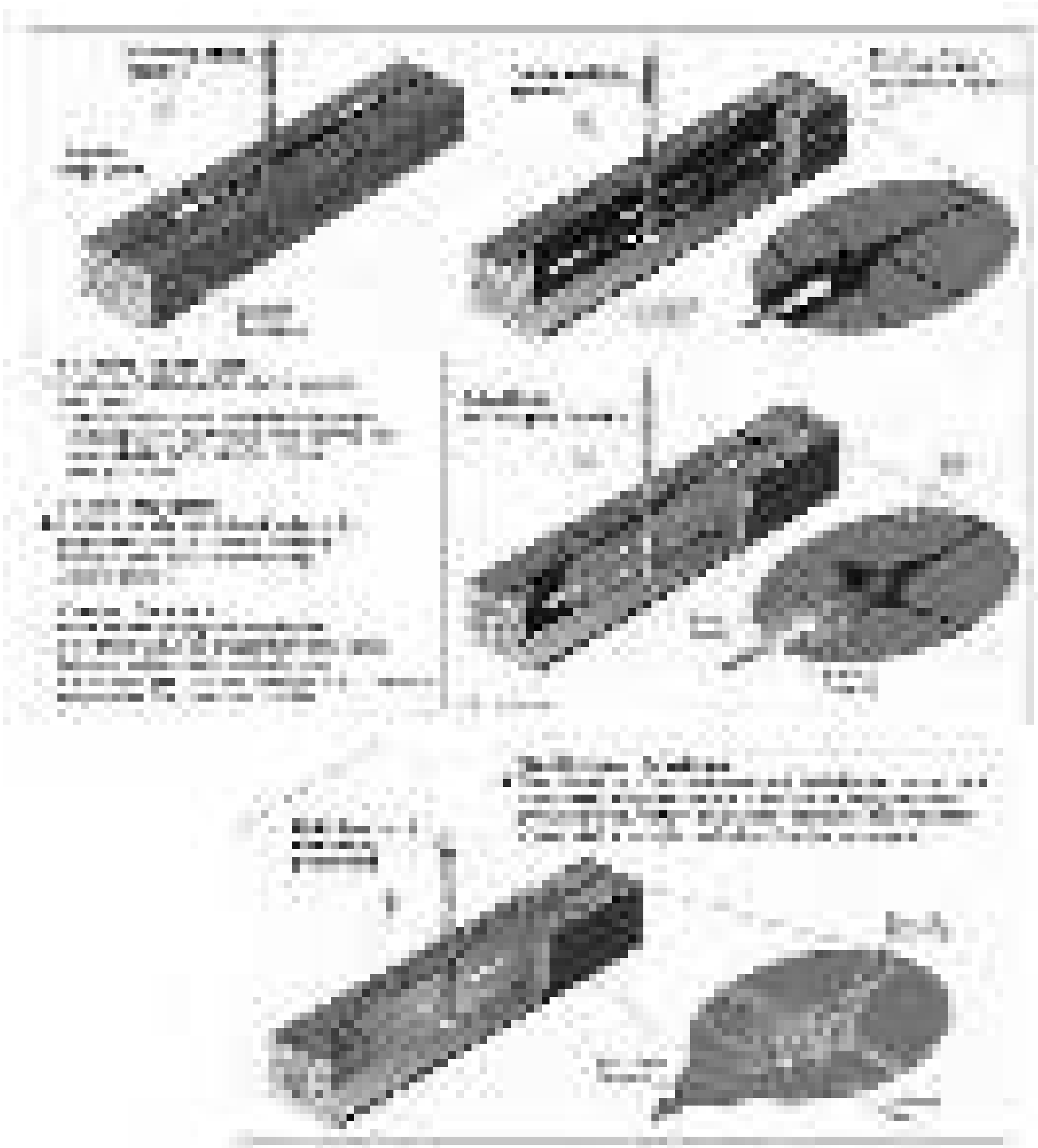


Figure 3-4. A Graphical Representation of the Hydraulic Fracturing Process in Coalbed Methane Wells (Continued)

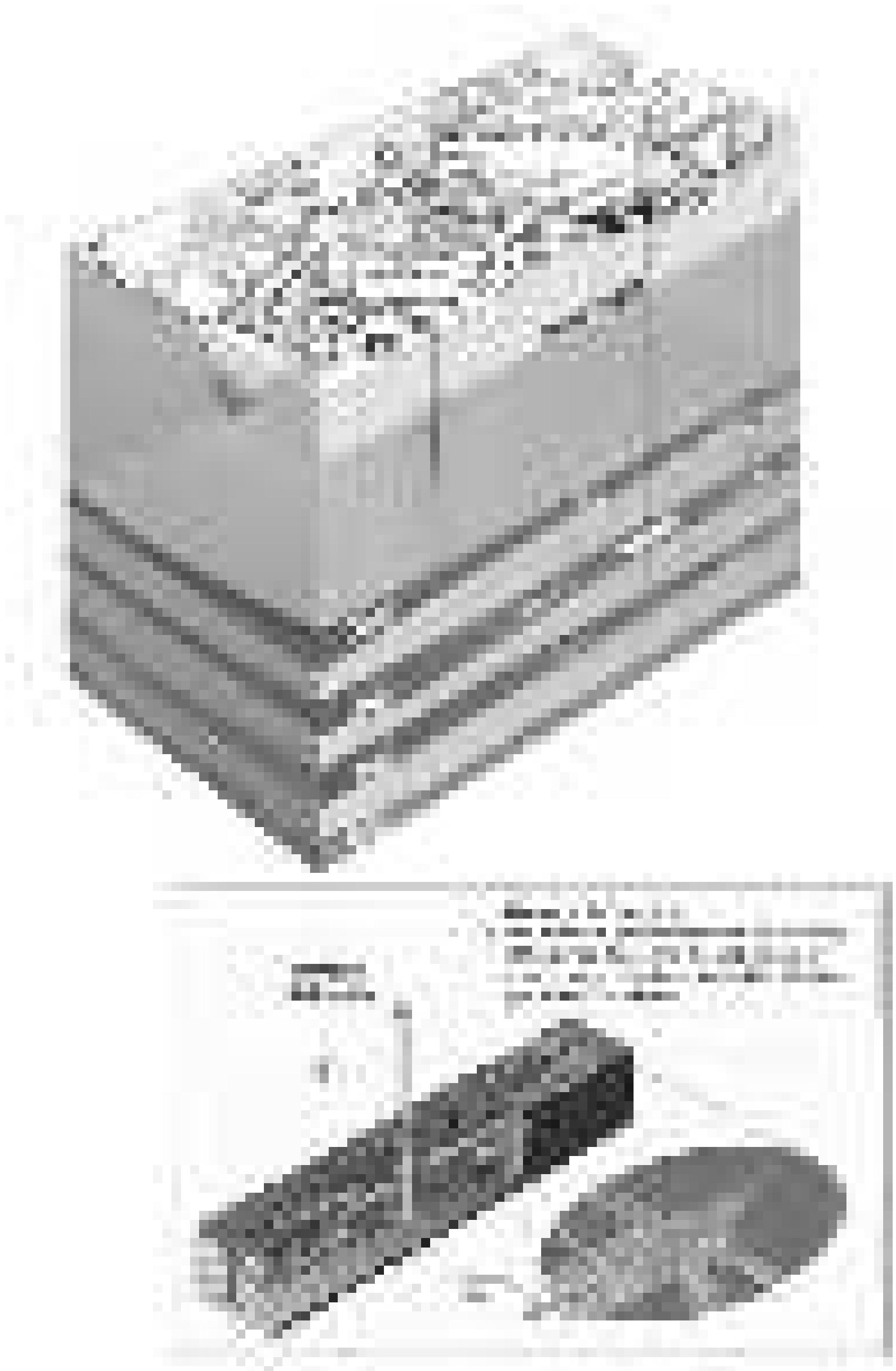


Figure 3-5. Water And Gas Production Over Time

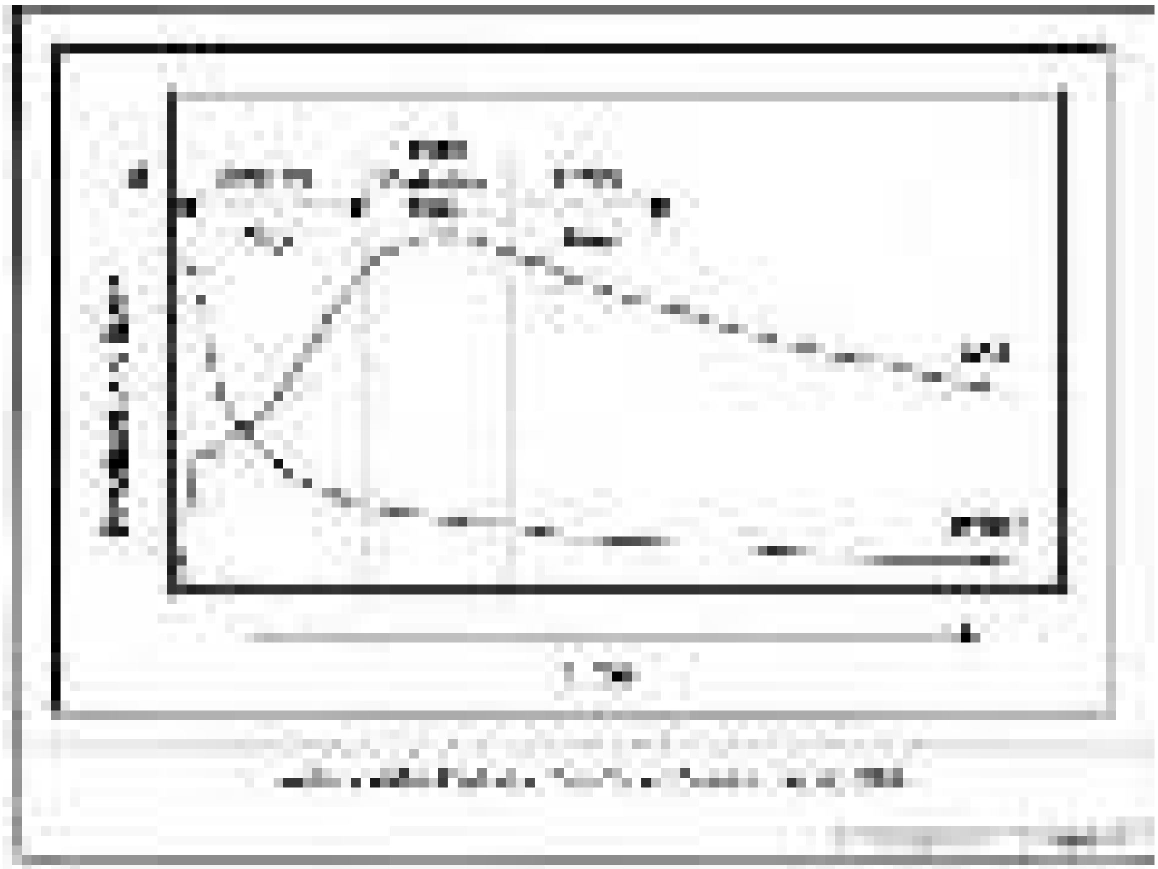


Figure 3-6. Side-View of a Vertical Hydraulic Fracture Typical of Coalbeds

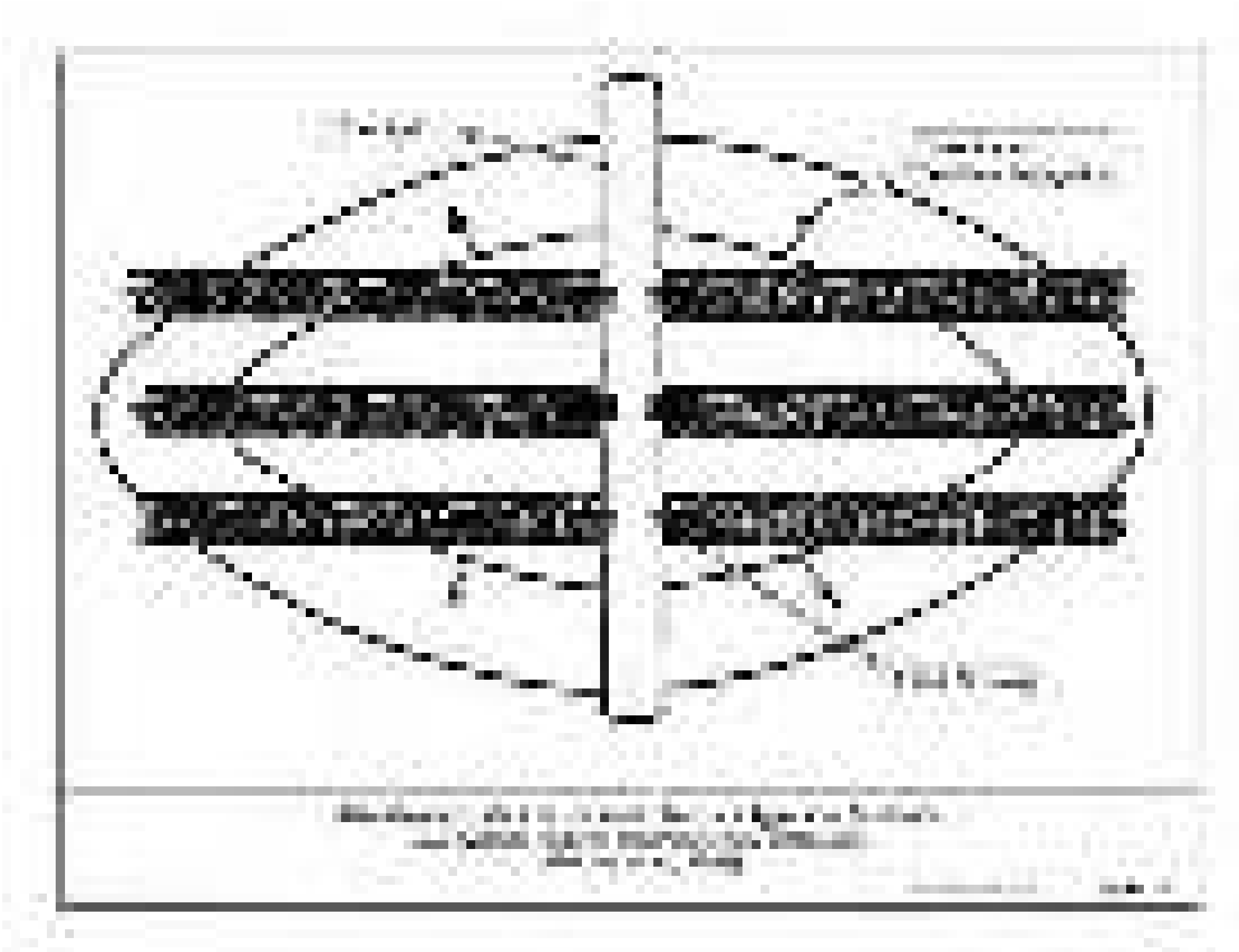


Figure 3-7. Plan View (Looking Down the Wellbore) of Vertical, Two-Winged Coalbed Methane Fracture Showing the Reservoir Region Invaded by Fracturing Fluid Leakoff

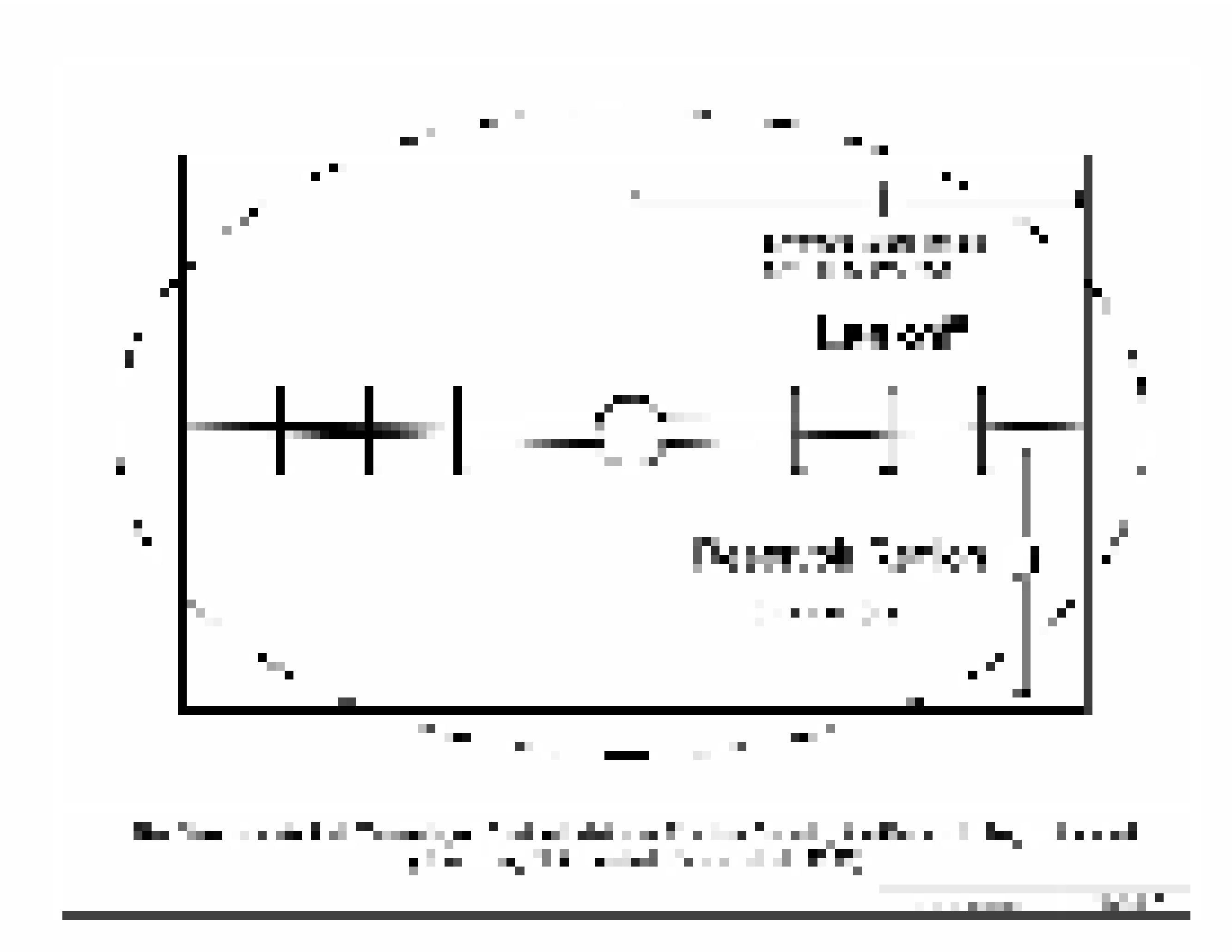
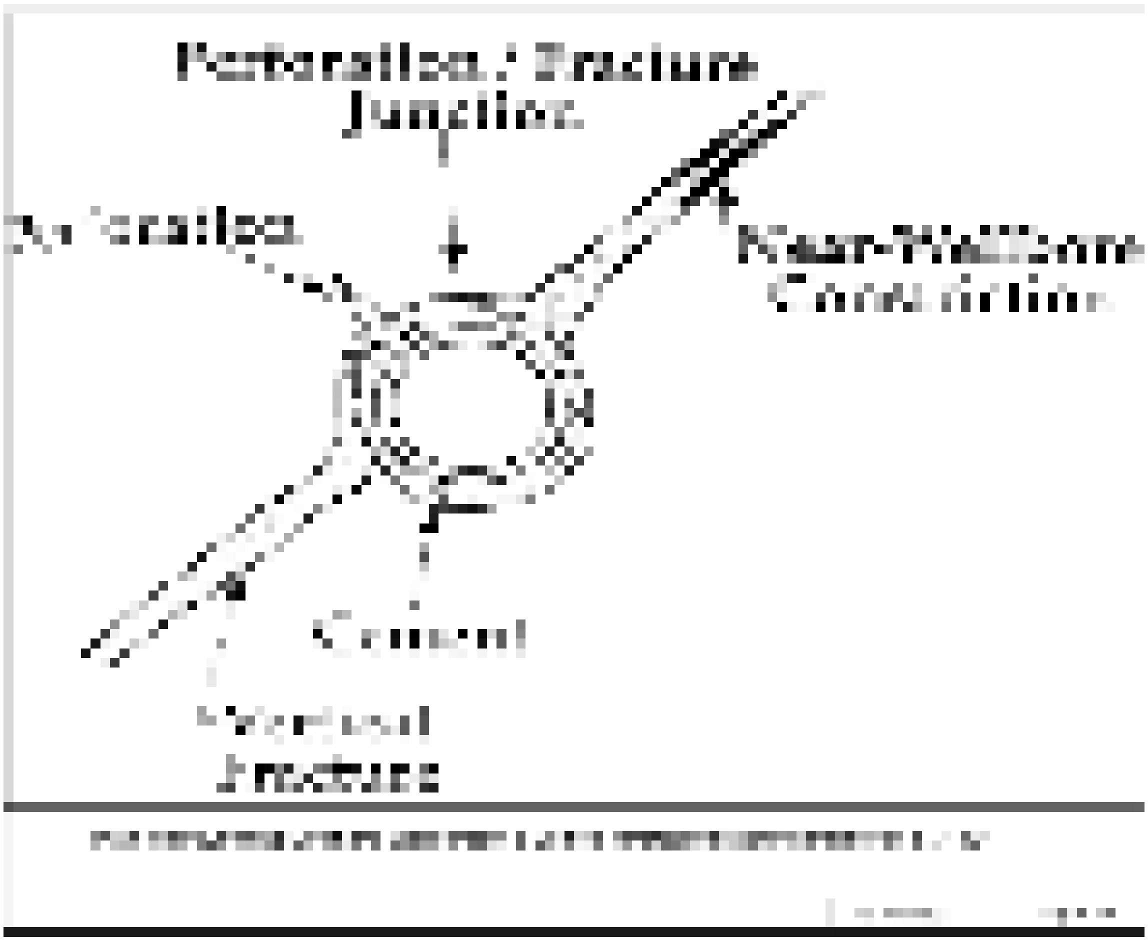


Figure 3-8. Plan View of a Vertical Hydraulic Fracture



Chapter 4
Hydraulic Fracturing Fluids

This chapter summarizes the information EPA collected on the types and volumes of fracturing fluids and additives that may be used for hydraulic fracturing of coalbed methane wells. This chapter also provides EPA's evaluation of the fate and transport of fracturing fluids that are injected into targeted coal layers during the hydraulic fracturing process. This evaluation was conducted to provide the Agency with information on whether a Phase II study is warranted. Captioned photographs in this chapter show the use of fracturing fluids at a coalbed methane well (Figures 4-1 through 4-11 at the end of this chapter).

4.1 Introduction

The types and use of fracturing fluids have evolved greatly over the past 60 years and continue to evolve. The U.S. oil and gas industry has used fluids for fracturing geologic formations since the early 1940s (Ely, 1985). *The Handbook of Stimulation Engineering* (Ely, 1985), a comprehensive history of the evolution of hydraulic fracturing fluids in the oil and gas industry, was used as a source of information for this chapter. In addition, EPA identified fluids and fluid additives commonly used in hydraulic fracturing through literature searches, reviews of relevant MSDSs provided by service companies, and discussions with field engineers, service company chemists, and state and federal employees.

Available scientific literature indicates that hydraulic fracturing fluid performance became a prevalent research topic in the late 1980s and the 1990s. Most of the literature pertaining to fracturing fluids relates to the fluids' operational efficiency rather than their potential environmental or human health impacts. There is very little documented research on the environmental impacts that result from the injection and migration of these fluids into subsurface formations, soils, and USDWs. Some of the existing literature does offer information regarding the basic chemical components present in most of these fluids. The composition of fracturing fluids and additives is discussed in detail in the next section.

The main goal of coalbed hydraulic fracturing is to create a highly conductive fracture system that will allow flow through the methane-bearing coal zone to the production well used to extract methane (and groundwater). Hydraulic fracturing fluids are used to initiate and/or expand fractures, as well as to transport proppant into fractures in coalbed formations. Proppants are sand or other granular substances injected into the formation to hold or "prop" open coal formation fractures created by hydraulic fracturing. The viscosity of fracturing fluids is considered when they are formulated, to provide for efficient transport and placement of proppant into a fracture. Most of the fracturing fluids injected into the formation are pumped back out of the well along with groundwater and methane gas (see section 3.3 in Chapter 3 for a more detailed discussion of fracturing fluid recovery).

4.2 Types of Fracturing Fluids and Additives

Service companies have developed a number of different oil- and water-based fluids and treatments to more efficiently induce and maintain permeable and productive fractures. The composition of these fluids varies significantly, from simple water and sand to complex polymeric substances with a multitude of additives. Each type of fracturing fluid has unique characteristics, and each possesses its own positive and negative performance traits. For ideal performance, fracturing fluids should possess the following four qualities (adapted from Powell et al., 1999):

- Be viscous enough to create a fracture of adequate width.
- Maximize fluid travel distance to extend fracture length.
- Be able to transport large amounts of proppant into the fracture.
- Require minimal gelling agent to allow for easier degradation or “breaking” and reduced cost.

Water-based fracturing fluids have become the predominant type of coalbed methane fracturing fluid (Appendix A: DOE, Hydraulic Fracturing). However, fracturing fluids can also be based on oil, methanol, or a combination of water and methanol. Methanol is used in lieu of, or in conjunction with, water to minimize fracturing fluid leakoff and enhance fluid recovery (Thompson et al., 1991). Polymer-based fracturing fluids made with methanol usually improve fracturing results, but require 50 to 100 times the amount of breaker (e.g., acids used to degrade the fracturing fluid viscosity, which helps to enhance post-fracturing fluid recovery) (Ely, 1985). In some cases, nitrogen or carbon dioxide gas is combined with the fracturing fluids to form foam as the base fluid. Foams require substantially lower volumes to transport an equivalent amount of proppant. Diesel fuel is another component of some fracturing fluids although it is not used as an additive in all hydraulic fracturing operations. A variety of other fluid additives (in addition to the proppants) may be included in the fracturing fluid mixture to perform essential tasks such as formation clean up, foam stabilization, leakoff inhibition, or surface tension reduction. These additives include biocides, fluid-loss agents, enzyme breakers, acid breakers, oxidizing breakers, friction reducers, and surfactants such as emulsifiers and non-emulsifiers. Several products may exist in each of these categories. On any one fracturing job, different fluids may be used in combination or alone at different stages in the fracturing process. Experienced service company engineers will devise the most effective fracturing scheme, based on formation characteristics, using the fracturing fluid combination they deem most effective.

The main fluid categories are:

- Gelled fluids, including linear or cross-linked gels.
- Foamed gels.
- Plain water and potassium chloride (KCl) water.
- Acids.
- Combination treatments (any combination of 2 or more of the aforementioned fluids).

Some of the fluids and fluid additives may contain constituents of potential concern. Table 4-1, at the end of section 4.2.6, lists examples of chemicals found in hydraulic fracturing fluids according to the MSDSs provided by service companies, and potential human health effects associated with the product. It is important to note that information presented in MSDSs is for pure product. Each of the products listed in Table 4-1 is significantly diluted prior to injection.

EPA also obtained two environmental impact statements that were prepared by the Bureau of Land Management (BLM). In these impact statements, BLM identified additional chemical compounds that may be in fracturing fluids including methyl tert butyl ether (MTBE) (U.S. Department of the Interior, CO State BLM, 1998). However, EPA was unable to find any indications in the literature, on MSDSs, or in interviews with service companies that MTBE is used in fracturing fluids to stimulate coalbed methane wells.

4.2.1 Gelled Fluids

Water alone is not always adequate for fracturing certain formations because its low viscosity limits its ability to transport proppant. In response to this problem, the industry developed linear and cross-linked fluids, which are higher viscosity fracturing fluids. Water gellants or thickeners are used to create these gelled fluids. Gellant selection is based on formation characteristics such as pressure, temperature, permeability, porosity, and zone thickness. These gelled fluids are described in more detail below.

Linear Gels

A substantial number of fracturing treatments are completed using thickened, water-based linear gels. The gelling agents used in these fracturing fluids are typically guar gum, guar derivatives such as hydroxypropylguar (HPG) and carboxymethylhydroxypropylguar (CMHPG), or cellulose derivatives such as carboxymethylguar or hydroxyethylcellulose (HEC). In general, these products are biodegradable. Guar is a polymeric substance derived from the seed of the guar plant

(Ely, 1985). Guar gum, on its own, is non-toxic and, in fact, is a food-grade product commonly used to increase the viscosity and elasticity of foods such as ice cream.

To formulate a viscous fracturing gel, guar powder or concentrate is dissolved in a carrier fluid such as water or diesel fuel. Increased viscosity improves the ability of the fracturing fluid to transport proppant and decreases the need for more turbulent flow. Concentrations of guar gelling agents within fracturing fluids have decreased over the past several years. It was determined that reduced concentrations provide better and more complete fractures (Powell et al., 1999).

Diesel fuel has been frequently used in lieu of water to dissolve the guar powder because its carrying capacity per unit volume is much higher (Halliburton, Inc., 2002). "Diesel is a common solvent additive, especially in liquid gel concentrates, used by many service companies for continuous delivery of gelling agents in fracturing treatments" (GRI, 1996). Diesel does not enhance the efficiency of the fracturing fluid; it is merely a component of the delivery system (Halliburton, Inc., 2002). Using diesel instead of water minimizes the number of transport vehicles needed to carry the liquid gel to the site (Halliburton, Inc., 2002).

The percentage of diesel fuel in the slurried thickener can range between 30 percent and almost 100 percent, based on the MSDSs summarized in Table 4-1. Diesel fuel is a petroleum distillate and may contain known carcinogens. One such component of diesel fuel is benzene, which, according to literature sources, can make up anywhere between 0.003 percent and 0.1 percent by weight of diesel fuel (Clark and Brown, 1977; R. Morrison & Associates, Inc., 2001). Slurried diesel and gel are diluted with water prior to injection into the subsurface. The dilution is approximately 4 to 10 gallons of concentrated liquid gel (guar slurried in diesel) per 1,000 gallons of make-up water to produce an adequate polymer slurry (Halliburton, Inc., Virginia Site Visit, 2001; Schlumberger, Ltd., 2001; Consolidated Industrial Services, Inc., Virginia Site Visit, 2001; BJ Services, 2001).

Cross-linked Gels

One major advance in fracturing fluid technology was the development of cross-linked gels. The first cross-linked gels were developed in 1968 (Ely, 1985). When cross-linking agents are added to linear gels, the result is a complex, high-viscosity fracturing fluid that provides higher proppant transport performance than do linear gels (Messina, Inc. Web site, 2001; Ely, 1985; Halliburton Inc., Virginia Site Visit, 2001). Cross-linking reduces the need for fluid thickener and extends the viscous life of the fluid indefinitely. The fracturing fluid remains viscous until a breaking agent is introduced to break the cross-linker and, eventually, the polymer. Although cross-linkers make the fluid more expensive, they can considerably improve hydraulic fracturing performance, hence increasing coalbed methane well production rates.

Cross-linked gels are typically metal ion-cross-linked guar (Ely, 1985). Service companies have used metal ions such as chromium, aluminum, titanium, and other metal ions to achieve cross-linking (Ely, 1985). In 1973, low-residue (cleaner) forms of cross-linked gels, such as cross-linked hydroxypropylguar, were developed (Ely, 1985).

According to MSDSs summarized in Table 4-1, cross-linked gels may contain boric acid, sodium tetraborate decahydrate, ethylene glycol, and monoethylamine. These constituents are hazardous in their undiluted form and can cause kidney, liver, heart, blood, and brain damage through prolonged or repeated exposure. According to a BLM environmental impact statement, cross-linkers may contain hazardous constituents such as ammonium chloride, potassium hydroxide, zirconium nitrate, and zirconium sulfate (U.S. Department of the Interior, CO State BLM, 1998). Concentrations of these compounds in the fracturing fluids were not reported in the impact statement. The final concentration of cross-linkers is typically 1 to 2 gallons of cross-linker per 1,000 gallons of gel (Halliburton, Inc., Virginia Site Visit, 2001; Schlumberger, Ltd., 2001).

4.2.2 Foamed Gels

Foam fracturing technology uses foam bubbles to transport and place proppant into fractures. The most widely used foam fracturing fluids employ nitrogen or carbon dioxide as their base gas. Incorporating inert gases with foaming agents and water reduces the amount of fracturing liquid required. Foamed gels use fracturing fluids with higher proppant concentrations to achieve highly effective fracturing. The gas bubbles in the foam fill voids that would otherwise be filled by fracturing fluid. The high concentrations of proppant allow for an approximately 75-percent reduction in the overall amount of fluid that would be necessary using a conventional linear or cross-linked gel (Ely, 1985; Halliburton, Inc., Virginia Site Visit, 2001). Foaming agents can be used in conjunction with gelled fluids to achieve an extremely effective fracturing fluid (Halliburton, Inc., Virginia Site Visit, 2001).

Foam emulsions experience high leakoff; therefore, typical protocol involves the addition of fluid-loss agents, such as fine sands (Ely, 1985; Halliburton, Virginia Site Visit, 2001). Foaming agents suspend air, nitrogen, or carbon dioxide within the aqueous phase of a fracturing treatment. The gas/liquid ratio determines if a fluid will be true foam or simply a gas-energized liquid (Ely, 1985). Carbon dioxide can be injected as a liquid, whereas nitrogen must be injected as a gas to prevent freezing (Halliburton, Inc., Virginia Site Visit, 2001).

According to the MSDSs summarized in Table 4-1, foaming agents can contain diethanolamine and alcohols such as isopropanol, ethanol, and 2-butoxyethanol. They can also contain hazardous substances including glycol ethers (U.S. Department of the Interior, CO State BLM, 1998). One of the foaming agent products listed in Table 4-1 can cause negative liver and kidney effects, although the actual component causing these effects is not specified on the MSDS. The final concentration is typically 3 gallons of foamer per 1,000 gallons of gel (Halliburton, Inc., Virginia Site Visit, 2001; Schlumberger, Ltd., 2001).

4.2.3 Water & Potassium Chloride Water Treatments

Many service companies use groundwater pumped directly from the formation or treated water for their fracturing jobs. In some coalbed methane well stimulations, proppants are not needed to prop fractures open, so simple water or slightly thickened water can be a cost-effective substitute for an expensive polymer or foam-based fracturing fluid with proppant (Ely, 1985). Hydraulic fracturing performance is not exceptional with plain water, but, in some cases, the production rates achieved are adequate. Plain water has a lower viscosity than gelled water, which reduces proppant transport capacity.

Similar to plain water, another fracturing fluid uses water with potassium chloride (KCl) in addition to small quantities of gelling agents, polymers, and/or surfactants (Ely, 1985). Potassium chloride is harmless if ingested at low concentrations.

4.2.4 Acids

Acids are used in limestone formations that overlay or are interbedded within coals to dissolve the rock and create a conduit through which formation water and coalbed methane can travel (Ely, 1985). Typically, the acidic stimulation fluid is hydrochloric acid or a combination of hydrochloric and acetic or formic acid. For acid fracturing to be successful, thousands of gallons of acid must be pumped far into the formation to etch the face of the fracture (Ely, 1985). Some of the cellulose derivatives used as gelling agents in water and water/methanol fluids can be used in acidic fluids to increase treatment distance (Ely, 1985). As discussed in section 4.2.5, acids may also be used as a component of breaker fluids.

In addition, acid can be used to clean up perforations of the cement surrounding the well casing prior to fracturing fluid injection (Halliburton, Inc., Virginia Site Visit, 2001; Halliburton, Inc., 2002). The cement is perforated at the desired zone of injection to ease fracturing fluid flow into the formation (Halliburton, Inc., Virginia Site Visit, 2001; Halliburton, Inc., 2002).

Table 4-1 provides information on formic and hydrochloric acids. Acids are corrosive, and can be extremely hazardous in concentrated form. Acids are substantially diluted with water-based or water-and-gas-based fluids prior to injection into the subsurface. The injected concentration is typically 1,000 times weaker than the concentrated versions presented in the product MSDSs (Halliburton, Inc., Virginia Site Visit, 2001; Schlumberger, Ltd., 2001).

4.2.5 Fluid Additives

Several fluid additives have been developed to enhance the efficiency and increase the success of fracturing fluid treatments. The major categories of these additives are defined and briefly described in the following sections.

Breakers

Breaker fluids are used to degrade the fracturing fluid viscosity, which helps to enhance post-fracturing fluid recovery, or flowback. Breakers can be mixed with the fracturing fluid during pumping, or they can be introduced later as an independent fluid. There are a variety of breaker types including time-release and temperature-dependent types. Most breakers are typically acids, oxidizers, or enzymes (Messina, Inc. Web site, 2001). According to a BLM environmental impact statement, breakers may contain hazardous constituents, including ammonium persulfate, ammonium sulphate, copper compounds, ethylene glycol, and glycol ethers (U.S. Department of the Interior, CO State BLM, 1998). Concentrations of these compounds in the fracturing fluids were not presented in the environmental impact statement.

Biocides

One hydraulic fracturing design problem that arises when using organic polymers in fracturing fluids is the incidence of bacterial growth within the fluids. Due to the presence of organic constituents, the fracturing fluids provide a medium for bacterial growth. As the bacteria grow, they secrete enzymes that break down the gelling agent, which reduces the viscosity of the fracturing fluid. Reduced viscosity translates into poor proppant placement and poor fracturing performance. To alleviate this degradation in performance, biocides, bactericides, or microbicides are added to the mixing tanks with the polymeric gelling agents to kill any existing microorganisms (e.g., sulfate-reducing bacteria, slime-forming bacteria, algae), and to inhibit bacterial growth and deleterious enzyme production. Bactericides are typically hazardous by nature (Messina, Inc. Web site, 2001). They may contain hazardous constituents, including polycyclic organic matter (POM) and polynuclear aromatic hydrocarbons (PAHs) (U.S. Department of the Interior, CO State BLM, 1998).

Information from MSDSs for a biocide and a microbicide is summarized in Table 4-1. These concentrated products are substantially diluted prior to injection into the subsurface. Typical dilution in the make-up water is 0.1 to 0.2 gallons of microbicide in 1,000 gallons of water (Halliburton, Inc., Virginia Site Visit, 2001; Schlumberger, Ltd., 2001).

Fluid-Loss Additives

Fluid-loss additives restrict leakoff of the fracturing fluid into the exposed rock at the fracture face. Because the additives prevent excessive leakoff, fracturing fluid effectiveness and integrity are maintained. Fluid-loss additives of the past and present include bridging materials such as 100 mesh sand, 100 mesh soluble resin, and silica flour, or plastering materials such as starch blends, talc silica flour, and clay (Ely, 1985).

Friction Reducers

To optimize the fracturing process, water-based fluids must be pumped at maximum rates and fluids must be injected at maximum pressures. Increasing flow velocities and pressures in this manner can lead to undesirable levels of friction within the injection well and the fracture itself. In order to minimize friction, friction reducers are added to water-based fracturing fluids. The friction reducers are typically latex polymers or copolymers of acrylamides. They are added to slick water treatments (water with solvent) at concentrations of 0.25 to 2.0 pounds per 1,000 gallons (Ely, 1985). Some examples of friction reducers are oil-soluble anionic liquid, cationic polyacrilate liquid, and cationic friction reducer (Messina, Inc. Web site, 2001).

Acid Corrosion Inhibitors

Corrosion inhibitors are required in acid fluid mixtures because acids will corrode steel tubing, well casings, tools, and tanks. The solvent acetone is a common additive in corrosion inhibitors (GRI, 1996). Information from MSDSs for acid inhibitors is summarized in Table 4-1. These products can affect the liver, kidney, heart, central nervous system, and lungs. They are quite hazardous in their undiluted form. These products are diluted to a concentration of 1 gallon per 1,000 gallons of make-up water and acid mixture (Halliburton, Inc., Virginia Site Visit, 2001; Schlumberger, Ltd., 2001). Acids and acid corrosion inhibitors are used in very small quantities in coalbed methane fracturing operations (500 to 2,000 gallons per treatment).

4.2.6 Proppants

The purpose of a proppant is to prop open a hydraulic fracture. An ideal proppant should produce maximum permeability in a fracture. Fracture permeability is a function of proppant grain roundness, proppant purity, and crush strength. Larger proppant volumes allow for wider fractures, which facilitate more rapid flowback to the production well. Over a period of 30 minutes, 4,500 to 15,000 gallons of fracturing fluid will typically transport and place approximately 11,000 to 25,000 pounds of proppant into the fracture (Powell et al., 1999).

4.3 The Fate and Transport of Stimulation Fluids Injected into Coal and Surrounding Rock During Hydraulic Fracturing of Coalbed Methane Reservoirs (with a Special Focus on Diesel Fuel)

Diesel fuel is sometimes a component of gelled fluids. Diesel fuel contains constituents of potential concern regulated under SDWA – benzene, toluene, ethylbenzene, and xylenes (i.e., BTEX compounds). The use of diesel fuel in fracturing fluids poses the greatest threat to USDWs because BTEX compounds in diesel fuel exceed the MCL at the point-of-injection (i.e. the subsurface location where fracturing fluids are initially injected).

The remainder of this section presents EPA's qualitative evaluation of the fate and transport of fracturing fluids injected into targeted coal layers in the subsurface during hydraulic fracturing. Although EPA's MOA with the three major service companies has largely eliminated diesel fuel from fracturing fluids injected directly into any USDWs, there may still be rare instances in which diesel fuel is used by other service companies or operators (USEPA, 2003). Therefore an evaluation of the use of diesel fuel in fracturing fluid, which also provides follow-up on the draft of this report published in August, 2002, is included in this chapter.

EPA revised its procedure for assessing the potential effects of fracturing fluid constituents on USDWs from the procedure presented in the August 2002 draft of this report as follows:

- EPA has revised the fraction of BTEX compounds in diesel used to estimate the point-of-injection concentrations from a single value to a documented broader range of values for the fraction of BTEX in diesel fuel. For example, the fraction of benzene in diesel was revised from 0.00006$g_{benzene}/g_{diesel}$ to a range with a minimum value of 0.000026 $g_{benzene}/g_{diesel}$ and a maximum value of 0.001 $g_{benzene}/g_{diesel}$. If the maximum value for benzene in diesel is used to estimate the concentration of benzene at the point-of-injection, the resulting estimate is 17 times higher than that presented in the Draft Report.

- In this report, EPA used more current values for two of the parameters used to estimate the point-of-injection concentrations of BTEX compounds. Specifically, the estimates in this report use a density of the diesel fuel-gel mixture of 0.87 g/mL compared to 0.84 g/mL in the Draft Report, and a fraction of diesel fuel in gel of 0.60 g_{diesel}/g_{gel} compared to 0.52 g_{diesel}/g_{gel} in the Draft Report. The use of these more current values does not affect the order of magnitude of the revised point-of-injection calculations.

- The August 2002 Draft Report included estimates of the concentration of benzene at an idealized, hypothetical edge of the fracture zone located 100 feet from the point-of-injection. Based on new information and stakeholder input, EPA concluded that the edge of fracture zone calculation is not an appropriate model for reasons including:

- Mined-through studies reviewed by EPA indicated that hydraulic fracturing injection fluids had traveled several hundred feet beyond the point-of-injection.

- The assumption of well-mixed concentrations within the idealized fracture zone is insufficient. One mined-through study indicated an observed concentration of gel in a fracture that was 15 times the injected concentration, with gel found to be hanging in stringy clumps in many fractures. The variability in gel distribution in hydraulic fractures indicates that the gel constituents are unlikely to be well mixed in groundwater.

- Based on more extensive review of the literature, the width of a typical fracture was estimated to be much thinner than that used in the Draft Report (0.1 inch versus 2 inches). The impact of the reduced width of a typical fracture is that the calculated volume of fluid that can fit within a fracture is less. After an initial volume calculation using the new width, EPA found that the volume of the space within the fracture area may not hold the volume of fluid pumped into the ground during a typical fracturing event. Therefore, EPA assumes that a greater volume of fracturing fluid must "leakoff" to intersecting smaller fractures than what was assumed in the Draft Report, or that fluid may move beyond the idealized, hypothetical "edge of fracture zone." This assumption is supported by field observations in mined-through studies, which indicate that fracturing fluids often take a stair-step transport path through the natural fracture system.

- In the Draft Report, EPA approximated the edge of fracture zone concentrations considering only dilution. Based on new information and stakeholder input on the Draft Report, EPA does not provide estimates of concentrations beyond the point-of-injection in the final report. Developing such concentration values with the precision required to compare them to MCLs would require the collection of significant amounts of site-specific data. This data in turn would be used to perform a formal risk assessment, considering numerous fate and transport scenarios. These activities are beyond the scope of this Phase I study.

The remainder of this section includes a discussion of the following components of EPA's analysis:

- The concentrations of BTEX at the point-of-injection.

- The percentage of fracturing fluids recovered during the recovery process.

- The influence of the capture zone.

- Factors that would increase or decrease the concentrations of BTEX remaining in the subsurface.

The first step in EPA's analysis of the potential threat to USDWs from the injection of fracturing fluids was calculating the point-of-injection concentrations of BTEX introduced from diesel fuel in the gelling agent. In Step 2, EPA considered factors that affect the degree to which hydraulic fracturing fluids are recovered. Steps 3, 4, and 5 provide analyses of physical/chemical, hydrogeological, and biological processes that could affect the fate and transport of hazardous chemicals introduced into coal seams. These steps are summarized in Table 4-2.

4.3.1 Point-of-Injection Calculation

The formulations or "recipes" for fracturing fluids differ among service companies and among sites; the amount of fracturing fluid used will also vary. Thus, a range of point-of-injection concentrations likely exists. According to field paperwork obtained during EPA's site visits (Consolidated Industrial Services, Inc., 2001; Halliburton, 2001) and information provided by three service company scientists (BJ Services, 2001; Halliburton, 2001; Schlumberger, Ltd., 2001), between 4 and 10 gallons of diesel-containing gelling agent are added to each 1,000 gallons of water used in hydraulic fracturing, when diesel is used. In addition, the fraction of BTEX in diesel may range by up to two orders of magnitude (Potter and Simmons, 1998). The lower and upper ranges of the values presented in Potter and Simmons (1998), as well as the three different values cited for gelling agent, were used to estimate point-of-injection concentrations for each of three fracturing fluid recipes (i.e., the ratio of fracturing gel to water). The resulting 24 point-of-injection calculations are provided in Table 4-2. These estimates provide the basis for a qualitative assessment regarding whether a Phase II study is warranted.

The following example illustrates how EPA estimated the concentrations of BTEX at the point-of-injection. Due to the variations in the recipe used by service companies, EPA's analysis begins with three different possible scenarios, as follows:

- Low ratio: 4 gallons of gel per 1,000 gallons of water
- Medium ratio: 6 gallons of gel per 1,000 gallons of water
- High ratio: 10 gallons of gel per 1,000 gallons of water

The concentration of benzene in fracturing fluid at the point-of-injection ($[\text{benzene}]_{inj}$) can be calculated using the following equation:

$$[\text{benzene}]_{inj} = (r_{gw}) \times (\rho_{dg}) \times (f_{dg}) \times (f_{bd}) \times (3{,}785\ mL_{gel}/gal_{gel}) \times (1\ gal_{water}/3.785\ L_{water}) \times (10^6\ \mu g/g)$$

Where:

r_{gw} = the ratio of diesel fuel-gel mixture to injection water ($gal_{gel}/1{,}000\ gal_{water}$) ($4\ gal_{gel}/1{,}000 gal_{water}$, $6 gal_{gel}/1{,}000\ gal_{water}$, and $10\ gal_{gel}/1{,}000\ gal_{water}$ represent the low, medium, and high ratios, respectively)

ρ_{dg} = the density of the diesel fuel-gel mixture (g_{gel}/mL_{gel}) = 0.84 g_{gel}/mL_{gel} (Halliburton, 2002)

f_{dg} = the fraction of diesel fuel in the gel (g_{diesel}/g_{gel}) = 0.52 g_{diesel}/g_{gel} (Halliburton, 2002)

f_{bd} = the fraction of benzene in diesel fuel ($g_{benzene}/g_{diesel}$) = 0.000026 to 0.001 $g_{benzene}/g_{diesel}$ (Potter and Simmons, 1998)

$3{,}785\ mL_{gel}/gal_{gel}$ = volume conversion factor

$1\ gal_{water}/3.785\ L_{water}$ = volume conversion factor

$10^6\ \mu g/g$ = mass conversion factor

The concentration of benzene at the point-of-injection is calculated for the three gel/water ratios and the minimum and maximum concentrations of benzene in diesel fuel.

Using $r_{gw} = 4\ gal_{gel}/1{,}000 gal_{water}$ and $f_{bd} = 0.000026\ g_{benzene}/g_{diesel}$ as an example, $[\text{benzene}]_{inj}$ is calculated as follows:

$$[\text{benzene}]_{inj} = (4\ gal_{gel}/1{,}000 gal_{water}) \times (0.84\ g_{gel}/mL_{gel}) \times (0.52\ g_{diesel}/g_{gel}) \times (0.000026\ g_{benzene}/g_{diesel}) \times (3{,}785\ mL_{gel}/gal_{gel}) \times (1\ gal_{water}/3.785\ L_{water}) \times (1{,}000\ mL/L) \times (10^6\ \mu g/g) = 45\ \mu g/L$$

Table 4-2 summarizes the estimated injection concentrations of each BTEX constituent for the three assumed gel/water ratios and the minimum and maximum concentrations of BTEX in diesel fuel. It also presents the MCL for each compound. Many of the estimated concentrations of BTEX exceed the MCL at the point-of-injection.

Table 4-2 and the remainder of this section provide a qualitative assessment of the fate and transport processes that could attenuate the concentrations of BTEX in groundwater. Factors that would influence the availability of constituents of potential concern in fracturing fluids and decrease their concentrations include:

- Fluid Recovery - much of the fluid is eventually pumped back to the surface.
- Adsorption and entrapment - some of these constituents will undergo adsorption to the coal or become entrapped in the formation.

- Biodegradation - some fracturing fluid constituents, such as benzene, may undergo partial biodegradation.

4.3.2 Fracturing Fluid Recovery

Following the injection of fracturing fluids into the subsurface through coalbed methane wells (i.e., production wells), considerable amounts of the fracturing fluids are removed. During the recovery process, the injected fluids and ambient groundwater are pumped out of the formation through the production well to reduce formation pressure, enabling methane desorption and extraction. Palmer et al. (1991a) found that 61 percent of fracturing fluids were recovered based on samples collected from coalbed methane wells over a 19-day period. Their study predicted total recovery to be between 68 and 82 percent.

Palmer et al. (1991a) also discussed the possibility that a "check-valve effect" could trap some of the fracturing fluid on one side (i.e., upgradient, during production) of a collapsed or narrowed fracture, preventing the fluid from flowing back to the production well. This check-valve effect can occur in both natural and induced fractures when the fractures narrow again after the injection of fracturing fluid ceases, formation pressure decreases, and extraction of methane and groundwater begins.

Another factor preventing full recovery of injected fluids is the high injection pressure used during hydraulic fracturing operations. Fracturing fluids are forced into the subsurface under high pressure to enlarge and propagate existing fractures. The hydraulic gradients that cause fluids to flow away from the well during injection are much greater than the hydraulic gradients that occur during fluid recovery. As a result, some of the fracturing fluids will travel beyond the capture zone of the production well. The capture zone of a production well is the portion of the aquifer that contributes water to the well. The size of this zone will be affected by regional groundwater gradients, and by the drawdown caused by the well (USEPA, 1987). Fluids that flow beyond the capture zone of the production well generally are not recovered during the flowback process.

Gel contained in fracturing fluids may be unrecovered because its properties differ from that of water and highly soluble constituents of fracturing fluids. One mined-through study reviewed by EPA described evidence of gel clumps within many fractures (Steidl, 1993). One observed concentration of gel in a fracture was 15 times the injected concentration. When the fluids exist as undissolved gel, they may remain attached to the sides of the fractures or be trapped within smaller fractures or pores present in formations that surround the coalbed. The mined-through studies suggest that such fluids are unlikely to flow with groundwater during production, but they may present a source of gel constituents to flowing groundwater subsequent to fluid recovery. Fate and transport processes discussed later in this section can serve to reduce gel constituent concentrations that may result from trapped fluids. Mechanisms that may affect the recovery of fracturing fluids are discussed in section 3.3.2 of Chapter 3.

4.3.3 The Influence of the Capture Zone

The recovery process typically lasts approximately 10-20 years. During that time, groundwater within the production well's capture zone flows toward the production well. Assuming complete mixing, the predicted recovery of injected BTEX is between 68 and 82 percent (Palmer et al., 1991a). Thus, between 20 and 30 percent of the BTEX injected is expected to remain in the formation. It is reasonable to expect that most of the unrecovered fluid lies outside the capture zone and that the residual concentrations of BTEX within the capture zone are substantially less than the injection concentrations. Chemicals such as BTEX that are not recovered from within the capture zone during groundwater production may be diluted by groundwater that flows into the formation to replace production water. Additional attenuation from sorption and biodegradation may occur. Subsequent to production, dispersion and diffusion may serve to reduce residual BTEX concentrations. The fracturing fluids that flow beyond the capture zone are affected by regional groundwater flow and may be diluted by groundwater.

4.3.4 Fate and Transport Considerations

BTEX that has moved beyond the production well's capture zone is of the greatest concern. The fate and transport mechanisms that may affect BTEX concentrations beyond the capture zone are evaluated in this section. Factors that would likely decrease exposure concentrations and/or availability of BTEX include attenuation through groundwater flow dynamics, biological processes, and adsorption.

BTEX outside of the capture zone will likely be transported by groundwater flowing according to regional hydraulic gradients. This flow and transport are not influenced by production pumping. Nevertheless, mechanical dispersion will cause BTEX to spread horizontally and vertically in the aquifer, thereby reducing the concentrations. The degree of mechanical dispersion depends in part on the velocity of flow and increases with increased travel distance. Dilution can have a significant effect on the BTEX concentrations that could migrate to drinking water wells, especially if these wells are hundreds to thousands of feet from a hydraulically induced fracture. The process of molecular diffusion (i.e., the movement of BTEX from areas of higher to lower concentration due to the concentration differences) will further reduce BTEX concentrations. Collectively, mechanical dispersion and molecular diffusion are referred to as hydrodynamic dispersion (Fetter, 1994).

The biodegradation of diesel fuel constituents, including BTEX, has been studied in other geologic settings and laboratory studies and may lead to reductions in concentrations in coalbeds given the appropriate site conditions. No information was found about the occurrence of biodegradation or biodegradation rates of BTEX in coalbeds or surrounding rock. In order for biodegradation to occur, organisms capable of using BTEX as a food source must be present and conditions such as favorable pH, salinity, and sometimes the availability of oxygen, nitrogen, and phosphorous must be met to ensure bacterial survival. Generally, substantial benzene degradation occurs in aerobic environments. The levels of oxygen in a particular formation vary widely depending primarily on the depth of coalbeds from the surface. Data regarding biodegradation of benzene in an anaerobic environment indicates a range from no degradation to relatively slow degradation (USEPA, 1999).

As groundwater flows through a formation, chemicals such as BTEX may be retarded by adsorption. Although adsorption in coalbeds is likely, quantification of adsorption is difficult in the absence of laboratory or site-specific studies (due to competition for adsorption between BTEX and more lipophilic and less soluble constituents of diesel fuel and coal, and fracture thickness). Other processes, such as desorption of BTEX from the coal surface, and dissolution of BTEX from the gel phase may play a role in BTEX transport. Entrapment of gel in pore spaces and fractures may also influence the degree to which BTEX is available to groundwater. In some cases, the gel may be entrapped in such a way that it is neither available to flow back towards the production well nor flow towards a USDW in response to regional groundwater gradients.

According to the information listed on MSDSs provided to EPA, several of the constituents of potential concern listed in Table 4-1 can have toxic effects when people are exposed to sufficiently high concentrations through the susceptible route(s) of exposure (i.e., inhalation, ingestion, skin contact). However, only the BTEX compounds originating from diesel fuel are regulated under SDWA. None of the other constituents in Table 4-1 appear on the Agency's draft Contaminant Candidate List (CCL). The drinking water CCL is the primary source of priority contaminants for evaluation by EPA's drinking water program. Contaminants on the CCL are known or anticipated to occur in public water systems and may require regulations under SDWA. Information on the GSA study is available at http://www.epa.gov/fedrgstr/EPA-WATER/2004/April/Day-02/w7416.htm**.**

Further, EPA does not believe that the other Table 4-1 constituents potentially contained in fracturing fluids are introduced through coalbed methane fracturing in concentrations high enough to pose a significant threat to USDWs. First, it is EPA's understanding, based on conversations with field engineers and on witnessing three separate fracturing events, that fracturing fluids used for coalbed methane fracturing do not contain most of the constituents listed in Table 4-1. Second, if the Table 4-1 constituents were used, EPA believes some of the same hydrodynamic phenomena listed in steps 2 and 4 (flowback, dilution and dispersion), step 3 (adsorption and entrapment), and potentially step 5 (biodegradation) would minimize the possibility that chemicals included in the fracturing fluids would adversely affect USDWs.

4.4 Summary

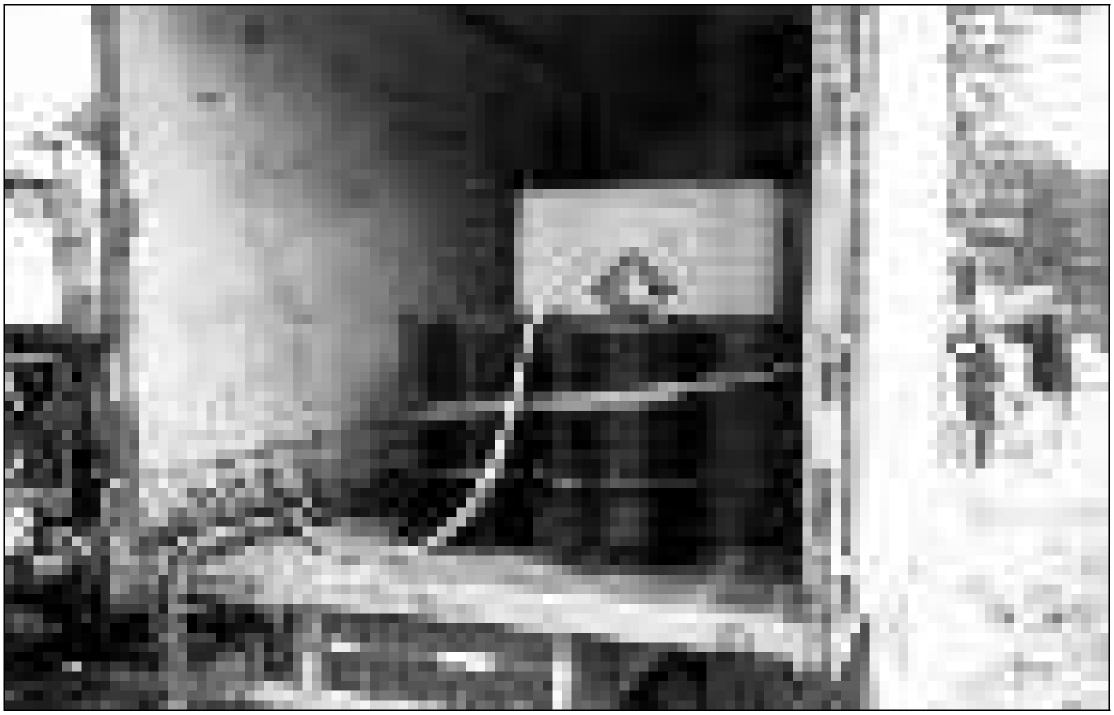
Fracture engineers select fracturing fluids based on site-specific characteristics including formation geology, field production characteristics, and economics. Hydraulic fracturing operations vary widely in the types of fracturing fluids used, the volumes of fluid required, and the pump rates at which they are injected. Based on the information EPA collected, water or nitrogen foam frequently constitutes the solute in fracturing fluids used for coalbed methane stimulation. Other components of fracturing fluids used to stimulate coalbed methane wells may contain only benign ingredients, but in some cases, they contain constituents such as diesel fuel that can be hazardous in their undiluted forms. Fracturing fluids are significantly diluted prior to injection.

Water with a simple sand proppant can be adequate to achieve a desired fracture at some sites. In some cases, water must be thickened to achieve higher proppant transport capabilities. Thickening can be achieved by using linear or cross-linked gelling agents. Cross-linkers are costly additives compared to simple linear gels, but a fluid's fracturing efficiency can be greatly improved using cross-linkers. Foam fracturing fluids can be used to considerably reduce the amount of injected fluid required. The reduced water volume requirement translates into a space and cost savings at the treatment site because fewer water tanks are needed. Foam fracturing fluids also promote rapid flowback and reduced volumes of flowback water requiring disposal.

The use of diesel fuel in fracturing fluids poses the greatest potential threat to USDWs because the BTEX constituents in diesel fuel exceed the MCL at the point-of-injection. Given the concerns with the use of diesel fuel, EPA recently entered into agreements with three major service companies to eliminate diesel fuel from hydraulic fracturing fluids injected directly into USDWs to stimulate coalbed methane production. Industry representatives estimate that these three companies perform approximately 95 percent of the hydraulic fracturing projects in the United States.

In situations when diesel fuel is used in fracturing fluids, a number of factors would decrease the concentration and/or availability of BTEX. These factors include fluid recovery during flowback, adsorption, dilution and dispersion, and potentially biodegradation of constituents. For example, Palmer et al. (1991a) documented that only about one-third of fracturing fluid that is injected is expected to remain in the formation. EPA expects fate and transport considerations would minimize the possibility that chemicals included in fracturing fluids would adversely affect USDWs.

Figures 4-1 and 4-2. Liquid nitrogen tanker trucks transport gas to the site for nitrogen foam fracturing. Nitrogen will travel through pipes to be mixed with water and a foaming agent at the wellhead prior to injection. The foam is used to create and propagate the fracture deep within the targeted coal seam.



Figures 4-3 and 4-4.
Chemicals are stored on site in a support truck. Fracturing fluid additives such as the foaming agent can be pumped directly from storage containers to mix tanks.

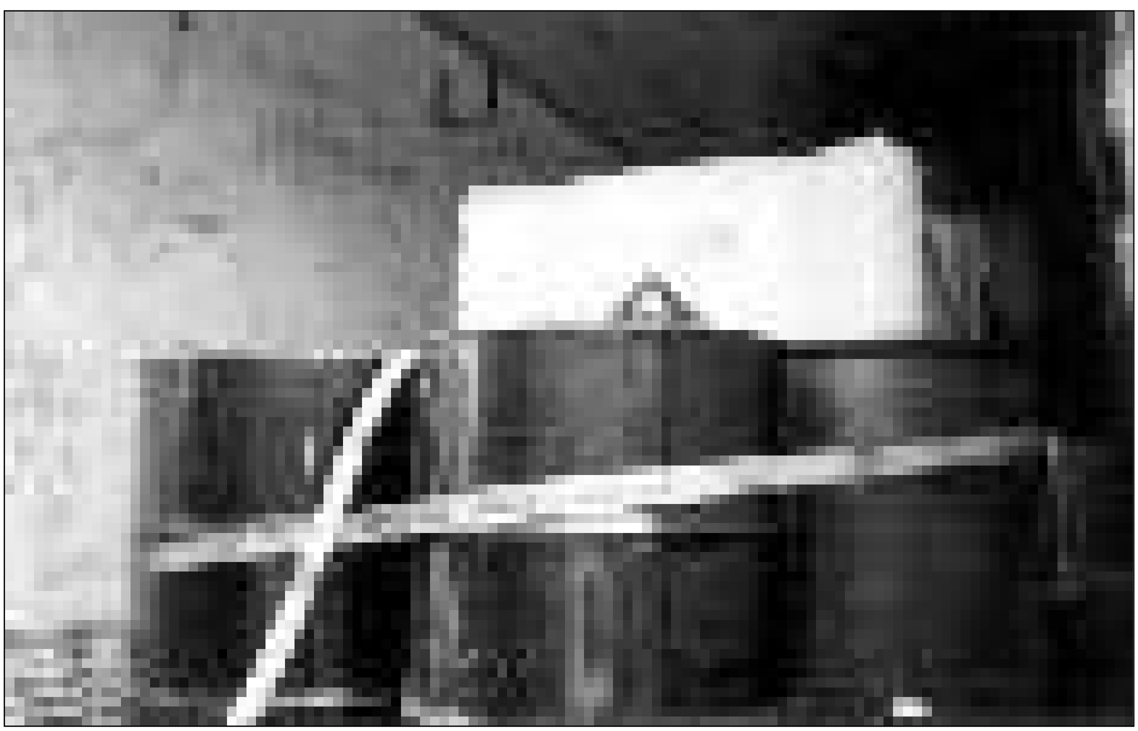
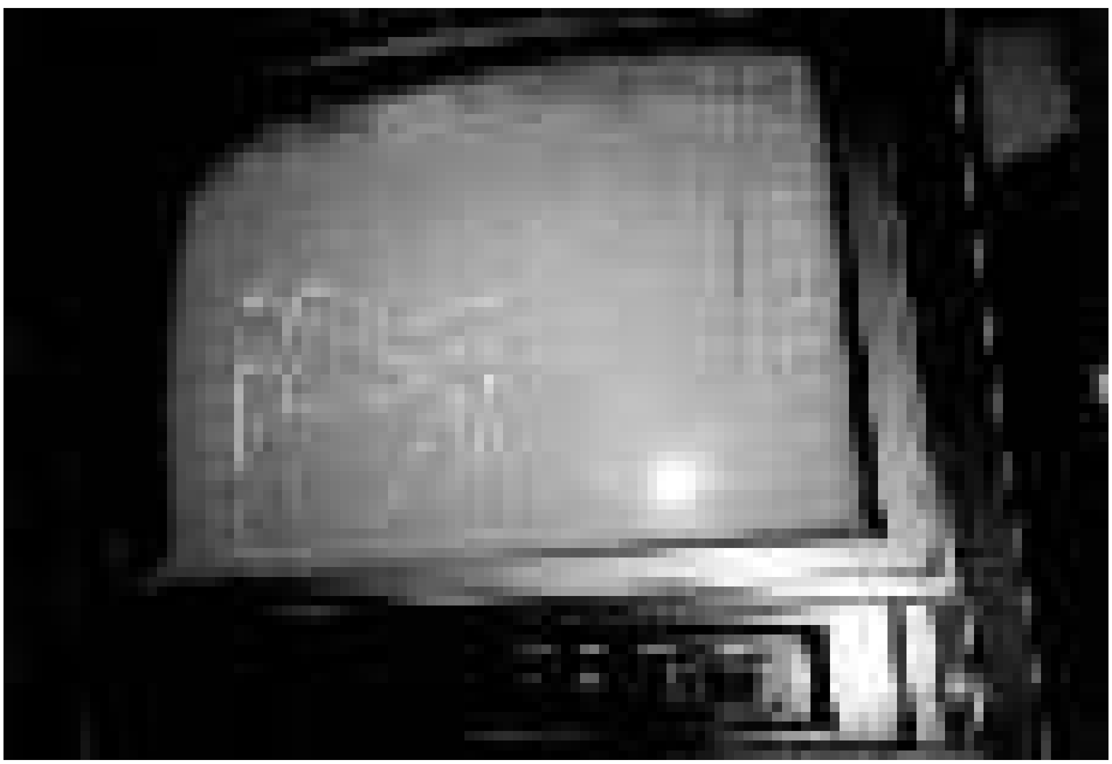


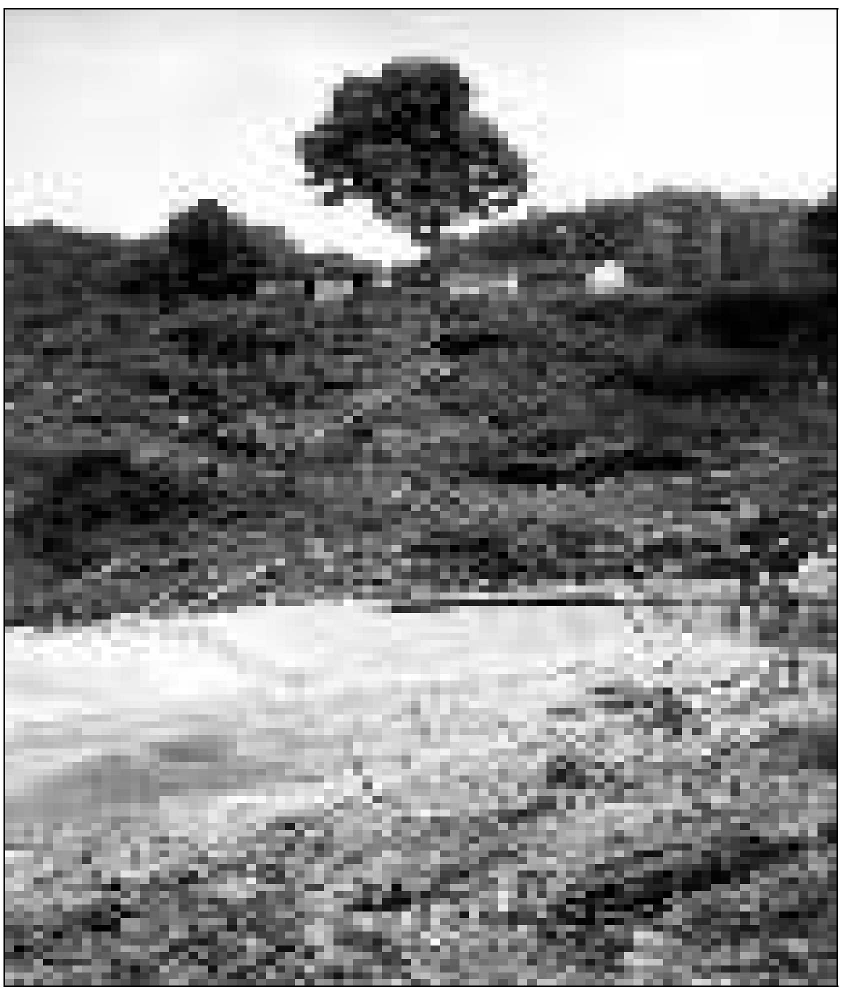
Figure 4-5.
The fracturing fluid (water with additives) is stored on site in large, upright storage tanks. Each tank contains mix water imported from off-site, or formation water extracted directly from the gas well.

Figure 4-6.
Gelled water is pre-mixed in a truck-mounted mixing tank. Photograph shows a batch of linear, guar-based gel. This gel is used to transport the sand proppant into the fracture propagated by the nitrogen foam treatment.

Figure 4-7.
The fracturing fluids, additives, and proppant are pumped to the wellhead and mixed just prior to injection. The flow rate of each injected component is monitored carefully from an on-site control center.

Figures 4-8 and 4-9.
Electronic monitoring systems provide constant feedback to the service company's operators. Fluid flow rates and pressure buildup within the formation are monitored to ensure that fracture growth is safe and controlled.





Figures 4-10 and 4-11. Fluid that is extracted from the well is sprayed through a diffuser and stored in a lined trench until it is disposed of off-site or discharged.

Chapter 5
Summary of Coalbed Methane Basin Descriptions

As part of the Phase I study EPA conducted an extensive literature review to collect information regarding the major coal basins in the United States. Eleven major coal basins were identified in the United States and are shown in Figure 5-1 at the end of the chapter (those basins shaded in red have the highest coalbed methane production volumes). The goals of this review were to assess the following for each of the 11 major coal basins:

- The physical relationship between the coalbeds and the USDWs.
- Whether hydraulic fracturing is or has been used to stimulate coalbed methane wells in production basins.
- The types of fluids used to create the fractures.
- If possible, whether the potential for contaminants to enter a USDW exists.

This information is necessary to evaluate whether hydraulic fracturing is practiced within a basin and the types of fluids used in the fracturing process. More importantly, this information establishes whether the coal formations lie within a USDW, creating the potential for hydraulic fracturing fluid injection to threaten USDWs. A USDW is not necessarily currently used for drinking water and may contain groundwater unsuitable for drinking without treatment. In some cases, very little information was uncovered by EPA regarding certain topics for some of the basins.

Each of the 11 major basins is described in this chapter and in Table 5-1 (immediately following section 5.12 of this chapter). In addition, a more comprehensive description of the geology, hydrology, and coalbed methane production activity for each basin is provided in Attachments 1 through 11 of this report.

5.1 The San Juan Basin

The San Juan Basin covers an area of about 7,500 square miles straddling the Colorado-New Mexico state line in the Four Corners region (Figure 5-1). It measures roughly 100 miles long north to south and 90 miles wide. The Continental Divide trends north to south along the east side of the basin.

The major coal-bearing unit in the San Juan Basin is known as the Fruitland Formation. Coalbed methane production occurs primarily in coals of the Fruitland Formation, but some coalbed methane is trapped in the underlying and adjacent Pictured Cliffs sandstone. Many wells are completed in both zones. The coals of the Fruitland Formation are very thick compared to coalbeds in eastern basins: the thickest coals range from 20 to over 40 feet. Total net thickness of all coalbeds ranges from 20 to over 80 feet throughout the San Juan Basin, compared to 5 to 15 feet in eastern basins.

Coalbed methane wells in the San Juan Basin range from 550 to 4,000 feet in depth, and about 2,550 wells were operating in 2001 (Colorado Oil and Gas Conservation Commission and New Mexico Oil Conservation Division, 2001). The San Juan Basin is the most productive coalbed methane basin in North America. In 1996, coalbed methane production there averaged about 800 thousand cubic feet per day per well and totaled over 800 billion cubic feet (Bcf) for that year (Stevens et al., 1996). This total rose to 925 Bcf in 2000 (GTI, 2002)).

The majority of coalbed methane development and hydraulic fracturing in the northern portion of the San Juan Basin takes place within a USDW. The waters in parts of the Fruitland Formation usually contain less than 10,000 mg/L TDS, which is the water quality criterion for a USDW. In the northern half of the formation, most waters contain less than 3,000 mg/L, and wells near the outcrop produce water that contains less than 500 mg/L TDS.

Fracturing fluids used in the San Juan Basin include hydrochloric acid; slick water (water mixed with solvent); linear and crosslinked gels; and, since 1992, nitrogen- or carbon dioxide-based foams (Harper et al., 1985; Jeu et al., 1988; Holditch et al., 1988; Palmer et al., 1993b; Choate et al., 1993). Data are not readily available concerning fracture growth and height within the Fruitland Formation.

5.2 The Black Warrior Basin

The Black Warrior Basin is the southernmost of the three basins that compose the Appalachian Coal Region of the eastern United States. The basin covers about 23,000 square miles in Alabama and Mississippi. It is approximately 230 miles long from west to east and approximately 188 miles wide from north to south (Figure 5-1). Basin coalbed methane production is limited to the bituminous coalfields of west-central Alabama, primarily in Jefferson and Tuscaloosa Counties.

Coalbed methane production in the Black Warrior Basin is confined to the Pennsylvanian-aged Pottsville Formation. The ancient coastline of prehistoric Alabama was characterized by 8 to 10 “coal-deposition cycles” of rising and falling sea levels. Each cycle features mudstone at the base of the cycle (deeper water) and coalbeds at the top (emergence). Most coalbed methane wells tap the Black Creek/Mary Lee/Pratt cycles and range from 350 to 2,500 feet deep (Holditch, 1990).

Coalbed methane production in the Black Warrior Basin is among the highest in the United States. In 1996, about 5,000 coalbed methane wells were permitted in Alabama. In 2000, this number increased to over 5,800 wells (Alabama Oil and Gas Board, 2002). Coalbed methane wells have production rates that range from less than 20 to more than 1 million cubic feet (Mcf) per day per well (Alabama Oil and Gas Board, 2002). Between 1980 and 2000, coalbed methane wells in Alabama produced roughly 1.2 trillion cubic feet (Tcf) of gas. According to GTI, annual gas production was 112 Bcf in 2000 (GTI, 2002).

Some portions of the Pottsville Formation contain waters that meet the quality criterion of less than 10,000 mg/L TDS for a USDW. According to the Alabama Oil and Gas Board, some waters in the Pottsville Formation do not meet the definition of a USDW and have TDS levels considerably higher than 10,000 mg/L.

Early literature indicates that most of the wells in production in the early 1990s have been hydraulically fractured an average of two to six times to achieve acceptable production rates (Holditch et al., 1988; Holditch, 1990; Palmer et al., 1993a and 1993b).

5.3 The Piceance Basin

The Piceance Coal Basin is entirely within the northwest corner of the Colorado (Figure 5-1). The coalbed methane reservoirs are found in the Upper Cretaceous Mesaverde Group, which covers about 7,225 square miles of the basin.

The Mesaverde Group ranges in thickness from about 2,000 feet on the west to about 6,500 feet on the east side of the basin (Johnson, 1989). The depth to the methane-bearing Cameo-Wheeler-Fairfield coal zone is about 6,000 feet. Two-thirds of the coalbed methane occurs in coals deeper than 5,000 feet, and the Piceance Basin is one of the deepest coalbed methane areas in the United States (Quarterly Review, August 1993).

The depth of the coals in the Piceance Basin inhibits permeability, making it difficult to extract the coalbed methane. This, in turn, has slowed coalbed methane development in the basin. However, it is estimated that 80 trillion to 136 Tcf of coalbed methane are contained in the Cameo-Wheeler-Fairfield coal zone of the basin (Tyler et al., 1998). Total coalbed methane production was 1.2 Bcf in 2000 (GTI, 2002).

The Piceance Basin contains both alluvial and bedrock aquifers. Unconsolidated alluvial aquifers (narrow and thin deposits of sand and gravel formed primarily along stream courses) are the most productive aquifers in the Piceance Basin. The bedrock aquifers are known as the upper and lower Piceance Basin aquifer systems. The upper aquifer system is about 700 feet thick, and the lower aquifer system is about 900 feet thick. Water at depth in the Piceance Basin appears to be of poor quality, minimizing its chance of being designated a USDW. In general, the potable water wells in the Piceance Basin extend no further than 200 feet in depth, based on well records maintained by the Colorado Division of Water Resources. A composite water quality sample taken from 4,637 to 5,430 feet deep in the Cameo coal zone exhibited a TDS level of 15,500 mg/L (Graham, 2001).

Hydraulic fracturing is practiced in this basin. A variety of fluids are used for fracturing, including water with sand proppant and gelled water and sand. In some cases, hydraulic stimulations created multiple short (100-foot), fractures around the wells (Quarterly Review, August 1993).

It is unlikely that any USDWs and coals targeted for methane production (generally currently located at great depth, such as 4,000 feet below the ground surface and deeper) would coincide in this basin. The thousands of feet of stratigraphic separation between the coal gas bearing Cameo Zone and the lower aquifer system in the Green River Formation should prevent any of the effects from the hydrofracturing of gas-bearing strata from reaching either the upper or the lower bedrock aquifers.

Research suggests that exploration may target areas where groundwater circulation may enhance gas accumulation in the coal and associated sandstones (Tyler et al., 1998). Under these exploration and development conditions, a USDW located in shallower Cretaceous rocks near the margins of the basin could be affected by hydraulic fracturing. The depth of methane-bearing coals (about 6,000 feet) seems to indicate that, in the Piceance Basin, the chances of contaminating any overlying, shallower USDWs (no deeper than about 1,000 feet) from injection of hydraulic fracturing fluids and subsequent subsurface fluid transport are minimal. The coalbed methane producing Cameo Zone and the deepest known aquifer, the lower bedrock aquifer, have a stratigraphic separation of over 6,000 feet.

5.4 The Uinta Basin

The Uinta Coal Basin is mostly within eastern Utah; a very small portion of the basin is in northwestern Colorado (Figure 5-1). The basin covers approximately 14,450 square miles (Quarterly Review, August 1993). The Uinta Basin is stratigraphically continuous with the Piceance Basin of Colorado, but is structurally separated from it by the Douglas Creek Arch, an uplift near the Utah – Colorado state line.

Coal seams occur in the Cretaceous Mancos Shale and the Upper Cretaceous Mesaverde Group (Quarterly Review, 1993). Two major formations targeted for coalbed methane exploration are the Mancos Shale's Ferron Sandstone Member, which include the coals most targeted (approximately 90 percent of the time) for exploration (Petzet, 1996) and the Mesaverde Group's Blackhawk Formation, which contains about 14 coal zones (Petzet, 1996). The Ferron Coals are interbedded with sandstone and form a wedge of clastic sediment 150 to 750 feet thick. Depths to coal in the Ferron Sandstone range from 1,000 to over 7,000 feet below ground surface (Garrison et al., 1997). The Blackhawk Formation consists of coal seams interbedded with sandstone and a combination of shale and siltstone. Coals tapped in the Blackhawk Formation are 4,200 to 4,400 feet below the surface (Gloyn and Sommer, 1993).

Full-scale exploration in the Uinta Basin began in the 1990s (Quarterly Review, 1993). The database covering the Uinta Basin indicates that there are about 1,255 coalbed methane wells in production in the basin (Osborne, 2002). The coalbed methane potential of the Uinta Basin, revised by the Utah Geological Survey in the early 1990s, has been estimated to be between 8 trillion and more than 10 Tcf (Gloyn and Sommer, 1993).

At some locations, the groundwater in the Ferron Coals and Blackhawk Formations would not qualify as USDWs. According to the Utah Department of Natural Resources (DNR), Division of Oil, Gas and Mining, the water there varies greatly by location, each location having some TDS levels below and some above 10,000 mg/L (Utah DNR, 2002). In general, the quality of Blackhawk water is higher than Ferron water. For example, the most recent UIC application noted the combined quality of input water to be approximately 31,000 mg/L TDS for the Drunkards Wash Field (Ferron) and 9,286 mg/L TDS for the Castlegate Field (Blackhawk).

Fracturing fluid use is documented in the literature pertaining to the Uinta Basin. One company reported performing hydraulic fracturing stimulations using cross-linked borate gel with 250,000 pounds of proppant (Quarterly Review, 1993). Others report that they fractured wells with low-residue gel fracturing fluids and foams (Quarterly Review, 1993). GTI places the annual coalbed methane production in the Uinta Basin at 75.7 Bcf in 2000 (GTI, 2002).

The Blackhawk Formation is underlain by 300 feet of shale and sandstone, which separate it from the Castlegate Sandstone aquifer. It is underlain by similar geologic strata, which separate it from the Star Point Sandstone. Only in highly faulted areas is there a reasonable possibility that hydraulic fracturing fluids could migrate down to the Star Point Sandstone.

5.5 The Powder River Basin

The Powder River Basin is in northeastern Wyoming and southern Montana (Figure 5-1). The basin covers approximately 25,800 square miles (Larsen, 1989), approximately 75 percent of which is in Wyoming. Fifty percent of the Powder River Basin is believed to have the potential for coalbed methane production (Powder River Coalbed Methane Information Council, 2000). Annual production volume was estimated at 147 Bcf in 2000 (GTI, 2002). In 2002, wells in the Powder River Basin produced about 823 Mcf per day of coalbed methane (DOE, 2002).

Coalbeds in this region are interspersed at varying depths with sandstones, mudstone, conglomerate, limestone, and shale. The majority of the potentially productive coal zones range from about 450 feet to over 6,500 feet below ground surface (Montgomery, 1999). The uppermost formation is the Wasatch Formation, extending from land surface to 1,000 feet deep. Most coal seams in the Wasatch Formation are continuous and thin (6 feet or less). The Fort Union Formation lies directly below the Wasatch Formation and can be as much as 6,200 feet thick (Law et al., 1991). The coalbeds in this formation are typically most abundant in the upper portion, called the Tongue River member. This member is typically 1,500 to 1,800 feet thick, of which up to a composite total of 350 feet of coal can be found in various beds. The thickest of the individual coalbeds is over 200 feet (Flores and Bader, 1999). Recent estimates of coalbed methane reserves in the Powder River Basin range from 7 trillion to 40 Tcf (Montgomery, 1999; PRCMIC, 2000).

The Fort Union Formation that supplies municipal water to the City of Gillette is the same formation that contains the coals that are developed for coalbed methane. The coalbeds contain and transmit more water than the sandstones. The sandstones and coalbeds have been used for the production of both water and coalbed methane. The water produced from the coalbeds meets the quality criterion for USDWs of less than 10,000 mg/L TDS.

EPA's understanding is that hydraulic fracturing currently is not widely used in this region due to concerns about the potential for increased groundwater flow into the coalbed methane production wells (due to fracturing of impermeable formations adjacent to the coal, and the creation of a hydraulic connection to adjacent aquifers) and the collapse of open hole wells in coal upon dewatering. According to the available literature, where hydraulic fracturing has been used in this basin, it has not been an effective method for extracting methane. Hydraulic fracturing has been done primarily with water, or gelled water and sand, although recorded use of a solution of potassium chloride was identified in the literature.

5.6 The Central Appalachian Basin

The Central Appalachian Coal Basin is the middle of three basins that compose the Appalachian Coal Region of the eastern United States. It includes parts of Kentucky, Tennessee, Virginia, and West Virginia (Figure 5-1) and covers approximately 23,000 square miles. The greatest potential for methane development is in a small, 3,000-square-mile area in southwest Virginia and south central West Virginia (Kelafant, et al., 1988).

The coal basin consists of six Pennsylvanian age coal seams (Zebrowitz et al., 1991, and Zuber, 1998). These coal seams typically occur as multiple coalbeds or seams that are widely distributed (Zuber, 1998). The coal seams, from oldest to youngest (West Virginia/Virginia name), are the Pocahontas No. 3, Pocahontas No. 4, Fire Creek/Lower Horsepen, Beckley/War Creek, Sewell/Lower Seaboard, and Iager/Jawbone (Kelafant et al., 1988). The Pocahontas coal seams include the Squire Jim and Nos. 1 to 7; Nos. 3 and 4 are the thickest and cover the most area. Most of the coalbed methane (2.7 Tcf) occurs in the Pocahontas seams (Kelafant et al., 1988). In southwest Virginia and south central West Virginia, target coal seams achieve their greatest thickness and occur at depths of about 1,000 to 2,000 feet (Kelafant et al., 1988).

The Nora Field in southwestern Virginia is one of the better-known coalbed methane production fields. According to the Virginia Division of Gas and Oil, over 700 coalbed methane wells were drilled in the Nora Field in 2002 (Virginia Division of Gas and Oil, 2002). The Virginia Division of Gas and Oil also indicated that, in 2002, more than 1,800 coalbed methane wells were drilled in southwestern Virginia's Buchanan County (VA Division of Gas and Oil, 2002.) GTI reported that the entire basin produced 52.9 Bcf of gas in 2000 (GTI, 2002).

Because most of the coal strata dip, a coalbed methane well's location in the basin may determine if hydraulic fracturing during the well's development will affect the water quality of surrounding USDW. For instance, on the northeastern side of the basin, the depth to the Pocahontas No. 3 coalbed is less than 500 feet. This depth gradually increases to over 2,000 feet farther westward across this portion of the basin, in the direction of the dip of the coal seam. Therefore, a well tapping this seam in the eastern portion of the basin may be within a USDW, but a well tapping the seam in the western portion of the basin may be below the base of a

USDW. In addition, the base of the freshwater is not flat, but rather undulating. These factors indicate that the relationship between a coalbed and a USDW must be determined on a site-specific basis.

Hydraulic fracturing is a common practice in this region. Foam and water are the fracturing fluids of choice, and sand serves as the proppant. Additives can include hydrochloric acid, scale inhibitors, and microbicides. Pocahontas Oil & Gas, a subsidiary of Consol Energy, Inc., invited EPA personnel to a well where a hydraulic fracturing treatment was being performed by Halliburton Energy Services, Inc. Halliburton staff said that typical fractures extend from 300 to 600 feet from the well bore in either direction, but that fractures have been known to extend from as few as 150 feet to as many as 1,500 feet in length (Halliburton Inc., Virginia Site Visit, 2001). According to the fracturing engineer on-site, fracture widths range from one-eighth of an inch to almost one and one-half inches (Halliburton, Inc., Virginia Site Visit, 2001).

Since some coalbed methane exploration has moved to shallower seams, the Commonwealth of Virginia has instituted a voluntary program concerning depths at which hydraulic fracturing may be performed (Virginia Division of Oil and Gas, 2002). The program involves an operator's determination of the elevation of the lowest topographic point and the elevation of the deepest water well within a 1,500-foot radius of any proposed extraction well (Wilson, 2001). Hydraulic fracturing should occur at least 500 feet beneath than the lower of these two points.

5.7 The Northern Appalachian Basin

The Northern Appalachian Coal Basin is the northernmost of the three basins that make up the Appalachian Coal Region of the eastern United States. It includes parts of Pennsylvania, West Virginia, Ohio, Kentucky, and Maryland (Figure 5-1). The basin lies completely within the Appalachian Plateau geomorphic province and covers approximately 43,700 square miles (Adams et al., 1984, as cited by Pennsylvania Department of Conservation and Natural Resources, 2002). The Northern Appalachian basin trends northeast to southwest. The Rome Trough, a major graben structure, forms the southeastern and southern structural boundaries. The basin is bounded on the northeast, north, and west by outcropping Pennsylvanian-aged sediments (Kelafant et al., 1988).

The six Pennsylvanian-aged coal zones composing the Northern Appalachian Coal Basin are the Brookville-Clarion, Kittanning, Freeport, Pittsburgh, Sewickley, and Waynesburg. These coal units are within the Pottsville, Allegheny, and the Monongahela Groups (Kelafant et al., 1988). Coal seam depths range from surface outcrops to as much as 2,000 feet below ground surface, with most coal occurring at depths shallower than 1,000 feet (Quarterly Review, 1993). These depth differences arise due to the dip of the coalbeds. The total thickness of the Pennsylvanian-aged coal system averages 25 feet; however, better developed seams within the coal zones can increase in thickness by up to twice the average (Quarterly Review, 1993).

Coalbed methane has been produced in commercial quantities from the Pittsburgh coalbed of the Northern Appalachian Coal Basin since 1932 (Lyons, 1997), after the discovery of the Big Run

Field in Wetzel County, West Virginia, in 1905 (Hunt and Steele, 1991). As of 1993, O'Brien Methane Production, Inc. had at least 20 wells in Pennsylvania's southern Indiana County (Quarterly Review, 1993). Coalbed methane production development in the Northern Appalachian Basin has lagged, however, due to insufficient reservoir knowledge, inadequate well-completion techniques, and coalbed methane ownership issues revolving around whether the gas is owned by the mineral owner or the oil and gas owner (Zebrowitz et al., 1991). Discharge of produced waters has also proven to be problematic (Lyons, 1997) for coalbed methane field operators in the Northern Appalachian Coal Basin. Total coalbed methane production stood at 1.41 Bcf in 2000 (GTI, 2002). As of October 2002, 185 coalbed methane wells were producing coalbed methane in Pennsylvania (Pennsylvania Department of Conservation and Natural Resources, 2002).

The Northern Appalachian Basin is situated in the Appalachian Plateau's physiographic province. The primary aquifer in this area is a Pennsylvanian sandstone aquifer underlain by limestone aquifers (USGS, 1984). Water quality data from eight historic Northern Appalachian Coal Basin projects show that estimated TDS levels ranged from 2,000 to 5,000 mg/L at depths of 500 to 1,025 feet below ground surface (Zebrowitz et al., 1991), well within EPA's water quality criterion of 10,000 mg/L TDS for a USDW (40 CFR §144.3). Depths to the bottoms of the USDWs vary greatly in the basin and are better determined on a site-specific basis.

Hydraulic fracturing fluids used in the Northern Appalachian Basin have included water and sand, and nitrogen foam and sand (Hunt and Steele, 1991). The Christopher Coal Company/Spindler Wells Project, which took place from 1952 to 1959, stimulated 1 well with 12 quarts of nitroglycerin (Hunt and Steele, 1991). In the Vesta Mines Project of Washington County, Pennsylvania, the United States Bureau of Mines used gelled water and sand to complete 5 wells in the Pittsburgh Seam (Hunt and Steele, 1991).

Because most of the coal strata dip, a well's location in a basin determines whether the well is coincident with a USDW. For example, in the Pittsburgh Coal Group in Pennsylvania, the depth to the top of the coal group varies from outcrop to about 1,200 feet in the very southwestern corner of the state (Kelafant et al., 1988). The approximate depth to the bottom of the USDW is 450 feet. Therefore, production wells operating down to approximately 450 feet could potentially be hydraulically connected to the USDW.

5.8 The Western Interior Coal Region

The Western Interior Coal Region comprises three coal basins, the Arkoma, the Cherokee, and the Forest City Basins, and encompasses portions of six states: Arkansas, Oklahoma, Kansas, Missouri, Nebraska, and Iowa (Figure 5-1). The Arkoma Basin covers about 13,500 square miles in Arkansas and Oklahoma. The Cherokee Basin is part of the Cherokee Platform Province, which covers approximately 26,500 square miles (Charpentier, 1995) in Oklahoma, Kansas, and Missouri. The Forest City Basin covers about 47,000 square miles (Quarterly Review, 1993) in Iowa, Kansas, Missouri, and Nebraska.

In the Arkoma Basin, major middle-Pennsylvanian coalbeds occur within the Hartshorne, McAlester, Savanna, and Boggy Formations (Quarterly Review, 1993). The Hartshorne coals of the Hartshorne Formation are the most important for methane production in the Arkoma Basin. Their depth ranges from 600 to 2,300 feet in two productive areas of southeastern Oklahoma (Quarterly Review, 1993). In the Cherokee Basin, the primary coal seams targeted by operators are the Riverton Coal of the Krebs Formation and the Weir-Pittsburg and Mulky coals of the Cabaniss Formation (Quarterly Review, 1993). The Riverton and Weir-Pittsburg seams are about 3 to 5 feet thick and range from 800 to 1,200 feet deep, while the Mulky Coal, which ranges up to 2 feet thick, occurs at depths of 600 to 1,000 feet (Quarterly Review, 1993). Individual coal seams in the Cherokee Group of the Forest City Basin range from a few inches to about 4 feet thick, with seams up to 6 feet thick (Brady, 2002; Smith, 2002). Depths to the top of the Cherokee Group coals range from approximately the surface to 230 feet below ground surface in the shallower portion of the basin, in southeastern Iowa, to about 1,220 feet in the deeper part of the basin, in northeastern Kansas (Bostic et al., 1993).

As of March 2000, there were 377 coalbed methane wells in the Arkoma Basin of eastern Oklahoma, ranging in depth from 589 to 3,726 feet (Oklahoma Geological Survey, 2001). The Arkoma Basin contains an estimated 1.58 to 3.55 Tcf of gas reserves, primarily in the Hartshorne coals (Quarterly Review, 1993). In the Cherokee Basin, unknown amounts of coalbed methane gas have been produced with conventional natural gas for over 50 years (Quarterly Review, 1993). Targeted coalbed methane production increased in the late 1980s, and at least 232 coalbed methane wells had been completed as of January 1993 (Quarterly Review, 1993). The Cherokee Basin contains an estimated 1.38 Mcf of gas per square mile (Stoeckinger, 1989) in the targeted Mulky, Weir-Pittsburg, and Riverton coal seams of the Cherokee Group (Quarterly Review, 1993). In total, the basin contains approximately 36.6 Bcf of gas. However, the Petroleum Technology Transfer Council (1999) indicates that there are nearly 10 Tcf of gas in eastern Kansas alone (PTTC, 1999). The Forest City Basin was relatively unexplored in 1993, with about 10 coalbed wells concentrated in Kansas' Atchison, Jefferson, Miami, Leavenworth, and Franklin Counties (Quarterly Review, 1993). The Forest City Basin contains an estimated 1 Tcf of gas (Nelson, 1999). For the entire region, coalbed methane production was 6.5 Bcf in 2000 (GTI, 2002).

According to the National Water Summary (1984), there are no principal aquifers in the portions of Oklahoma and Arkansas in the Arkoma Basin, only small alluvial aquifers bounding rivers. Water quality test results from the targeted Hartshorne seam in Oklahoma have shown the water to be highly saline (Quarterly Review, 1993). The base of fresh water in Arkansas is about 500 to 2,000 feet below ground surface (Cordova, 1963). However, Cordova (1963) does not define "fresh water." While the majority of the Cherokee Basin does not contain a principal aquifer, the Ozark and Douglas aquifers are contained within the basin (National Water Summary, 1984). The confined Ozark Aquifer, composed of weathered and sandy dolomites, typically contains water wells that extend from 500 to 1,800 feet in depth (National Water Summary, 1984). The usually unconfined Douglas Aquifer is a sandstone channel of the Pennsylvanian Age (National Water Summary, 1984). Wells are usually 5 to 400 feet deep in this aquifer. In Kansas, depth to the base of the Ozark Aquifer is roughly 1,750 feet below ground surface (Ozark Aquifer Base Map, 2001). In Oklahoma, the Cherokee Basin also contains the Garber-Wellington and

Vamoosa-Ada aquifers (National Water Summary, 1984). Water well depths in these two aquifers usually range from 100 to 900 feet (National Water Summary, 1984). The Forest City Basin contains the Jordan Aquifer, the Dakota Aquifer, and glacial drift, alluvial, and Paleozoic-aged rock aquifers. Wells in these aquifers commonly range in depth from 300 to 2,000 feet, 100 to 600 feet, 10 to 300 feet, 10 to 150 feet, and 30 to 2,200 feet, respectively (National Water Summary, 1984). Throughout the Western Interior Coal Region, water quality sampling has shown TDS levels to range from 500 to 40,000 mg/L (Missouri Division of Geological Survey and Water Resources, 1967).

Hydraulic fracturing is common in the Western Interior Coal Basin. Fracturing fluids such as linear gel, acid, and nitrogen foam were used extensively in the Western Interior coal region before 1992, and slick water treatments became common in 1993. Hydraulic fracturing is still practiced in the basin.

Based on depths to the Hartshorne Coal (0 to 4,500 feet in Arkansas) and the base of fresh water (500 to 2,000 feet in Arkansas), it appears that coalbed methane extraction wells in the Arkoma Basin could be coincident with potential USDWs in Arkansas (Andrews et al., 1998; Cordova, 1963). Based on maps provided by the Oklahoma Corporation Commission (2001) showing the depths of the 10,000 mg/L TDS groundwater quality boundary in Oklahoma, coalbed methane wells and USDWs would most likely not coincide in Oklahoma. This is based on depths to coals typically greater than 1,000 feet (Andrews et al., 1998) and depths to the base of the USDW typically shallower than 900 feet (OCC Depth to Base of Treatable Water Map Series, 2001).

In the Cherokee Basin, coalbed methane wells targeting the Cherokee Group coals in Kansas coincide with USDWs. Depths to the top of coalbeds range from 800 to 1,200 feet (Quarterly Review, 1993) while the depth to the base of fresh water is estimated at 1,750 feet (Mapped information from the Kansas Data Access and Support Center (DASC), 2001a). More information concerning water quality is required prior to any determination of coalbed methane well/USDW co-location in Missouri. However, current levels of coalbed methane activity are minimal in that state. In addition, since only a very small portion of the Cherokee Basin falls within Missouri, this portion of the basin needs to be delineated more precisely to see which USDWs are in this small part of the basin. Last, in the Forest City Basin, there appears to be little relationship between water supplies and coalbeds that may be used for coalbed methane extraction. However, aquifer and well information from the National Water Summary (1984) indicates that a co-location of the two could exist in Nebraska. More information is needed to define the relationship between coalbeds and USDWs in the Forest City Basin.

5.9 The Raton Basin

The Raton Basin covers about 2,200 square miles in southeastern Colorado and northeastern New Mexico (Figure 5-1). It is the southernmost of several major coal-bearing basins along the eastern margin of the Rocky Mountains. The basin extends 80 miles north to south and as much as 50 miles east to west (Stevens et al., 1992). It is an elongate, asymmetric syncline, 20,000 to 25,000 feet thick in the deepest part.

There are two major coal formations in the Raton Basin, the Vermejo and the Raton. The Vermejo coals range from 5 to 35 feet thick, while the Raton coal layers range from 10 to more than 140 feet thick. Although the Raton Formation is much thicker and contains more coal than the Vermejo Formation, individual coal seams in the Raton are less continuous and generally thinner.

Methane resources for the basin have been estimated at approximately 10.2 Tcf in the Vermejo and Raton Formations (Stevens et al., 1992). As of 1992, about 114 coalbed methane exploration wells had been drilled in the basin (Quarterly Review, 1993). According to GTI, the average coalbed methane production rate of wells in the Raton Basin was close to 300 thousand cubic feet per day, and annual production in 2000 was 30.8 Bcf (GTI, 2002).

The coal seams of the Vermejo and Raton Formations developed for methane production also contain water that meets the criterion for a USDW. The underlying Trinidad Sandstone and other sandstone beds in the Vermejo and Raton Formations, as well as intrusive dikes and sills, also contain water of sufficient quality to be used as drinking water.

Coalbed methane well stimulation using hydraulic fracturing techniques is common in the Raton Basin. Records show that fracturing fluids used are typically gels and water with sand proppants. Hemborg (1998) showed that in most cases water yield decreased dramatically as methane production continued over time. However, some wells exhibited increased water production as methane production continued or increased. Two causal factors were suggested (Hemborg, 1998) for the rise in water production:

1. Well stimulation had increased the well's zone of capture to include adjacent water-bearing sills or sandstones that were hydraulically connected to recharge areas, or;

2. Well stimulation had created a connection between the coal seams and the underlying water-bearing Trinidad Sandstone.

5.10 The Sand Wash Basin

The Sand Wash Basin is in northwestern Colorado and southwestern Wyoming. It is part of the Greater Green River Coal Region, which includes the Washakie Basin, the Great Divide (Red Desert) Basin, and the Green River Basin (Figure 5-1). These sub-basins are separated by uplifts caused by deformation of the basement rock. For example, the Sand Wash Basin is separated from the adjacent Washakie Basin by the Cherokee Arch, an anticline ridge that runs east to west along the Colorado – Wyoming border. The Greater Green River Coal Region, in total, covers an area of approximately 21,000 square miles. The Sand Wash Basin covers approximately 5,600 square miles, primarily in Moffat and Routt Counties of Colorado.

The coal-bearing formations in the region include the Iles, Williams Fork, the Fort Union, and the Wasatch Formations. The total thickness of the coal seams in these formations can be up to 150 feet (Quarterly Review, 1993). Of all the formations, the Williams Fork is the most significant coal-bearing unit because it has the thickest and most extensive coalbeds. Coal-bearing strata are 5,000 feet deep along the basin's western portions and outcrop along its southern and eastern margins. The coal seams are interbedded with sandstones and shale. The thickest total coal deposits in the Williams Fork Formation, up to 129 feet, are centered on Craig, CO. These deposits are composed of several separate seams up to 25 feet thick interspersed between layers of sedimentary rock.

Coalbed methane resources in the Sand Wash Basin have been estimated at 101 Tcf. Approximately 90 percent of this gas is in the Williams Fork Formation. Approximately 24 Tcf of coalbed methane are located less than 6,000 feet below ground surface (Kaiser et al., 1994a). Some investigation and very limited commercial development of this resource have occurred, mostly in the late 1980s and early 1990s. Records from the Colorado Oil and Gas Commission indicate that approximately 31 Bcf of coalbed methane was produced in Moffat County during 1995 (Colorado Oil and Gas Conservation Commission, 2001). There appears to be no commercial production at present (GTI, 2002). Development of coalbed methane resources in the Sand Wash Basin has been slower than in many other areas due to limited economic viability. The need for extensive dewatering in most wells has been a limiting factor, compounded by relatively low coalbed methane recovery. In recent years, permits for new gas wells have been issued, indicating that there may be some continued interest in this area (Colorado GIS, 2001).

Kaiser and Scott (1994) summarized their extensive investigation of groundwater movement within the Fort Union and Mesaverde Group. The Mesaverde Group is a highly transmissive aquifer. The coal seams within the group may be the most permeable part of the aquifer. Lateral flow within the Fort Union Formation is slower. Groundwater quality in the basin varies greatly. Typically, chloride and TDS concentrations within the coal-bearing Mesaverde Group are low and potentially within potable ranges in the eastern portion of the basin, implying the existence of a USDW. TDS concentrations increase as the water migrates toward the central and western margins of the basin. TDS concentrations significantly higher than the 10,000 mg/L USDW water quality standard have been detected in the western portion of the basin.

The use of fracturing fluids, specifically water and sand proppant, has been reported for this basin. No record of any other fluid types has been noted. Although variable, the water quality within the fractured coals indicates the presence of USDWs within the coalbeds.

5.11 The Washington Coal Regions (Pacific and Central)

The Pacific Coal Region (Figure 5-1) is approximately 6,500 square miles and lies along the western and eastern flanks of the Cascade Range, from Canada into northern Oregon within the Puget downwarp structure. Bellingham, Seattle, Tacoma, and Olympia in Washington, and Portland, Oregon, lie in or adjacent to the sub-basins. The Central Coal Region (Figure 5-1) primarily lies within the Columbia Plateau, between the Cascade Range to the west and the Rocky Mountains to the east, in Idaho. This region extends from the Okanogan highlands to the north to the Blue Mountains to the south, and encompasses approximately 63,320 square miles.

The coal-bearing deposits of the Pacific and the Central Coal Regions are Cretaceous to Eocene Age and formed within fluvial and deltaic depositional environments prior to the uplift of the Cascade Mountain Range. The thick coalbeds of the Pacific and Central Basins are thought to result from peat accumulations in poorly drained swamps of the lower deltas, while the thinner coalbeds probably formed in the better drained upper deltas (Buckovic, 1979 as cited by Choate et al., 1980). The complex stratigraphy and structural deformation of the coals of the Pacific Coal Region are major obstacles to the exploration and development of gas fields. Although the coals of the Central Coal Region may not be as greatly deformed and unpredictable as those in the Pacific Coal Region, they are obscured by the Columbia River Basalt Group, in which individual basalt flows up to 300 feet thick can cover thousands of square miles.

The occurrence of methane in groundwater is one factor leading to the identification of the gas potential in Washington. Methane in groundwater occurs in the basalts, but only in confined aquifers (porous or fractured zones near the top or bottom of a basalt layer) and is thought to have migrated upward from underlying coalbeds. Choate et al. (1980) estimated coalbed methane resources for four target sub-basins representing 1,800 square miles of the Pacific Coal Region to be 0.3 trillion to 24 Tcf. Methane had been encountered in 67 oil and gas exploration wells drilled in this region by 1984. Gas was found at depths of less than 500 feet in 25 wells, less than 1,000 feet in 38 wells, and less than 2,000 feet in 50 wells. Pappajohn and Mitchell (1991) estimated the coalbed methane potential of the Central Coal Region to be more than 18 Bcf per square mile. The operation of the Rattlesnake Hills gas field between 1913 and 1941 in the western part of the Central Coal region indicates that greater potential for development may exist. According to the available literature, there were no producing fields in either the Pacific Coal Region or the Central Coal Region in Washington as of 2000 (GTI, 2001).

Water supply wells and irrigation wells in the Columbia River Basalts and water wells in numerous different lithologies in the Pacific Coal Region have been recognized as containing methane. Data demonstrating the co-location of a coal seam and a USDW were found for Pierce County, where methane gas test well results report TDS levels far less than the 10,000 mg/L USDW water quality threshold (Dion, 1984). These aquifers can be classified as USDWs. Data demonstrating the co-location of a coal seam and a USDW was found for Pierce County, where methane gas test well results report TDS levels of 1,330 to 1,660 mg/L, which is far less than the USDW classification limit (Dion, 1984). Development of methane in the Central Coal Region may have some impact on highly productive basalt aquifers already used as large sources of irrigation water for agriculture (Dion, 1984).

Hydraulic fracturing of coalbed methane wells using sand and nitrogen foam treatments has been documented (Quarterly Review August, 1993). However, optimal stimulation and completion methods for use in the structurally difficult Pacific gas region are yet to be applied and proven.

5.12 Summary

Hydraulic fracturing of coalbed methane production wells has been documented in each basin, although it is not widely practiced in the Powder River, Sand Wash Basin, or the Washington Coal Regions. Ten of the eleven major coal basins in the United States are located at least partially within USDWs. The literature also indicates that hydraulic fracturing may have increased or have the potential to increase the communication between coal seams and adjacent aquifers in two of the basins: the Powder River and Raton Basins. This may be the explanation for higher than expected withdrawal rates for production water in the Raton Basin following some fracturing treatments. In the Powder River Basin, concerns over the creation of such a hydraulic connection are cited as one reason why hydraulic fracturing of coalbed methane reservoirs is not widely practiced in the region.

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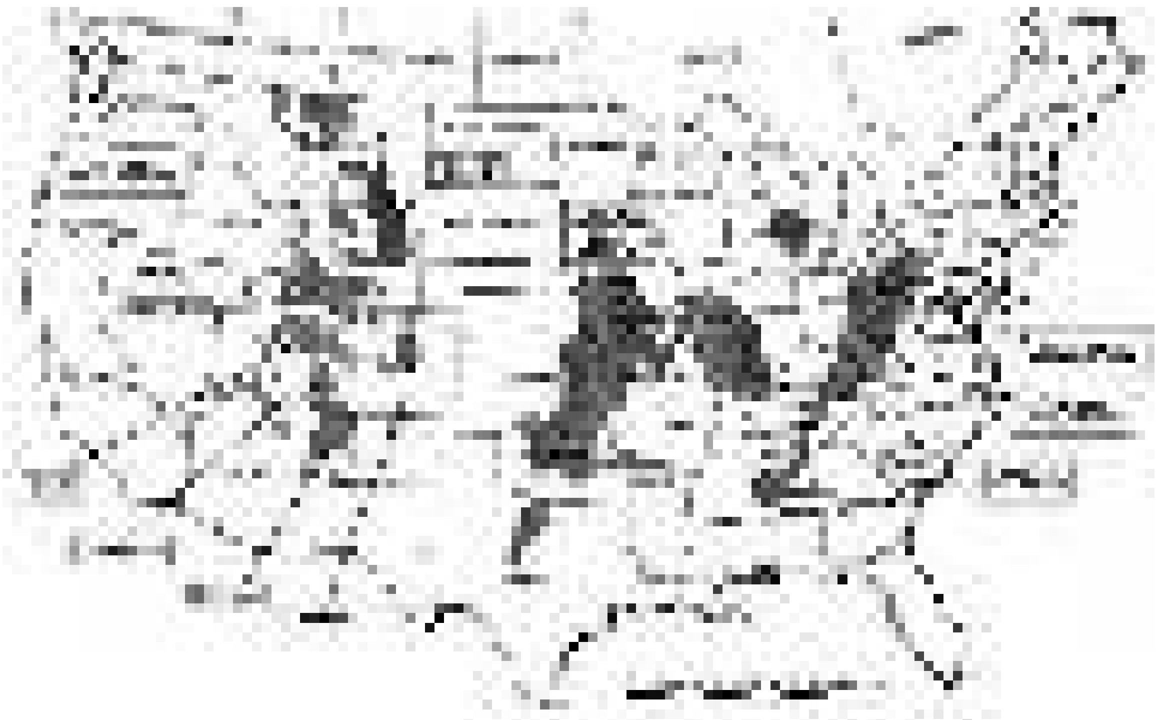
Figure 5-1. Locus Map of Major United States Coal Basins



Chapter 6
Water Quality Incidents

While Chapters 3 through 5 describe the theoretical and technical background for the potential contamination of USDWs from hydraulic fracturing fluid injection into coalbed methane wells, this chapter summarizes citizens' accounts of water quality and quantity incidents. These reports reflect the opinions of citizens living near coalbed methane operations who expressed concerns about contaminated drinking water wells and wells experiencing water quantity impacts such as reduced production. EPA has, through letters and telephone calls, contacted and been contacted by citizens who believed their water wells were affected by coalbed methane production in the San Juan, Black Warrior, Central Appalachian, and Powder River Basins. Stakeholders commenting on the study methodology (65 FR 45774 (USEPA, 2000)) asked that EPA consider personal experiences regarding coalbed methane impacts on drinking water wells in addition to data from formal studies.

As a result of the stakeholder comments, EPA published a request in the *Federal Register* (66 FR 39396 (USEPA, 2001)) for information from the public, as well as governmental and regulatory agencies, regarding incidents of groundwater contamination believed to be associated with hydraulic fracturing of coalbed methane wells. In addition, the Agency notified over 500 local and county agencies in areas of potential coalbed methane production making them aware of the *Federal Register* notice, but EPA received no information regarding citizen complaints from these officials. Therefore, EPA believes it knows the major geographic areas where citizens have reported problems that they attribute to coalbed methane development. These areas are concentrated in the most active basins: the San Juan, Black Warrior, Central Appalachian, and Powder River Basins. The Agency has included relevant information from the water quantity and quality incident reports that it has received.

Many of the reported incidents (such as impacts to water supply quantities or the effects of discharged groundwater extracted during the coalbed methane production process) are outside of the scope of SDWA and beyond the scope of this Phase I of the study. However, all incidences reported in response to the *Federal Register* request are included so that this study can be as inclusive as possible with respect to reported incidences and not inadvertently exclude a relevant reported incident. This study is specifically focused on assessing the potential for contamination of USDWs from the injection of hydraulic fracturing fluids into coalbed methane wells, and determining based on these findings, whether further study is warranted.

It is important to note that activities or conditions other than hydraulic fracturing fluid injection may account for some of the reported incidences of the contamination of drinking water wells. These potential causes include surface discharge of fracturing and production fluids, poorly sealed or poorly installed production wells, and improperly abandoned production wells.

For this phase of the study (Phase I), EPA consulted with state agencies to determine if they had received reports of groundwater problems, to learn of any follow-up steps typically taken by the state, and to determine the states' overall findings regarding any impacts that hydraulic fracturing of coalbed methane wells may have had on groundwater.

This chapter summarizes correspondence EPA has had with individual citizens and states, organized by basin, as follows:

- San Juan Basin (Colorado and New Mexico).
- Powder River Basin (Wyoming and Montana).
- Black Warrior Basin (Alabama).
- Central Appalachian Basin (Virginia and West Virginia).

6.1 The San Juan Basin (Colorado and New Mexico)

For over a decade, citizens in the San Juan Basin region have reported that coalbed methane development has resulted in increased concentrations of methane and hydrogen sulfide in their water wells. Other complaints about coalbed methane development include the loss of water, the appearance of anaerobic bacteria in water wells, and the transient appearance of particulates in well water. In conversations with EPA, most citizens and local government officials did not specify hydraulic fracturing as the cause of well water problems. Summaries of reported incidents and state follow-up are discussed in sections 6.1.1 and 6.1.2, respectively.

EPA reviewed the BLM study summarizing the history of methane seeps, citizen complaints, and follow-up investigations related to conventional gas and coalbed methane development in the San Juan Basin to determine if they contained information pertaining to coalbed methane hydraulic fracturing and its impact, if any, on the quality of water in drinking water aquifers in the basin. A summary of pertinent findings is provided in section 6.1.3.

6.1.1 Summary of Reported Incidents

- EPA spoke with a former county employee who, earlier in his career, had worked for Exxon performing hydraulic fracturing jobs (Holland, 1999). As a county employee, he took measurements for methane and hydrogen sulfide inside homes in response to citizen complaints. He indicated that there were no significant problems until the shallowest formation of coal (the Fruitland Formation) began being developed. He believed that the main route of contamination is from older, poorly cemented wells, and he estimated that hundreds of wells have been affected. He said the biggest problems associated with the apparent effects of coalbed methane development are the explosive levels of methane and the toxic levels of hydrogen sulfide in homes. In his opinion, this is due to the removal of water, rather than to hydraulic fracturing.

- The San Juan Citizens Alliance estimated that hundreds of private water wells have been affected by coalbed methane production in the area of Durango, CO. These complaints include the following:

  - A lawyer representing several Durango citizens whose wells were contaminated, allegedly due to coalbed methane development, said there have always been methane seeps in the river, which have manifested as bubbling water (McCord, 1999). In the early 1980s, however, people began to see increased concentrations of natural gas in their water wells shortly after companies began producing methane from the Fruitland Formation.

  - One individual reported that two of his wells were degraded because of increased methane levels. According to this individual, his neighbor's pump house door was blown off, presumably as a result of explosive levels of methane. Amoco bought three ranches after county officials tested indoor air and found extremely high levels of methane. This individual also told EPA staff that an area of the Southern Ute tribal land has increased levels of hydrogen sulfide at the surface. He reported he had also heard of black water due to pulverized coal.

  - Another private well owner claimed that her neighbors' wells are contaminated by gas infiltration from dewatering. First methane contaminates the well, then hydrogen sulfide, then anaerobic bacteria. She claimed that data exists showing that methane concentrations in water have increased by 1,000 parts per million (ppm).

- EPA Region 8 received letters from citizens concerned that coalbed methane development had contaminated their water with methane and hydrogen sulfide.

- During a visit to Durango, CO, EPA met with several citizens who claimed to have experienced problems with their water due to coalbed methane development. Most of the citizens experienced water loss, but two well owners from New Mexico claimed that the quality of their water was affected by hydraulic fracturing. According to their accounts, the water turned cloudy with grayish sediment a day or two after nearby fracturing events. Eventually, the well water returned to its normal appearance.

EPA also toured the area during that visit. EPA staff viewed areas where patches of grass and trees were turning brown and dying. In some places, large, old-growth trees located within the patch indicated that the area previously had prolonged normal soil conditions. Many citizens and some local officials believed that the areas suffered from increased methane and decreased air in the soil gas in the shallow root zone.

- A La Plata County official reported that citizens have called to complain that well water flow decreases when coalbed methane wells are hydraulically fractured (Keller, 1999). He reported that "a lot" of people are hauling water due to water loss. The county official said that, in two separate reports, well owners noticed problems with their well water approximately 2 weeks after nearby fracturing events. They reportedly believe hydraulic fracturing is responsible because the timing of the water loss coincides with the fracturing. Citizens know when gas producers fracture wells because they can see and hear the operation, which involves several trucks, tanks, manifolds, and mobile trailers. The county official noted that the formation being developed, the Fruitland Formation, is located approximately 2,400 feet below ground surface (bgs), and water wells are generally drilled from 100 feet to 200 feet bgs. He qualified his statements by indicating that wells do go dry for a variety of reasons.

- EPA contacted the Colorado Department of Health (CDH), which has primacy for the UIC Program under SDWA. An official with whom EPA spoke said CDH believes that water removal associated with coalbed methane development has caused problems in private water wells (Bodnar, 1999).

- EPA received one complaint from a citizen living in the Raton Basin in Trinidad, CO. She reported that water wells in her area have begun to decline in production and quality, often producing more and more gas. She believes the decline of water wells in her area is due to dewatering associated with coalbed methane production.

6.1.2 State Agency Follow-Up in the San Juan Basin

Colorado Oil and Gas Conservation

The Colorado Oil and Gas Conservation Commission (COGCC) is responsible for environmental issues related to oil and gas production in the state. The COGCC responds to every complaint called in to its office (Baldwin, 2000).

The COGCC staff believes that increased methane concentrations in water wells and buildings in some areas are partially due to old, improperly abandoned gas wells and older, deeper conventional gas wells in which the Fruitland Formation was not completely isolated. The state bases its opinions on monitoring and studies conducted in the San Juan Basin in response to complaints (see section 6.1.3). According to COGCC officials, the state's mitigation program focused on sealing old, improperly abandoned gas wells and appears to have reduced methane concentrations in approximately 27 percent of the water wells sampled. They believe that methane concentrations will decrease over time in other water wells where the source of the methane was gas wells. There are other areas of the San Juan Basin where the methane in water wells is produced by methanogenic bacteria in the aquifer. Methane concentrations in water wells in these areas probably will not decrease.

Officials cite studies that use stable carbon and hydrogen isotopes of methane and gas composition to differentiate between thermogenic methane from the Fruitland, Mesaverde, and Dakota Formations, and biogenic methane that is produced in shallower formations by naturally occurring methanogenic bacteria. By 1998, approximately two-thirds of the water wells for which gas isotopic analyses had been performed appeared to contain biogenic gas, while one-third appeared to contain thermogenic gas.

The state also noted that, in the interior basin, 1,100 feet of shale separates the Fruitland Formation and the shallow formations in which private wells are completed.

New Mexico Oil Conservation Division

EPA spoke with a District Geologist employed by the New Mexico Oil Conservation Division (NMOCD). He said that several years ago the office received many complaints that methane had contaminated water wells (Chavez, 2001). The state held water fairs at which anyone could have his or her water tested. In addition, the state initiated a program for cemented wells (some active, some abandoned) that prohibited open holes 100 feet above the casing string. The District Geologist indicated that the program seemed to solve the problem and that NMOCD has not received many subsequent complaints.

6.1.3 Major Studies That Have Been Conducted in the San Juan Basin

As noted previously, EPA reviewed a BLM study on the San Juan Basin to determine if it contained information pertaining to coalbed methane hydraulic fracturing and its impact, if any, on the quality of water in drinking water aquifers in the basin. EPA's review of this report focused on the two potential mechanisms by which hydraulic fracturing may affect the quality of USDWs: 1) direct injection of hydraulic fracturing fluids into a USDW or injection of fracturing fluids into a coal seam already in hydraulic communication with a USDW (e.g., through a natural fracture system), and 2) creation of a hydraulic connection between the coalbed formation and an adjacent USDW. The reports did not specifically address hydraulic fracturing, and only very little information indirectly addresses the question specific to this study: *Does the injection of hydraulic fracturing fluids into coalbed methane wells contaminate USDWs*?

The studies provided information on evidence that a hydraulic connection exists between coalbeds in the Fruitland Formation and overlying shallow aquifers and on possible conduits that may be the basis of the hydraulic connection. For example, the presence in a shallow aquifer of methane documented to be from the underlying Fruitland Formation is indirect evidence of a hydraulic connection, through some type of conduit, between the Fruitland Formation and shallower formations.

Evidence that a hydraulic connection exists between coalbeds and the shallow aquifer

The U.S. Department of the Interior's BLM (1999) provides a history of gas seeps and methane contamination of drinking water wells in the San Juan Basin. This section will review the evidence that indicates the existence of a hydraulic connection between the deep coalbeds and shallow USDWs.

Even prior to oil and gas drilling operations, shallow water wells in the San Juan Basin produced methane gas. Some wells in the Cedar Hill, NM, area of the basin were reported to have a strong sulfur odor. Some shallow water wells around the basin rim penetrated the Fruitland and Menefee coalbeds and produced methane (BLM, 1999). Thus, coalbed methane was the source of at least some of the observed methane contamination. Water from the Fruitland coalbed discharges in the western part of the basin and migrates upward across the Kirtland shale into the Animas and San Juan Rivers (Stone et al., 1983). In areas such as La Plata County, CO, along the northern and western rims of the basin, the methane presumably moves through natural fractures.

In the interior of the basin, gas seeps were observed in pastures in the Animas River Valley south of Durango near Bondad, CO, and Cedar Hill, NM, in the early to mid-1980s. Bubbles were also observed in the Animas River and in the tap water of rural properties in these areas. Methane was responsible for explosions in several pump houses. A landowner in New Mexico reported that gas was bubbling out of his alfalfa field and in the Animas River in 1985. Gas seeps were likely the cause of patches of dead grass growing in soils overlying the Mesaverde sandstone (BLM, 1999). Thus, conduits between methane-containing units and the surface were present both at the rim and in the interior of the basin.

After coalbed methane production began in the basin in the late 1980s, a local citizens' group voiced concerns that natural gas contamination of drinking water wells had increased in La Plata County. One study reported that 34 percent of the 205 domestic water wells tested in the county showed measurable concentrations of methane (BLM, 1999). This appears to indicate that there is a conduit for fluid to flow to the shallower USDW and its drinking water wells.

Shortly after the start of coalbed methane production in the basin, 11 coalbed methane wells were drilled within 2 miles of the Pine River Ranches Subdivision at the rim of the San Juan Basin. Nine to 35 feet of alluvium separate the surface from the Fruitland Formation coals in this area. A number of problems were reported following the onset of coalbed methane production. A man who complained that his well was contaminated with methane saw streams of gas bubbles in the nearby Los Pinos River. His report of methane contamination was confirmed by the San Juan Regional Authority (SJRA), which investigated reported contamination of this well and nearby wells. The other wells were also contaminated with methane. Two of the 4 residences near the 11 coalbed methane wells contained explosive levels of methane in crawl spaces (BLM, 1999). The methane sampled in the shallow wells and the bubbling river and the high concentrations of methane detected in residences suggest that coalbed methane was following some conduit from the Fruitland Formation to the surface or to shallow USDWs.

Evidence that methane in shallow drinking water wells originates in the Fruitland Formation (location of the coalbeds targeted by hydraulic fracturing)

Several lines of evidence show that methane detected in alluvial wells is not a result of sewage-derived methane contamination (BLM, 1999). Rather, the methane in the domestic wells studied originates either in conventional gas reservoirs such as the Dakota sandstone and the Lewis Shale or in the coals of the Fruitland Formation.

The composition of the gas in samples from shallow, private drinking water wells was analyzed to confirm the well owners' observations. The data obtained showed that the methane in approximately half of the samples appeared to have originated in the Fruitland Formation coalbeds and not from other possible sources such as septic tanks (BLM, 1999).

Similar sampling and analyses conducted in an additional study cited by BLM (1999) concluded that gas in a domestic well in alluvium overlying the Fruitland Formation had the same gas composition and carbon-13 isotope ratio as gas from a nearby gas well also in the Fruitland Formation. This study found that C13 isotopic signatures of individual near-surface gas samples correlated with production gas from discrete formations beneath the study area (BLM, 1999). In addition, an area resident's well contained 680 ppm TDS, primarily sodium bicarbonate. Fruitland-produced water has the same composition, although other domestic wells in the area do not. (TDS values tend to be in the 100 to 200 ppm in these other domestic wells.) Both the gas and the water analyses indicate that the shallow aquifer in the area (from which the methane-contaminated domestic wells draw drinking water) is in hydraulic communication with the deeper Fruitland Formation coalbeds.

Possible conduits for fluid movement from the coalbeds to the aquifer

Several studies have assessed possible natural or manmade conduits to account for the confirmed occurrence of methane in wells tapping the shallow aquifer that overlies the deeper coalbeds in the Fruitland Formation. Possible pathways enabling methane to move from a deep source to a shallow aquifer include natural fractures, hydraulically induced fractures, disposed of produced water from coalbed methane wells, and poorly constructed, sealed, or cemented conventional gas wells, coalbed methane wells, shallow drinking water wells, and cathodic protection wells installed to protect oil and gas pipelines from corrosion (BLM, 1999).

The history of documented gas seeps and methane occurrence in water wells indicates that natural fractures probably serve as conduits in parts of the basin where coal formations are near or at the surface and in the interior of the basin, where the coal formations are deeper. These conduits may enable hydraulic fracturing fluids to travel from targeted coalbeds to shallow aquifers. However, there is no unequivocal evidence that this fluid movement is occurring and, even given the presence of these possible conduits, other hydrogeologic conditions (such as certain pressure gradients, etc.) would be required for fluid movement from targeted coalbeds to shallow aquifers.

A study comparing soil-gas-methane concentrations adjacent to 352 gas-well casings and 192 groundwater wells found that the gas-well annuli (i.e., the spaces between the steel well casings and the walls of the drilled bore holes) were frequently the reason methane moved from the coalbeds to the near-surface environment (BLM, 1999). Thus, gas-well annuli are clearly one type of conduit for movement of methane from deeper sources up to overlying shallow aquifers.

The possibility of leaking gas wells acting as conduits through which methane flows from the Fruitland Formation to shallow aquifers was investigated by a joint Colorado Oil and Gas Conservation Commission/BLM study (BLM 1999). One hundred twenty water wells were tested for methane before and after nearby gas wells were “remediated” (better sealed). The study concluded that the relationship between gas well remediation and lower methane concentrations in drinking water was “complex” and may have been affected by the lingering presence of methane in drinking water after gas well remediation. More than half the water wells showed no significant changes in methane occurrence, a quarter showed lower methane levels, and one-tenth showed increased methane.

In summary, there appears to be evidence that methane seeps and methane in shallow geologic strata and water wells may occur because the methane moves through a variety of conduits. These conduits include natural fractures; poorly constructed, sealed, or cemented manmade wells used for various purposes. No reports provide direct information regarding hydraulic fracturing. Methane, fracturing fluid, and water with a naturally high TDS content could possibly move through any of these conduits. In some cases, improperly sealed gas wells have been remediated, resulting in decreased concentrations of methane in drinking water wells.

6.2 The Powder River Basin (Wyoming and Montana)

EPA spoke with several individuals familiar with coalbed methane activity in the Powder River Basin area who believe coalbed methane production is causing water quantity issues. These individuals have reported that dewatering during coalbed methane production resulted in loss of water from wells and in flooding problems on the surface. Many of the drinking water wells in the Powder River Basin are screened and completed in the same formation being dewatered for methane production. According to a consulting hydrogeologist, as much as 1 million gallons of water are pumped from each coalbed methane production well during its lifetime. Consequently, the aquifer has dropped 200 feet in some areas (Merchat, 1999). EPA has also learned that, as of 1999, oil and gas companies have drilled 2,000 wells in the Powder River Basin, and they reportedly plan to drill 15,000 in total (Merchat, 1999). However, deeper aquifers are available, and the oil and gas companies have drilled new water wells in those aquifers for private individuals.

Reports of incidents in the Powder River Basin are summarized below. However, hydraulic fracturing is performed infrequently in the Powder River Basin, and no one living in that area has reported problems relating to the process. Many of the complaints relate to water quantity issues, which are beyond the scope of this study.

EPA contacted the state and local offices of the Wyoming Health Department and the Water Quality Division of the Wyoming Department of Environmental Quality to determine if these departments had received complaints of water quality degradation due to coalbed methane production. Local authorities reported one complaint of black sediments in drinking water, but most concerns centered on water loss and flooding caused by large quantities of water discharged at the surface (Heath, 1999). There has been discussion among stakeholders regarding the handling of large volumes of water brought to the surface during coalbed methane production. Some individuals remain concerned about the consequences of dewatering aquifers, which include loss of the resource, effects on soil chemistry, flooding, and the potential for coalbed fires and subsidence.

EPA spoke with a consultant for the Powder River Basin Resource Council (PRBRC), a citizen's group formed around environmental issues associated with coalbed methane production (Merchat, 1999). He stated that the biggest concern among people in the area is loss of water. However, some have had problems with increased methane content in their water. He said people reported methane in the water results in frothing and bubbles. The water is generally used for agricultural purposes and for drinking water. He said that each methane well produces millions of gallons of water in its lifetime. The discharge of water has created new ponds and swamps that are not naturally occurring in that region. The secondary effects from pumping water are subsidence and clinker beds (burning coal). When underground coal catches fire from lightning, it burns until it reaches groundwater. However, if there is no groundwater, the fire will continue to burn. The cost of manually extinguishing those fires is enormous. Furthermore, the burning of the coal can leave behind benzo(a)pyrene and other polycyclic aromatic hydrocarbons that are toxic and/or carcinogenic and could affect drinking water.

EPA Region 8 is participating in a study that addresses the environmental effects of all aspects of coalbed methane development and not just hydraulic fracturing.

6.3 The Black Warrior Basin (Alabama)

The *LEAF v. EPA* case arose from an alleged water quality degradation related to activities in Alabama. As discussed in Chapter 1, the Eleventh Circuit Court's 1997 decision in *LEAF v. EPA,* 118F.3d 1467, held that because hydraulic fracturing of coalbeds to produce methane is a form of underground injection, Alabama's EPA-approved UIC Program must effectively regulate this practice (11th Cir, 1997). In response to the Court's decision, Alabama supplemented its rules governing the fracturing of wells to include additional requirements that govern the protection of USDWs during the hydraulic fracturing of coalbed methane. Summaries of reported incidents are presented in section 6.3.1 below.

6.3.1 Summary of Reported Incidents

- In the drinking water well case that precipitated *LEAF v. EPA*, an individual complained that drinking water from his well contained a milky white substance and had strong odors shortly after a fracturing event. He also reported that six months after the fracturing event his water had increasingly bad odors and occasionally contained black coal fines. The EPA Administrative Record regarding the Alabama Class II UIC Program contains other similar descriptions of well water problems.

- Another Alabama citizen reported to EPA problems with her drinking water well that began in 1989. In her letter, the citizen reported that her property was located near a coalbed methane gas well and that there was coal mining in the area. She wrote that she believes hydraulic fracturing of the coalbed methane well adversely affected her drinking water well, and coal resource exploitation in the area caused various, significant environmental damage. The individual believed that the hydraulic fracturing contributed to well contamination because, shortly after a fracturing event, her kitchen water contained globs of black, jelly-like grease and smelled of petroleum. She said her drinking water turned brown and contained slimy, floating particles. She reported that her neighbors also said their water smelled like petroleum.

  She included, as an attachment, a letter from the Alabama Oil and Gas Board (OGB) approving the use of proppants tagged with radioactive material. Their approval was based on the hydrogeology and the absence of water wells in the immediate area, the depths of the coal intervals to be fractured, well construction, and adherence to a program designed to monitor and contain radioactive material at the surface. Also attached was a letter from EPA Region 4 describing analytical results for samples the Agency collected from her drinking water well on June 26, 1990. The results indicated no purgeable and extractable organic compounds were detected. In addition, the letter said that a water/oil inter-phase detector was used to determine if petroleum products were floating in the well, and none was detected.

- An Alabama homeowner complained to the Natural Resources Defense Council that recovered hydraulic fracturing fluid from a nearby coalbed methane well installation was allowed to drain from the coalbed methane well site to a location near her home. She claimed that this fluid was initially obtained from an abandoned strip-mining quarry that had been used as a landfill for municipal and industrial waste. As this fluid drained from the fracturing site, the homeowner asserted, it killed all animal and plant life in its path. She further stated that shortly after this fracturing event and the associated runoff, her 110-foot deep drinking water well became contaminated with brown, slimy, petroleum-smelling fluid similar to the discharged fracturing fluid from the coalbed methane well site.

- In response to EPA's July 2001 call for information on water quality incidents (found in Water Docket W-01-09), an individual reported that her drinking water well had become filled with methane gas, causing it to hiss (66 FR 39396 (USEPA, 2001)); the tap water became cloudy, oily, and had a strong, unpleasant odor. In addition, the tap water left behind an oily film and contained fine particles. The drinking water well owner had her well tested by a private consultant, who confirmed the presence of methane.

  The Alabama OGB tested this drinking water well, but only looked for naturally occurring contaminants. EPA also sampled and tested this drinking water well, but not until 6 months after the event. No mention is made of the analytical results obtained from the drinking water well by these agencies.

6.3.2 State Agency Follow-Up (Alabama Oil and Gas Board)

LEAF v. EPA originated in Alabama. The water well that was reportedly contaminated as a result of hydraulic fracturing operations was sampled independently by the Alabama OGB, the Alabama Department of Environmental Management (ADEM), and EPA Region 4. Water analyses performed by these agencies indicated that the water well had not been contaminated as a result of the fracturing operation. The Alabama OGB reported to EPA that it investigates every complaint it receives, and it does not believe that hydraulic fracturing has affected water wells. Investigations include research into historical water quality data, some of which pre-dates coalbed methane activity. Such historical information is important because the coal-bearing Pottsville Formation often contains high concentrations of iron. Groundwater from this formation may contain iron-reducing bacteria, which can sometimes result in such water having an unpleasant taste or odor, or containing a white or red-brown, stringy, gelatinous material (Valkenburg and others, 1975, as cited by the Alabama OGB, 2002). In addition, sudden iron staining can occur in water with a history of good quality. Water well yield can also decline due to the presence of iron-reducing bacteria in high concentrations.

According to the Alabama OGB, one factor considered in each investigation is whether historical data are available on water quality in a particular area, including data that pre-date coalbed methane activity. Published reports and open-file data show that the quality of water in the coal-bearing Pottsville Formation can vary from good to very poor. Data collected from the 1950s through 1970s in localities throughout a large area where the Pottsville Formation has served as a source of water contain reports of water having "bad taste," "bad odors," "oily films or sheens," and waters causing "red stains" and "black stains" (Geological Survey of Alabama, 1930s to Present; Johnston, 1933, as cited by the Alabama OGB, 2002).

The Alabama OGB reported to EPA that it has investigated several complaints of methane gas in water wells. In each instance, the Alabama OGB determined that the water well problem was unrelated to coalbed methane extraction operations, which often were not occurring in the areas of reported water problems. Moreover, in some areas methane gas was reported in water wells many years before the advent of underground mining and the commercial development of this resource (Geological Survey of Alabama, 1930s to Present, as cited by the Alabama OGB, 2002). The problem of methane gas in water wells has generally occurred where water wells, usually less than 200 feet deep, penetrated gas-bearing coal strata, particularly following low rainfall years that caused a lowering of water tables. In these areas, there commonly had been a recent increase in the drilling of water wells and an acceleration in the rates of water withdrawal from the aquifer. When sufficient amounts of water are removed from these water wells, methane can begin to desorb from the coal seams and be produced.

Alabama's regulations have been approved by EPA for incorporation into Alabama's Class II UIC Program. Operators must provide written certification to the Board that the proposed fracturing operation will not occur in a USDW or that the fracturing fluids do not exceed the MCLs in 40 CFR §141 Subparts B and G. Fracturing is prohibited from ground surface to 299 feet bgs. For all fracture jobs performed between 300 feet and 749 feet bgs, the company must perform a reconnaissance of fresh-water supply wells within ¼ mile of the well to be fractured, submit a fracturing program to the OGB, and perform a cement bond log analysis. For fracturing events performed between 750 feet and 1,000 feet bgs, only a cement bond log is required. For fracturing events performed below 1,000 feet bgs, operators must submit to the Alabama OGB the depth to be fractured, well construction information, cementing specifications, and logs identifying overlying, impervious strata.

In Alabama, Rule 400-3-8-.03 states that coalbeds shall not be hydraulically fractured until written approval of the Oil and Gas Supervisor has been obtained. The Supervisor must be notified when an approved fracturing operation is to occur so that an agent of the Board may be present. In order to receive approval, operators must submit details of the proposed fracturing operation. The Board's staff evaluates each proposal for compliance to ensure USDW protection. Basic information that must be submitted with an operator's proposal to hydraulically fracture a well includes details on the depths of coalbeds to be fractured; construction of the well, including casing and cementing specifications; a geophysical log showing the type and thickness of impervious strata overlying the uppermost coalbed to be fractured; and, if the operation is to be performed in a USDW-bearing interval, a statement certifying that fracturing fluids will not exceed the MCLs of federally mandated primary drinking water regulations (40 CFR §141 Subparts B and G). In addition to the basic information, a fracturing program, a water well inventory within a ¼-mile radius, and a cement bond log must be provided with fracturing proposals in the depth interval 300 to 749 feet. Since water supply wells are generally shallower than coalbeds, Alabama's Rule 400-3-8-.03 was designed to increasingly strengthen the requirements for USDW protection with decreasing depths of proposed fracturing operations. Furthermore, the fracturing of coalbeds shallower than 300 feet is prohibited.

6.4 The Central Appalachian Basin (Virginia and West Virginia)

EPA became aware of several complaints relating to the effects of coalbed methane production on sources of drinking water in the southwestern portion of Virginia through correspondence initiated by citizens. Information about water quality incidents was gathered through meetings and telephone conversations with members of the Virginia Division of Oil and Gas within the Department of Mines, Minerals and Energy (VDMME); local health officials; and representatives of a county citizen's group. In total, VDMME provided EPA with over 70 "Complaint Detail Reports" (registered between 1990 and 2001) that related to drinking water source impacts by coalbed methane development.

Although the majority of the incidents outlined in the complaints pertain to water-loss issues, approximately one-quarter relate to water quality. Virginians living near coalbed methane production areas reported private well and spring water contamination evidenced by oily films, soaps, iron oxide precipitates, black sediments, methane gas, and bad odor and taste. Reports of water loss in the well ranged from noticeably reduced supply rates to total loss of water from domestic drinking water wells. Summaries of reported incidents and state follow-up are discussed in sections 6.4.1 and 6.4.2, respectively.

6.4.1 Summary of Virginia Incidents

- The state received complaints of soap bubbles flowing from residential household fixtures. VDMME attributes soap coming out of water faucets to the drilling process associated with both conventional wells and coalbed methane wells. Soaps are used to extract drilling cuttings from the borehole because the foam expands, rises, and, as it rises, carries the cuttings to the surface (Wilson, 2001). These soaps may migrate from the borehole into the drinking water zone that supplies private wells during drilling of the shallow portion of the hole and before the required groundwater casing is cemented in place. In the few occurrences of soap contamination, water was provided until the soap was completely purged from the contributing area surrounding their water well.

- In early August 2001, EPA met with approximately 15 to 20 residents of Buchanan and Dickenson Counties in Virginia. Coalbed methane production activity is steadily increasing in the area surrounding Buchanan County since the coal reserves in this area have proven to be extremely profitable sources for coalbed methane in recent years (Wilson, 2001). The subjects of the citizen complaints were very similar to those logged in the VDMME complaint reports. Residents described the presence of black sediments, iron precipitates, soaps, diesel fuel smells, and increased methane gas in drinking water from their wells. One resident brought a water sample collected from her drinking water well. The water was translucent with a dark gray color and with dark black suspended sediment. Several other citizens reported drinking water supplies diminishing or drying up entirely. One resident of Buchanan County said that he had an ample water supply from his drinking well for over 54 years, until shortly after coalbed methane wells were installed on his property. He reported that within 60 days of the coalbed methane well installations, his 276-foot deep drinking water supply well, which used to produce over 20 gallons per minute of potable flow, dried up. The resident mentioned that over 380 homes in the region do not have potable water as a result of coalbed methane mining activities.

  Most of the residents said that their complaints to the state usually resulted in investigations without resolution. Some residents mentioned that the gas companies were providing them with potable water to compensate for the contamination or loss of their drinking water wells. However, the residents said that this was not adequate compensation for the impacts to, or loss of, their private drinking water supplies.

- EPA was able to record numerous complaints through telephone conversations and e-mails with Virginia residents, who reported that they believed their drinking water wells had been affected by coalbed methane industry activities. All the logged complaints were from Buchanan and

Dickenson Counties. Complaints include water loss, soapy water, diesel odors, iron and sulfur in wells, rashes from showering, gassy taste, and murky water. One report discusses a miner who was burned by a fluid, possibly hydrochloric acid used in hydraulic fracturing, that infiltrated a mineshaft. Another report describes the contamination of a stream and the resulting fish kills caused by the runoff from drilling fluids. One complainant explained that several thousand wells had "gone dry, overnight." According to the individuals EPA spoke with, compensation to homeowners for these impacts is in the form of money, newly drilled wells to replace dry or contaminated wells or temporary provision of potable water, which is supplied "until things clear out."

6.4.2 State Agency Follow-Up (VDMME)

VDMME, Division of Gas and Oil, is responsible for responding to environmental issues associated with oil and gas development; it investigates every water problem reported. Responses may include an interview with the citizen reporting the problem, a site visit, water well testing, or a review of the physical aspects of the water well and surrounding activities. According to Robert Wilson of VDMME, his agency tests for contaminants that may be introduced by drilling such as chlorides, oil and grease, and volatile organics. The results of those analyses are compared to baseline values. VDMME witnesses surface casing and plugging jobs as part of its oversight duties. VDMME reviews information from drilling and completion reports to assist with investigations into complaints.

Based on investigations of the more than 70 complaints received, VDMME believes that coalbed methane production has not affected private drinking water wells. VDMME recognizes soap migrating into drinking water wells, but considers this only a transient problem. While a number of complaints report a noticeable reduction in or a total loss of drinking water supply, in almost all cases, the state investigator determined that the water loss was not likely to be caused by local hydraulic fracturing events or coalbed methane production activity because:

- The distance from the private well to the nearest coalbed methane well is too far (1,500 feet or more) to have any impact.
- There is no hydrologic connection between the water contribution zones of the private and coalbed methane wells; therefore, it is physically impossible for coalbed methane wells to affect private drinking water wells.
- The well was constructed according to VDMME regulatory guidelines; therefore, a sufficient buffer exists between the private well and the coalbed methane well.

- The existing supply was reduced because of recent drought conditions in the region.

- The complainant experienced mechanical difficulty with his or her pumping system, which led to a reduction in pumped water; however, the supply was not affected.

According to VDMME, these citizen complaints refer to incidents that can occur during the drilling of any type of well, not just coalbed methane. The few incidents of this kind were equally divided between conventional wells and coalbed wells (VDMME, 2002).

6.5 Summary

In this chapter, EPA has presented information (in addition to technical, conceptual, or theoretical information presented previously) on personal experiences with regard to coalbed methane activities and their potential (or perceived potential) to impact drinking water wells. These personal accounts of potential incidences in four producing coal basins across the United States do not present scientific findings. However, the body of reported problems considered collectively suggest that water quality (and quantity) problems might be associated with some of the production activities common to coalbed methane extraction. These activities include surface discharge of fracturing and production fluids, aquifer/formation dewatering, water withdrawal from production wells, methane migration through conduits created by drilling and fracturing practices, or any combination of these. Other potential sources of drinking water problems include various aspects of resource development, naturally occurring conditions, population growth and historical practices.

In several of the coalbed methane investigation areas, local agencies concluded that hydraulic fracturing could not affect drinking water wells. Generally, these conclusions were based on there being a significant horizontal and/or vertical distance between the coalbed methane production wells and the drinking water wells.

Chapter 7
Conclusions and Recommendations

Under SDWA, EPA's UIC Program is responsible for ensuring that fluids injected into the ground do not endanger USDWs. The goal of the Phase I study was to assess the potential for contamination of USDWs due to the injection of hydraulic fracturing fluids into coalbed methane wells, and to determine, based on these findings, whether further study is warranted.

EPA's approach for evaluating the potential for contamination of USDWs was an extensive information collection and review of empirical and theoretical data. EPA reviewed water quality incidents potentially associated with hydraulic fracturing and evaluated the theoretical potential for hydraulic fracturing to affect the quality of USDWs through one of two mechanisms:

1. Direct injection of fracturing fluids into a USDW in which the coal is located, or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. Creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

7.1 Reported Water Quality Incidents

Citizens from Wyoming, Montana, Alabama, Virginia, Colorado, and New Mexico contacted EPA because they were concerned that their water wells were affected by coalbed methane production. The major geographic areas where citizens reported experiencing problems due to coalbed methane development are concentrated in the coal basins with the most coalbed methane activities – the San Juan, Black Warrior, Central Appalachian, and Powder River Basins. This study was initiated, partly, in response to those citizens' concerns. EPA followed-up on letters and telephone calls from citizens and resulting leads to understand specific complaints and citizens' concerns.

EPA published a *Federal Register* notice (66 FR 39396 (USEPA, 2001) requesting information on water quality incidents believed to be associated with hydraulic fracturing of coalbed methane wells. EPA notified over 500 local and/or county agencies in areas with potential coalbed methane production activity to make them aware of the *Federal Register* notice requesting information on coalbed methane-related complaints. The Agency received no information on complaints from these officials.

EPA reviewed responses and follow-up actions conducted by state agencies to address groundwater complaints involving coalbed methane. Hydraulic fracturing is not widely practiced in the Powder River Basin (which includes Wyoming and Montana) and concerned citizens from that area reported surface water and groundwater quantity problems rather than specifying hydraulic fracturing as a problem. Studies of groundwater quality in the San Juan Basin (which includes parts of Colorado and New Mexico) do not address hydraulic fracturing directly. However, problems with groundwater quantity and quality in Colorado may have plausible explanations other than hydraulic fracturing activities. For example, natural fractures, and poorly constructed, sealed, or cemented wells used for various purposes, may provide conduits for methane to move into shallow geologic strata and water wells, or even to surface water (BLM, 1999). The New Mexico Oil Conservation Division reported that citizens began reporting increased levels of methane in their water wells after coalbed methane development began in the San Juan Basin. New Mexico initiated a plugging and abandonment program to seal old, improperly abandoned production wells, which appears to have mitigated the problem (Chavez, 2001).

EPA also obtained individual incident reports from Virginia. None of Virginia's follow-up investigations provided evidence that hydraulic fracturing of coalbed methane wells had caused drinking water well problems. Incidents in Alabama were investigated by the Alabama Oil and Gas Board, the Alabama Department of Environmental Management, and EPA Region IV. Samples from drinking water wells did not test positive for constituents found in fracturing fluids. After reviewing all the available data and incident reports, EPA sees no conclusive evidence that water quality degradation in USDWs is a direct result of injection of hydraulic fracturing fluids into coalbed methane wells and subsequent underground movement of these fluids.

7.2 Fluid Injection Directly into USDWs or into Coal Seams Already In Hydraulic Communication with USDWs

To determine if USDWs are threatened by the direct injection of fracturing fluids into a USDW, EPA: 1) reviewed information on 11 major U.S. coal basins mined for coalbed methane to determine if coal seams lie within USDWs, and 2) identified components of fracturing fluids. EPA also used the information on the 11 major U.S. coal basins as well as information collected on water quality incidents potentially associated with hydraulic fracturing to determine if coal seams are already in hydraulic communication with USDWs. Hydraulic fracturing has been, or is being, performed in every basin reviewed. As summarized in Table 5-1 in Chapter 5, evidence suggests that coalbeds in 10 of the 11 major coal basins in the United States are located at least partially within USDWs. The coalbeds in the Piceance Basin in Colorado, however, are several thousand feet below USDWs, and are unlikely to be in hydraulic communication with USDWs.

Hydraulic fracturing fluids injected into coalbed methane wells consist primarily of water, or inert nontoxic gases, and/or nitrogen foam and guar (a naturally occurring substance derived from plants). According to information gathered from MSDSs, on-site reconnaissance of fracturing jobs, and interviews with service company employees, some hydraulic fracturing fluids may contain constituents of potential concern. Table 4.1 in Chapter 4 lists examples of chemicals found in hydraulic fracturing fluids according to the MSDSs. Constituents of potential concern include the following substances either alone or in combination: bactericides, acids, diesel fuel, solvents, and/or alcohols. Although the largest portion of fracturing fluid constituents is nontoxic (>95% by volume), direct fluid injection into USDWs of some potentially toxic chemicals does take place.

For example, potentially hazardous chemicals are introduced into USDWs when diesel fuel is used in fracturing fluids in operations targeting coal seams that lie within USDWs. Diesel fuel contains constituents of potential concern regulated under SDWA – benzene, toluene, ethylbenzene, and xylenes (i.e., BTEX compounds). However, the threat posed to USDWs by introduction of these chemicals is reduced significantly by coalbed methane production's dependence on the removal of large quantities of groundwater (and injected fracturing fluids) soon after a well has been hydraulically fractured. EPA believes that this groundwater production, combined with the mitigating effects of dilution and dispersion, adsorption, and potentially biodegradation, minimize the possibility that chemicals included in the fracturing fluids would adversely affect USDWs.

Because of the potential for diesel fuel to be introduced into USDWs, EPA requested, and the three major service companies agreed, to eliminate diesel fuel from hydraulic fracturing fluids that are injected directly into USDWs for coalbed methane production. Industry representatives estimate that these three companies perform approximately 95 percent of the hydraulic fracturing projects in the United States. These companies signed an MOA on December 15, 2003 and have indicated to EPA that they no longer use diesel fuel as a hydraulic fracturing fluid additive when injecting into USDWs for coalbed methane production (USEPA, 2003).

7.3 Breach of Confining Layer

The second mechanism by which hydraulic fracturing may affect the quality of USDWs is fracturing through a hydrologic confining layer, and creation of a hydraulic communication between a coal seam and an overlying USDW. If sufficiently thick and relatively unfractured shale is present, however, it may act as a barrier not only to fracture height growth, but also to fluid movement.

A hydraulic fracture will propagate perpendicularly to the minimum principal stress. In some shallow formations, the least principal stress is the overburden stress; thus, the hydraulic fracture will be horizontal. In deeper reservoirs, the least principal stress will likely be horizontal; thus, the hydraulic fracture will be vertical. In general, horizontal fractures are most likely to exist at shallow depths (less than 1,000 feet) (Nielsen and Hansen, 1987 as cited in Appendix A: DOE, Hydraulic Fracturing). Most coal seams currently used for methane production are relatively shallow compared to conventional oil production wells, but still lie deeper than 1,000 feet.

Hydraulic fracturing may have increased or have the potential to increase the communication between coal seams and adjacent formations in some instances. For example, in the Raton Basin, some fracturing treatments resulted in higher than expected withdrawal rates for production water. Those increases, according to literature published by the Colorado Geologic Survey, may be due to well stimulations creating a connection between targeted coal seams and an adjacent sandstone aquifer (Hemborg, 1998). In the Powder River Basin, concerns over the creation of such a hydraulic connection are cited as one reason why hydraulic fracturing of coalbed methane reservoirs is not widely practiced in the region. Some studies that allow direct observation of fractures (i.e., mined-through studies) also provided evidence that fractures move through interbedded layers, sometimes taking a stair-step pathway through complex fracture systems, and sometimes enter or propagate through geologic strata above the coal (i.e., roof rock) (Diamond, 1987a and b; Diamond and Oyler, 1987; Jeffrey et al., 1993).

Fracture height is important to the issue of whether or not hydraulic fracturing fluids can affect USDWs because shorter fractures are less likely to extend into a USDW or connect with natural fracture systems that may transport fluids to a USDW. The extent of a fracture is controlled by the characteristics of the geologic formation (including the presence of natural fractures), the volume and types of fracturing fluid used, the pumping pressure, and the depth at which the fracturing is being performed. Deep vertical fractures can propagate vertically to shallower depths and develop a horizontal component (Nielsen and Hansen, 1987, as cited in Appendix A: DOE, Hydraulic Fracturing). In these “T-fractures,” the presence of coal fines or a zone of stress contrast may cause the fracture to “turn” and develop horizontally, sometimes at the contact of the coalbed and an overlying formation (Jones et al., 1987b; Morales et al., 1990).

The low permeability of relatively unfractured shale may help to protect USDWs from being affected by hydraulic fracturing fluids in some basins. At some sites, shale may act not only as a hydraulic barrier, but also as a barrier to fracture height growth. Shale’s ability to act as a barrier to fracture height growth is due primarily to the stress contrast between the coalbed and the higher-stress shale (see Appendix A)

Another factor controlling fracture height can be the highly cleated nature of some coalbeds. In some cases, highly cleated coal seams will prevent fractures from growing vertically. When the fracture fluid enters the coal seam, it is contained within the coal seam's dense system of cleats and the growth of the hydraulic fracture will be limited to the coal seam (see Appendix A).

Mined-through studies indicate many hydraulic fractures that penetrate into, or sometimes through, formations overlying coalbeds can be attributed to the existence of pre-existing natural fractures. However, given the concentrations and flowback of injected fluids, and the mitigating effects of fate and transport processes, EPA does not believe that possible hydraulic connections under these circumstances represent a significant potential threat to USDWs.

7.4 Conclusions

Based on the information collected and reviewed, EPA has concluded that the injection of hydraulic fracturing fluids into coalbed methane wells poses little or no threat to USDWs and does not justify additional study at this time. This decision is consistent with the process outlined in the April, 2001 Final Study Design, in which EPA indicated that it would determine whether further investigation was needed after analyzing the Phase I information. Specifically, EPA determined that it would not continue into Phase II of the study if the investigation found that no hazardous constituents were used in fracturing fluids, hydraulic fracturing did not increase the hydraulic connection between previously isolated formations, *and* reported incidents of water quality degradation were attributed to other, more plausible causes.

Although potentially hazardous chemicals may be introduced into USDWs when fracturing fluids are injected into coal seams that lie within USDWs, the risk posed to USDWs by introduction of these chemicals is reduced significantly by groundwater production and injected fluid recovery, combined with the mitigating effects of dilution and dispersion, adsorption, and potentially biodegradation. Additionally, EPA has reached an agreement with the major service companies to voluntarily eliminate diesel fuel from hydraulic fracturing fluids that are injected directly into USDWs for coalbed methane production.

Often, a high stress contrast between adjacent geologic strata results in a barrier to fracture propagation. This may occur in those coal zones where there is a geologic contact between a coalbed and a thick, higher-stress shale that is not highly fractured. Some studies that allow direct observation of fractures (i.e., mined-through studies) indicate many fractures that penetrate into, or sometimes through, formations overlying coalbeds can be attributed to the existence of pre-existing natural fractures. However, and as noted above, given the concentrations and flowback of injected fluids, and the mitigating effects of dilution and dispersion, fluid entrapment, and potentially biodegradation, EPA does not believe that possible hydraulic connections under these circumstances represent a significant potential threat to USDWs.

EPA also reviewed incidents of drinking water well contamination believed to be associated with hydraulic fracturing and found no confirmed cases that are linked to fracturing fluid injection into coalbed methane wells or subsequent underground movement of fracturing fluids. Although thousands of coalbed methane wells are fractured annually, EPA did not find confirmed evidence that drinking water wells have been contaminated by hydraulic fracturing fluid injection into coalbed methane wells.

Public Comment and Response Summary for the Study on the Potential Impacts of Hydraulic Fracturing of Coalbed Methane Wells on Underground Sources of Drinking Water

FINAL

Office of Water
Office of Ground Water and Drinking Water (4606M)
EPA 816-R-04-004
www.epa.gov/safewater
June 2004

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Public Comment and Response Summary for the Study on the Potential Impacts of Hydraulic Fracturing of Coalbed Methane Wells on Underground Sources of Drinking Water

FINAL

June 2004

United States Environmental Protection Agency
Office of Water
Office of Ground Water and Drinking Water
Drinking Water Protection Division
Prevention Branch
1200 Pennsylvania Avenue, NW (4606M)
Washington, DC 20460

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LIST OF ACRONYMS AND ABBREVIATIONS

BLM	Bureau of Land Management
BTEX	Benzene, Toluene, Ethylbenzene, and Xylenes
CBM	Coalbed Methane
CCL	Contaminant Candidate List
CFR	Code of Federal Regulations
COGCC	Colorado Oil and Gas Conservation Commission
EIS	Environmental Impact Statement
EPA	United States Environmental Protection Agency or Agency
FR	Federal Register
GWPC	Ground Water Protection Council
MCL	Maximum Contaminant Level
MOA	Memorandum of Agreement
MSDS	Material Safety Data Sheet
MTBE	Methyl Tert Butyl Ether
NAS	National Academy of Science
PWS	Public Water System
RfD	Reference Dose
SDWA	Safe Drinking Water Act
UCMR	Unregulated Contaminant Monitoring Regulation
UIC	Underground Injection Control Program
USDW	Underground Source of Drinking Water

Public Comment and Response Summary for the Study on the Potential Impacts of Hydraulic Fracturing of Coalbed Methane Wells on Underground Sources of Water

I. INTRODUCTION

The United States Environmental Protection Agency's (EPA's) Office of Ground Water and Drinking Water completed its Phase I study, which assesses the potential for contamination of underground sources of drinking water (USDWs) from the injection of hydraulic fracturing fluids into coalbed methane (CBM) wells. EPA (or the Agency) began collecting information on hydraulic fracturing in the fall of 2000. Based on the information collected and reviewed, EPA has concluded that the injection of hydraulic fracturing fluids into CBM wells poses little or no threat to USDWs and does not justify additional study at this time.

The draft report, titled, "Draft Evaluation of Impacts to Underground Sources of Drinking Water by Hydraulic Fracturing of Coalbed Methane Reservoirs" (hereafter referred to as the draft report), was made available for public comment by an announcement in the *Federal Register* on August 28, 2002.[1] The 60-day public comment period officially ended on October 28, 2002.

The Agency received and reviewed comments from 105 commenters. Several of these were signed by multiple parties (which were counted as one commenter), including a few coalitions of environmental organizations. The commenters include private citizens; environmental and citizen groups; government agencies at the local, state, and national levels; oil and gas companies; trade associations; and four other commenters that do not fit these categories. Table 1 below provides a listing of these commenters.

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[1] US Environmental Protection Agency. 2002. Underground Injection Control (UIC) Program; Hydraulic Fracturing of Coalbed Methane (CBM) Wells Report--Notice. *Federal Register*. Vol. 67, No. 167. p. 55249, August 28, 2002.

TABLE 1: LIST OF PUBLIC COMMENTERS		
Docket ID[1]	**Edocket ID (OW-2001-0002)[2]**	**Organization (State)**
Environmental/Citizens Groups		
II-D1.014	045	Bull Mountain Landowners Association (MT)
II-D1.025	055	Land and Water Fund of the Rockies (CO)
II-D1.040	068	Dickenson County Citizens Committee (VA)
II-D1.046	074	Western Organization of Resource Councils and Coalition of 11 Other Environmental/Citizens Groups (DC)
II-D1.055	043	Coalition of 28 Environmental/Citizens Groups (varies)
II-D1.060	085	Oil & Gas Accountability Project and Coalition of 34 Other Environmental/Citizens Groups (CO)
II-D1.072	100	National Resources Defense Council (DC)
II-D1.101	139	San Juan Citizen's Alliance (CO)
II-D1.076; II-D2.001; II-D2.002	106 - 109	Kentucky Resources Council, Inc. (KY)
Private Citizens		
II-D1.004	033	Citizen (AK)
II-D1.050	031	Citizen (AL)
II-D1.012; II-D1.017	041; 048	Citizen (CA - 2)
II-D1.002; II-D1.003; II-D1.006; II-D1.008; II-D1.009; II-D1.011; II-D1.016; II-D1.018; II-D1.022; II-D1.023; II-D1.024; II-D1.026; II-D1.030; II-D1.031; II-D1.032; II-D1.034; II-D1.037; II-D1.038; II-D1.043; II-D1.044; II-D1.049; II-D1.058; II-D1.065; II-D1.067; II-D1.081; II-D1.083; II-D1.084; II-D1.085; II-D1.086; II-D1.087; II-D1.088; II-D1.089; II-D1.093; II-D1.095; II-D1.097; II-D1.099; II-D1.100; II-D1.102; II-D2.008	110; 032; 035; 037; 038; 040; 047; 049; 052; 053; 054; 056; 060; 061; 112; 128; 065; 066; 071; 072; 075; 083; 092; 094; 118; 120; 121; 122; 123; 124; 125; 126; 131; 133; 135; 137; 138; 140; 148	Citizen (CO - 39)
II-D1.015; II-D1.027; II-D1.029; II-D1.041; II-D1.098	046; 057; 059; 069; 136	Citizen (FL - 5)
II-D1.007	036	Citizen (KS)
II-D1.039; II-D1.048; II-D2.007	067; 030; 142	Citizen (MT - 3)
II-D1.005; II-D1.033; II-D1.051	034; 062; 076	Citizen (NM - 3)
II-D1.013; II-D1.019	044; 050	Citizen (NY - 2)
II-D1.042	070	Citizen (UT)
II-D1.028; II-D1.094	058; 132	Citizen (state unknown - 2)
State/Local/Federal Agencies		
II-D1.010	039	Sandia National Laboratories (NM)
II-D1.045	073	San Miguel County Board of Commissioners (CA)
II-D1.047	029	Alabama Oil and Gas Board (AL)
II-D1.057	082	State of New Mexico Energy, Minerals and Natural Resources Department (NM)
II-D1.059	084	Virginia Division of Gas and Oil (VA)
II-D1.061	086	Colorado Geological Survey (CO)
II-D1.062	087; 088	Michigan Department of Environmental Quality (MI)
II-D1.063	089	Pennsylvania Department of Conservation and Natural Resources (PA)
II-D1.064	090	State of Utah Department of Natural Resources, Division of Oil, Gas and Mining (UT)
II-D1.066	093	Alaska Oil and Gas Conservation Commission (AK)

TABLE 1: LIST OF PUBLIC COMMENTERS		
Docket ID[1]	**Edocket ID (OW-2001-0002)**[2]	**Organization (State)**
II-D1.068	095	State of South Dakota (SD)
II-D1.069	096	Ohio Department of Natural Resources (OH)
II-D1.073	101; 102	Conservation Division of the Kansas Corporation Commission (KS)
II-D1.079	116	State of Louisiana, Department of Natural Resources (LA)
II-D1.080	117	Colorado Oil & Gas Conservation Commission (CO)
II-D1.082	119	State of Missouri Department of Natural Resources, Geological Survey & Resource Assessment Division (MO)
II-D1.092	130	Indiana Department of Natural Resources, Division of Oil and Gas (IN)
II-D1.096	134	State of Oklahoma, Office of the Secretary of Energy (OK)
II-D1.103	147	Delta County Commissioners (CO)
II-D2.006	141	Office of Fossil Energy, Department of Energy (DC)
II-D2.009	149	Ohio Department of Natural Resources, Division of Mineral Resources Management (OH)
Oil and Gas Companies		
II-D1.070	097	Halliburton Energy Services (TX)
II-D1.075	105	Chevron Texaco North American Upstream (TX)
II-D1.090	127	Shell Exploration & Production Company (TX)
Trade Associations		
II-D1.035	113	Domestic Petroleum Council (DC)
II-D1.036	064	Independent Petroleum Association (DC)
II-D1.052	077	Interstate Oil and Gas Compact Commission (OK)
II-D1.053	080	Independent Oil & Gas Association of West Virginia (WV)
II-D1.054	042	Coalbed Methane Association of Alabama (AL)
II-D1.056	081	Oklahoma Independent Petroleum Association (OK)
II-D1.071	099	Ground Water Protection Council (OK)
II-D1.074	104	American Petroleum Institute (DC)
Other		
II-D1.020	051	Pace Law School (NY)
II-D1.021	111	University of Montana, Montana Bureau of Mines and Geology, Montana Tech (MT)
II-D1.077	114	Steven Harper, Attorney at Law (CO)
II-D1.078	129	Hansen Environmental Consultants (WA)

[1] Docket Identification numbers are assigned by the Water Docket in order to track each public comment with a unique identification number. Note that if a comment has a prefix of "II-D2," it indicates that the comment was received after the October 28, 2002 comment deadline. Comments with the following docket logs were updates, repeats, or clarifications of other comments: II-D1.91; II-D2.03; II-D2.04; and II-D2.05.

[2] An electronic version of each public comment is available through EPA's electronic public docket and comment system, EPA Dockets at http://www.epa.gov/edocket/. Each comment begins with the prefix "OW-2001-0002-". Edocket numbers were assigned to comment materials, as well as other relevant background documents in the order they were posted to the edocket Web site.

The remainder of this document contains summaries of the major public comments and EPA's responses related to the Agency's August 2002 report. The document is divided into seven other major sections as follows:

- **Section II: Scope of the Study** discusses public comments and EPA's responses on areas not included in the study, the literature used for the review, the number of coal basins included in the study, citizen complaints regarding water well contamination, and the peer review panel who reviewed the initial draft of the report.

- **Section III: Fracturing Fluids** describes public comments and EPA's responses related to the components of fracturing fluids, EPA's comparison of the concentration of fracturing fluid constituents to maximum contaminant levels (MCLs), EPA's estimates for the concentrations of fracturing fluid chemicals at the point-of-injection and the edge of the fracture zone, the amount of fracturing fluids that is recovered from CBM reservoirs, the amount of fracturing fluids used in hydraulic fracturing procedures, and the movement of "stranded" fluids in the coalbed formations.

- **Section IV: Fracture Behavior and Practices** discusses comments raised and EPA's responses to these comments regarding fracture growth, multiple fracturing of the same well, the relationship of drinking water wells to hydraulic fracturing activities, and differences in state geology.

- **Section V: Regulation of Hydraulic Fracturing Practices** describes comments and the Agency's responses regarding the states' authority over hydraulic fracturing practices, and the regulation of hydraulic fracturing under the Safe Drinking Water Act (SDWA).

- **Section VI: Language Used in the Report** summarizes specific comments and the Agency's responses related to the use of the term "USDW" in the report, use of scientific terms, and the tone of the language in the report.

- **Section VII: Chapter-Specific Comments** describes comments and the Agency's response regarding the glossary, executive summary, and Chapters 1 through 7 that were not already covered under Sections II through VI of this document.

- **Section VIII: Basin Descriptions** describes comments that pertain to the basin-specific descriptions in Attachments 1 through 11 of the report and EPA's response to these comments. The comments and responses in Section VIII do not include comments that were already discussed in Sections II through VII of this document.

II. SCOPE OF THE STUDY

A. Areas Not Included in the Review

1. Focus of the Report

Summary of Comments: One commenter indicated that the report should have focused on the possible impacts to human health instead of the hydraulic fracturing process. This commenter added that Chapter 4 of the report should have focused on dose-response curves and not on the properties of hydraulic fracturing fluids. The commenter also stated that EPA should have been able to conduct this analysis because the Agency should have access to research conducted on the toxicity of all constituents used in CBM production.

Another commenter stated that the study did not address the uncertainty in the risk assessment due to omissions and errors in the data used for the study. This commenter indicated that some of the reasons for these omissions and errors could be inadequate reporting by private well owners and counties, inadequate testing, and inadequate enforcement which would result in an underassessment of risk. This commenter also indicated that the report does not address risk resulting from deviations and failures in drilling, fracturing, and monitoring practices, especially for newer wells, or sufficiently address the testing error for volatile chemicals used in hydraulic fracturing.

EPA Response: The Phase I study was not intended to be a risk assessment, but rather, to be a fact-finding effort based primarily on existing literature to assess the potential threat to USDWs from the injection of hydraulic fracturing fluids into CBM wells and to determine based on these findings, whether additional study is warranted. The study is tightly focused on hydraulic fracturing of CBM wells and does not include other aspects of drilling or CBM production. EPA reviewed water quality incidents potentially associated with hydraulic fracturing, as well as evaluated the theoretical potential for hydraulic fracturing to affect USDWs. EPA researched over 200 peer-reviewed publications, interviewed approximately 50 employees from industry and state or local government agencies, and communicated with approximately 40 citizens and groups who are concerned that CBM production affected their drinking water wells.

For the purposes of this study, EPA assessed USDWs impacts by the presence or absence of documented drinking water well contamination cases caused by CBM hydraulic fracturing, clear and immediate contamination threats to drinking water wells from CBM hydraulic fracturing, and the potential for CBM hydraulic fracturing to result in USDW contamination based on two possible mechanisms described below.

1. Direct injection of fracturing fluids into a USDW in which the coal is located, or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. Creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

EPA's report includes a discussion of the types of fracturing fluids and additives, and fluid volumes that may be used in hydraulic fracturing operations. This discussion is intended to provide further background on the hydraulic fracturing process. In addition, the study provides a review of the fate and transport of injected fluids in the subsurface in order to determine whether a detailed risk assessment is warranted.

2. Monitoring

Summary of Comments: Several commenters questioned how EPA could decide whether hydraulic fracturing poses a risk to USDWs without collecting or reviewing monitoring data. Several commenters wanted EPA to proceed to Phase II of the study and to install monitoring wells in areas where hydraulic fracturing of CBM wells was occurring. One commenter recommended that, at a minimum, EPA identify whether any type of monitoring has been conducted by consulting firms, local or state agencies, or members of the academic community, and if this monitoring exists, to include the results in the report.

Another commenter recommended that EPA, in cooperation with the National Academy of Science (NAS), conduct unannounced inspections of hydraulic fracturing projects in order to collect samples of hydraulic fracturing fluids, and observe and measure the total volume of injected hydraulic fracturing fluid. This commenter also recommended that EPA establish reference doses (RfDs) and MCLs for all chemicals currently used in hydraulic fracturing fluids in significant volumes.

EPA Response: EPA has researched and reviewed a variety of monitoring information that may be related to the issue of possible conduits for fracturing fluid transport into USDWs. These data are discussed in Chapter 6 of the report. For example, EPA reviewed a 1999 Bureau of Land Management (BLM) report which focused on monitoring and data interpretation of methane concentrations in groundwater in the San Juan Basin area. EPA reviewed this report to determine if it contained information pertaining to hydraulic fracturing of CBM and its impacts, if any, to the quality of water in drinking water aquifers in this basin.

Chapter 6 of the report provides a detailed discussion of citizen complaints and state responses to their concerns. Complaints were responded to by various state agencies, and many of those responses included testing of water for contaminants. For example, the Virginia Department of Mines, Minerals and Energy is responsible for: responding to environmental issues associated with oil and gas development (including CBM); investigating all reported water problems; and testing water samples for contaminants that may be introduced by drilling (such as chlorides, oil and grease, and volatile organics).

EPA disagrees that monitoring data is needed to determine whether a Phase II study is warranted. As discussed in the previous response, EPA conducted an extensive literature review, conducted numerous interviews, reviewed water quality incidents potentially associated with hydraulic fracturing, and evaluated the theoretical potential for hydraulic fracturing to affect USDWs. EPA's decision that the injection of hydraulic fracturing fluids into CBM wells poses little or no threat to USDWs and does not justify additional study at this time is consistent with the process outlined in the April, 2001 Final Study Design. In its final study design, EPA indicated that the Agency would make a determination regarding whether further investigation was needed after analyzing the Phase I information.

EPA has recently taken a specific and important measure to address one of the primary concerns regarding hydraulic fracturing fluid – the use of diesel fuel. During EPA's research, the Agency realized that diesel is sometimes used a component of fracturing fluids and is of specific concern because it contains BTEX compounds (benzene, toluene, ethylbenzene, and xylenes) for which MCLs have been established under SDWA. Because of the potential problem diesel can cause, EPA requested its removal from hydraulic fracturing fluids. On December 15, 2003, EPA entered into a Memorandum of Agreement (MOA) with three major service companies – BJ Services Company, Halliburton Energy Services, Inc., and Schlumberger Technology Corporation – to voluntarily eliminate diesel fuel from hydraulic fracturing fluids that are injected directly into USDWs for CBM production. If necessary, these companies will select replacements that will not cause hydraulic fracturing fluids to endanger USDWs. Industry representatives estimate that these three companies conduct an estimated 95 percent of the hydraulic fracturing projects in the United States. These three have indicated to EPA that they no longer use diesel fuel as a hydraulic fracturing fluid additive when injecting into USDWs.

EPA, through its Underground Injection Control (UIC) Program, as authorized under SDWA Part C, Sections 1421-1426), is responsible for ensuring that fluids injected into the ground do not endanger USDWs or cause a public water system (PWS) to violate its drinking water standards due to the contamination of a USDW by these injected fluids. Most states have primary enforcement authority (primacy) for implementation of the UIC Program, and thus have the authority under SDWA to place controls on any injection activities that may threaten USDWs. 40 CFR 145.12, Requirements for Compliance Evaluation Programs, requires that authorized states have programs for periodic inspections of injection operations. States may also have additional authorities by which they can regulate hydraulic fracturing. While surprise inspections are not specifically mandated, state programs have a responsibility to conduct inspections, as necessary, to determine compliance with permit conditions, and to verify the accuracy of monitoring data and other information. EPA requires that all UIC inspectors be certified in, and that inspectors be knowledgeable about, proper operation of injection facilities, protection of USDWs, and SDWA requirements.

Regarding the establishment of RfDs and MCLs for all hydraulic fracturing fluid chemicals used in significant volumes, EPA follows an established procedure for identifying the contaminants for which these standards will be set. The Contaminant Candidate List (CCL) and the Unregulated Contaminant Monitoring Regulation (UCMR) are the primary review mechanisms by which EPA identifies drinking water contaminants which pose the most urgent threat to public health. The CCL process uses the best available information on contaminants of concern and emerging contaminants to prioritize according to potential public health threat, and identify candidates for possible regulation. The UCMR provides occurrence information for determining human exposure, establishing the baseline for health effects and economic analyses, contaminant co-occurrence analyses, and treatment technology evaluation (related to the CCL contaminants). After identifying the top priorities for regulatory determination, EPA begins the process of determining RfDs and associated enforceable standards for protection of public health.

3. Use of Modeling Results

<u>Summary of Comments:</u> One commenter recommended that EPA compare the results of hydraulic fracturing after the process to "modeling" conducted before the process to "provide some degree of predictability of the impact of the fracturing before the actual work is done."

This commenter also recommended that any modeling should consider the effect of other existing activities and conditions that could affect the outcome of the model (e.g., existing oil and gas wells, water wells, location and type of surface structures). This commenter also stated that consideration of the impact of these "man induced activities and conditions" should be an integral part of any fracture program and of any analysis of CBM fracturing impact. This commenter stated that the fracturing process and fluids alone may not cause "harm" within the study's parameters, but when coupled with the existing "man induced conditions" could cause "considerable damage and risk."

EPA Response: As discussed in Chapter 3 of the report, operators use a number of techniques to estimate fracture dimensions to design fracture stimulation treatments. Operators have a financial incentive to keep the hydraulically induced fracture generally within the target coal zone, so that expenditures on hydraulic horsepower, fracturing fluids, and proppants are minimized. For precise and statistically reliable measurements, however, fracture height and length can be measured (as opposed to modeled) accurately by microseismic monitoring. Tiltmeter measurements can also provide fracture height and length measurements somewhat accurately. The results of hydraulic fracturing "after the process" have also been investigated in the mined-through studies by the U.S. Bureau of Mines and others. These studies provide important, directly-measured characteristics of hydraulic fracturing in coal seams and surrounding strata. In addition, paint tracer studies conducted as part of mined-through studies can provide lower bound estimates on the extent of fluid movement.

During its analysis of the threat of CBM fracturing practices on USDWs, EPA considered the impact of human activities (such as improperly sealed or abandoned wells). Chapter 6 of the report summarizes citizen complaints and resulting investigations by state agencies into possible impacts of hydraulic fracturing on drinking water wells and surface waters. In some cases, improperly sealed gas wells have been remediated, resulting in decreased concentrations of methane in drinking water wells.

B. Literature Used for the Study

Summary of Comments: Some commenters indicated that the literature used for the study was outdated. Another commenter questioned whether the search terms that the Agency used to find references for the report would locate "health-related" literature. This commenter also questioned whether the acronym "USDW" and/or "underground sources of drinking water" was used as a search term. Another commenter stated that the report was "simply a compilation of existing data, with no new information, references, or conclusions."

EPA Response: The search terms used by the Agency did not include health-related terms because the study's goals did not include conducting a human-health risk assessment or conducting a new investigation into the toxicity of any of the components of hydraulic fracturing fluids.

As stated in the study design (66 FR 39396)[2], EPA focused the study on a review of existing data. EPA's literature search included publications and documents that were publically available as of December 2000/January 2001. EPA reviewed over 200 peer-reviewed publications. Much of the appropriate literature comes from the mid-1990s when funding was available for this kind of research. EPA also reviewed additional studies recommended by commenters and the peer review panelists, and incorporated information from these documents into the study, when appropriate. Further, EPA obtained information for the study through interviews with approximately 50 employees from industry and state or local government agencies, and communication with approximately 40 citizens and groups who are concerned that CBM production affected their drinking water wells.

C. Basins Included in the Study

Summary of Comments: One commenter questioned why EPA's report only included 11 basins. This commenter indicated that there are 16 separate basins considered to have CBM resources in the lower 48 states. Further, the commenter stated that the Illinois Basin, which was not discussed in the study, is a major coal-bearing region in the central Midwest.

EPA Response: EPA's literature search did not find any CBM activity or hydraulic fracturing in the Illinois Basin. Other basins which have little or no current CBM production activity (e.g., Alaska) were also omitted from the study.

D. Citizen Complaints/Instances of Water Well Contamination

Summary of Comments: Many commenters stated that EPA and state agencies have not done an adequate job of investigating citizen complaints related to contamination of water wells near hydraulically fractured CBM wells. Some commenters also stated that the Agency disregarded these complaints by concluding in its draft report that hydraulic fracturing of CBM wells poses a low risk. Some commenters also believed that the volume of complaints was enough to warrant the need for the Agency to continue its study. One commenter criticized the Agency for only having a 30-day collection period associated with the July 30, 2001 *Federal Register* notice in which the Agency requested information on groundwater contamination incidents that could be due to hydraulic fracturing of CBM wells. This commenter added that EPA's outreach efforts were unlikely to have reached the general public, and also recommended that EPA set up hotlines and make resources available to "allow immediate, comprehensive investigations of citizen complaints related to hydraulic fracturing impacts on USDWs."

Conversely, others commenters indicated that based on the volume of hydraulic fracturing activities, that if the threat to public health from hydraulic fracturing of CBM wells were significant, confirmed instances of water well contamination would exist. Some of these

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[2] US Environmental Protection Agency. 2001. Underground Injection Control; Request for Information of Ground Water Contamination Incidents Believed To Be Due to Hydraulic Fracturing of Coalbed Methane Wells. *Federal Register*. Vol. 66, No. 146. p. 39396, July 30, 2001.

commenters indicated that EPA's report should acknowledge the 1998 study conducted by the Ground Water Protection Council (GWPC), "Survey Results On Inventory and Extent of Hydraulic Fracturing in Coalbed Methane Wells in the Producing States," GWPC (December 15, 1998) because this survey of state oil and gas regulators provides further support for EPA's study conclusions.

EPA Response: The response of state agencies and EPA to citizen complaints are documented in Chapter 6. EPA has responded to complaints, particularly at the Regional level. For instance, in the Powder River Basin, located in Wyoming and Montana, citizen complaints dealt primarily with water quantity issues, which were beyond the scope of this study. EPA Region 8 is participating in a study that addresses the environmental effects of all aspects of CBM development and not just hydraulic fracturing. In response to citizen complaints, the Alabama Department of Environmental Management and EPA Region 4 also conducted independent sampling on wells in the Black Warrior Basin. Water analyses indicated that the wells had not been contaminated as a result of the hydraulic fracturing activities.

In some regions responses to citizen complaints are made primarily at the state level. For example, the Colorado Department of Health and the Colorado Oil and Gas Conservation Commission (COGCC) responds to many complaints. In Colorado, the primary response of the COGCC to citizen complaints has been the remediation of old, improperly sealed gas wells. The remediation of such wells has reduced methane concentrations in approximately 27 percent of the water wells sampled. Reduction of methane concentrations in many of the additional wells is expected over time due to the COGCC's efforts.

Regarding public outreach efforts need improvement, EPA has made considerable efforts to ensure its outreach and communications reach the general public. In addition to making the August 2002 draft available for public comments, EPA's outreach steps included:

- Publishing *Federal Register* notices (EPA's primary mechanism for communicating with the public):
    - requesting comment on how an EPA study should be structured (65 FR 45774)[3];
    - requesting information on any impacts to groundwater believed to be associated with hydraulic fracturing (66 FR 39396) (see footnote 2) including a mailing to over 200 county agencies making them aware of the *Federal Register* notice; and
    - requesting comments on the August 2002 draft of the study (67 FR 55249) (see footnote 1).

- Holding a public meeting on August 24, 2000, to obtain additional stakeholder input on the study. Several of these commenters recommended that EPA's study include accounts of personal experiences with regard to CBM impacts on drinking water wells. These experiences are discussed in Chapter 6.

[3] US Environmental Protection Agency. 2000. Underground Injection Control (UIC) Program; Proposed Coal Bed Methane (CBM) Study Design. *Federal Register*. Vol. 65, No. 143. p. 45774, July 25, 2000.

- Providing periodic updates for stakeholders, including citizens groups, in the form of written communication; and

- Maintaining a Web site where stakeholders can view the project documents; get updates on the progress of the project (including announcements of the release of *Federal Register* notices); and provide information to EPA.

Regarding the comment that EPA only provided 30 days for the public to provide information on CBM-related groundwater contamination incidents following the July 30, 2001 *Federal Register* notice, note that the Agency has considered all complaints received from the public, regardless of the time at which EPA received them. In addition, EPA's Web site www.epa.gov/safewater/uic/cbmstudy.html has a link to a form that allows people to submit information on the potential effects of hydraulic fracturing.

In response to the commenter's suggestion regarding hotlines, EPA has its Safe Drinking Water Hotline, which callers within the United States may reach at (800) 426-4791. Citizens are welcome to contact EPA or the states regarding these issues.

Regarding the comment about the volume of CBM activities and lack of confirmed instances of water well contamination, during its review, EPA found no confirmed cases that are linked to fracturing fluid injection into CBM wells or subsequent underground movement of fracturing fluids. Although thousands of CBM wells are fractured annually, EPA did not find confirmed evidence that drinking water wells have been contaminated by hydraulic fracturing fluid injection into CBM wells. EPA has included language to that effect in its final report, "Evaluation of Impacts to Underground Sources of Drinking Water by Hydraulic Fracturing of Coalbed Methane Reservoirs", June 2004, EPA document number: EPA 816-R-04-003 (hereafter referred to as final report).

E. Peer Review Panel

Summary of Comments: Many commenters questioned the composition of EPA's peer review panel, who reviewed the initial draft report. These commenters stated that this panel was heavily biased toward industry that has a stake in the outcome of the study. These commenters recommended that EPA convene a panel that is free of conflict of interest. Some recommended using members of the NAS as panelists.

One commenter indicated that he could not ascertain the composition of the panel although Appendix B of the report is supposed to contain a table with the list of the peer review panel. Another commenter stated that EPA made it very difficult for the public to obtain a copy of the peer review report, and that these comments were not attached in an appendix as originally promised.

EPA Response: EPA has a formal Agency Peer Review Policy that establishes the criteria and requirements for independent evaluation of scientific and technical studies and documents. Consistent with that policy, the Agency established a seven-member technical expert peer review panel, who performed a technical review of the study. Panel members were selected by identifying individuals with scientific or technical expertise in hydraulic fracturing through reviewing peer-reviewed publications in scientific journals and through communications with professional societies, trade and business associations, state organizations, and other federal agencies. EPA considered over 20 candidates before selecting 7 individuals based on their experience in the fields of hydraulic fracturing, rock mechanics, and/or natural gas production, and for their varying perspectives (industry, state government, academia, and a national laboratory). The charge to this committee was to review the report to determine if: 1) the report is complete, thorough, and accurate; and 2) the scientific/technical studies reviewed are applied in a sound, unbiased manner.

EPA posted the list of these reviewers and their qualifications on its Web site at www.epa.gov/safewater/uic/cbmstudy.html. EPA inadvertently omitted the table that identifies the peer reviewers in Appendix B of the draft report. This table is included in the final report.

III. FRACTURE FLUIDS

A. Components of Fracturing Fluids

1. Health Effects

Summary of Comments: Many commenters were concerned about the amount and health effects of certain chemicals used in hydraulic fracturing fluids and cited these concerns as reasons to continue the study. Some argued that very small quantities of toxic chemicals, such as benzene or methyl tert butyl ether (MTBE), could contaminate millions of gallons of groundwater.

Other commenters were concerned about the way in which the constituents of fracturing fluids and their potential health effects were presented in the draft report. For example, one commenter wanted the report to clearly convey the following: a wide variety of fracturing fluids exist, the health effects identified in the report apply to only some of the constituents that may or may not be present in the fracturing fluid, the health effects are associated with the product in its "pure form," and all the fluids additives are greatly diluted during fracturing operations.

EPA Response: As discussed in section II.A.2, EPA has recently entered into agreements with three major service companies to voluntarily eliminate diesel fuel from hydraulic fracturing fluids injected directly into USDWs for CBM production. Compounds such as benzene are components of diesel. These agreements will significantly reduce the use of diesel fuel in hydraulic fracturing fluids that are injected directly into USDWs for CBM production.

Chapter 4 of the final report provides a general description of the fate and transport processes which would minimize potential exposure to chemicals used in hydraulic fracturing fluids. Based on a 1991 fracturing fluid recovery study conducted in coal by Palmer et al., as much as 68 to 82 percent of the fracturing fluids may be removed when the methane is extracted.[4] This study is discussed in Chapter 3 of the report. As detailed in Chapter 4 of the report, the

[4] Palmer, I.D., Fryar, R.T., Tumino, K.A., and Puri, R. 1991. Comparison between gel-fracture and water-fracture stimulations in the Black Warrior basin; Proceedings 1991 Coalbed Methane Symposium, University of Alabama (Tuscaloosa), pp. 233-242).

unrecovered fluids will undergo processes that may limit their availability, concentration, and movement. These fluids may be significantly diluted and dispersed as they are transported through the subsurface. They may also interact chemically or physically with geologic material which may retard their movement and further disperse their concentrations.

EPA identified fluids and fluid additives commonly used in hydraulic fracturing through literature searches, reviews of relevant material safety data sheets (MSDSs) provided by service companies, and discussions with field engineers, service company chemists, and state and federal employees. The draft and final reports provide a discussion of the wide variety of hydraulic fracturing fluids that may be used. Table 4-1 of the report lists components that may be contained in fracturing fluids based on MSDSs. The final report emphasizes that not all fracturing fluid constituents, identified in Table 4-1 of this report, may be present in fracturing fluids, that the potential human health effects presented in the table apply to these compounds in their pure form, and that these compounds are significantly diluted prior to use.

An environmental impact statement (EIS) prepared by the BLM also identified MTBE as a compound that may be found in fracturing fluid (U.S. Department of the Interior, CO State BLM, 1998).[5] However, EPA was unable to find any indications in the literature, on MSDSs, or in interviews with service companies that MTBE is used in fracturing fluids to stimulate CBM wells.

2. Diesel Fuel

Summary of Comments: Several commenters supported EPA's recommendation that the industry use "water-based" alternatives in lieu of hazardous constituents such as diesel fuel. Some argued that EPA should make this a requirement and not a recommendation. Some of these commenters pointed to EPA's recommendation to "remove any threat whatsoever" from hydraulic fracturing fluid as a contradiction to the study's conclusions and as a reason to continue the study.

Conversely, several commenters indicated that there are valid reasons for using certain chemicals to enhance CBM production and that in choosing alternatives, the CBM well operators must take into account the specific geologic conditions of the site. These commenters recommended that EPA "encourage flexibility" with respect to the production of methane. One of these commenters noted that the draft report suggests that water-based alternatives are: currently available, feasible, and acceptable substitutes for diesel-based gels. This commenter indicated that the report findings should recognize that more research is needed on these potential alternatives. This commenter added that not all of the potential alternatives to the use of diesel may be water-based, citing polymer-based alternatives as one possibility. This commenter recommended that the term "water-based alternatives" be changed to read "non-diesel-based alternatives."

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[5] U.S. Department of the Interior, Bureau of Land Management, Colorado State Office. 1998. Glenwood Spring Resource Area: Oil & Gas Leasing Development, Draft Supplemental Environmental Impact Statement, June 1998.

One commenter indicated that in the State of Alabama, diesel is not used nor is it approved for hydraulic fracturing. The commenter added that service companies in his state primarily use a linear gel composed of guar gum, a surfactant, and silica.

EPA Response: The discussion of potential alternatives to the use of diesel is not included in the final report because it is outside the scope of the study. Instead, the report highlights the MOA with three major service companies to voluntarily eliminate the use of diesel fuel in hydraulic fracturing fluids injected directly into USDWs for the purpose of CBM production and if necessary, select replacements that will not cause hydraulic fracturing fluids to endanger USDWs (see the response to comment in section II.A.2).

Regarding the comment on the use of diesel in the State of Alabama, Table A2-1 in Attachment 2 of the draft and final report indicates that diesel is not used in that state.

3. MTBE

Summary of Comments: Several commenters were concerned about the use of MTBE in fracturing fluids. Many of them included the following statement in their comments: "only 28 tablespoons of MTBE could contaminate millions of gallons of groundwater."

One commenter indicated that the report contains several inconsistent statements regarding MTBE as a component of fracturing fluids. This commenter noted that in Chapter 4 of the draft report, EPA states that, based on its literature reviews and interviews with service companies, the Agency did not find any evidence that MTBE is used in fracturing fluids. This commenter also indicated that later in the same chapter, EPA states that "some gelling agents can contain hazardous substances including . . . [MTBE.]," and cites as its source a Supplemental EIS issued by BLM. This commenter provided arguments why he believed that the supplemental EIS was in error in listing MTBE as a potential component in fracturing fluids. This commenter further recommended that EPA should not have used this EIS as a source for identifying constituents in fracturing fluids or at a minimum, should have indicated the shortcomings associated with using this type of document to determine the components of fracturing fluids. This commenter provided a detailed discussion of some of the problems with using this particular EIS.

EPA Response: As stated in the response to comment in section III.A.1, an EIS prepared by the Colorado State BLM (1998) identified MTBE as a compound that may be found in fracturing fluid. EPA found no information in the literature, MSDSs, or through interviews with service companies indicating that MTBE is used in fracturing fluids to stimulate CBM wells. MTBE is not used during the manufacture of diesel fuel. It is generally only added to gasoline. However, in an effort to be fully inclusive of all the Agency's literature search findings, EPA included the information found in the EIS and noted that EPA was not able to confirm MTBE use in fracturing fluids.

B. Comparison of Concentrations of Hydraulic Fracturing Fluid Components to MCLs

Summary of Comments: A few commenters questioned the appropriateness of EPA's use of MCLs to compare the projected concentrations of fracturing fluids that may be injected into

USDWs. The commenters argued that MCLs apply to "treated water" and that the water associated with the formations in which hydraulic fracturing occurs would not be suitable for drinking water without first being treated.

EPA Response: Under the mandate of SDWA, EPA establishes MCLs as enforceable maximum permissible levels for contaminants in drinking water, to ensure the safety of public drinking water supplies. Because the concern about contamination relates to USDWs, which are actual or future supplies of drinking water for human consumption, MCLs are used in this study as standard reference points to compare calculated or anticipated levels of contaminants in hydraulic fracturing fluids and in the subsurface. MCLs provide a context for discussions regarding the concentrations of individual contaminants.

C. Concentrations of Constituents in Fracturing Fluids/Fluid Recovery Rates

1. Estimates of Concentrations of Constituents in Fracturing Fluids

Summary of Comments: EPA received several comments on its estimates of the concentrations of the constituents of concern in fracturing fluids that may be present at the point-of-injection and at the edge of the fracture zone. Many commenters were alarmed about the estimated concentrations of some of these constituents such as benzene because they were above the MCL. Further, some were concerned that EPA had revised its estimates since publication of the report. Conversely, other commenters indicated that EPA had overstated these concentrations. Each of these comments is discussed in more detail below.

One commenter indicated that EPA's estimates for the constituents of concern at the edge of the fracture zone, which assume a dilution factor of 30, still exceed drinking water standards for benzene, aromatics, 1-methylnapthalene, and methanol. This commenter added that EPA estimated high concentrations for the estimated point-of-injection for some chemicals for which drinking water standards have not yet been developed. This commenter acknowledged that these concentrations will be reduced as they mix with groundwater; however, he stated that very small amounts of some chemicals like benzene and MTBE can contaminate millions of liters of groundwater. Further, this commenter noted that most CBM wells are hydraulically fractured more than once, and therefore, "the groundwater in which it resides," will receive multiple doses of the fracturing fluids chemicals. The commenter stated a figure from the report that between 50,000 and 350,000 gallons of fracturing fluids are typically used in coalbed fracture treatments. Another commenter indicated that the report does not recognize that some of the constituents in fracturing fluids may affect human health at very low concentrations. This commenter added that with the potentially thousands of CBM wells being developed, the problem is magnified.

Several commenters claimed that EPA revised its calculations after the draft report was released. Some of these commenters indicated that EPA changed its scientific and policy conclusions under pressure from industry. One commenter provided detailed comments on the revised calculations. This commenter argued that EPA changed some of the parameters that were used in the draft report (such as length and height of a fracture, volume of injected hydraulic fracturing fluids, percentage of unrecovered hydraulic fracturing fluids) and they resulted in smaller estimated concentrations, including a revised estimate for benzene that does not exceed the MCL. This commenter questioned the basis for EPA's revising its estimates.

Other commenters were concerned that EPA did not adequately explain the assumptions used to generate its calculations. For example, one commenter indicated that it was unclear whether EPA based its estimates at the edge of the fracture zone on a specific fracture length or fracture radius. Some commenters also stated that EPA did not consider factors that would influence the availability and decrease the concentrations of the constituents at the edge of the fracture zone. These factors included: the recovery of the majority of the fracturing fluid, the relatively low permeability of coalbed formations will limit the movement of groundwater away from the wellbore, the coal will adsorb some of the constituents onto its surfaces, acids react with certain rock constituents and become spent, and some fracturing fluid constituents such as benzene will biodegrade. Some commenters also recommended that EPA's report should further emphasize that any constituents of concern in fracturing fluids are present only in very minimal amounts.

One commenter indicated that EPA had "significantly mischaracterized the nature of its estimates at both the point-of-injection and the edge of the fracture zone" because EPA had used a "worst case" scenario for estimating these concentrations. The commenter stated that, although the report indicates that EPA used mid-range values, the Agency used the maximum amount of diesel fuel that service companies reported to EPA instead of an average value. This commenter also explained why he believed that some of the point-of-injection concentrations that were presented in Table 4-2 of the draft report, such as that estimated for methanol, appeared to be inconsistent with the discussion in the text. Further, this commenter also recommended that EPA include its newer calculations in the report.

<u>EPA Response:</u> The values presented in the draft report are oversimplified estimates based on dilution alone and are not accurate enough to predict that a 30 times decrease is above or below the MCL. In the final report, EPA has revised its procedure for assessing the potential effect of fracturing fluid constituents on USDWs from that presented in the August 2002 draft as follows:

- The draft report included point-of-injection calculations for all constituents that may be contained in fracturing fluids. The final report focuses only on those constituents for which MCLs are established (i.e., BTEX compounds).

- EPA has revised the fraction of BTEX compounds in diesel used to estimate the point-of-injection concentrations from a single value to a documented broader range of values for the fraction of BTEX in diesel fuel. For example, the fraction of benzene in diesel was revised from $0.00006 g_{benzene}/g_{diesel}$ to a range with a minimum value of $0.000026\ g_{benzene}/g_{diesel}$ and a maximum value of $0.001\ g_{benzene}/g_{diesel}$. If the maximum value for benzene in diesel is used to estimate the concentration of benzene at the point-of-injection, the resulting estimate is 17 times higher than that presented in the draft report.

- In the final report, EPA used more current values for two of the parameters used to estimate the point-of-injection concentrations of BTEX compounds. Specifically, the estimates in this report use a density of the diesel fuel-gel mixture of 0.87 g/mL compared to 0.84 g/mL in the draft report, and a fraction of diesel fuel in gel of 0.60 g_{diesel}/g_{gel} compared to 0.52 g_{diesel}/g_{gel} in the draft report. The use of these more current values does not affect the order of magnitude of the revised point-of-injection calculations.

- The August 2002 draft report included estimates of the concentration of benzene at an idealized, hypothetical edge of the fracture zone located 100 feet from the point-of-injection. Based on new information and stakeholder input, EPA concluded that the edge of fracture zone calculation is not an appropriate model for reasons including:

  - Mined-through studies reviewed by EPA indicated that hydraulic fracturing injection fluids had traveled several hundred feet beyond the point-of-injection.

  - The assumption of well-mixed concentrations within the idealized fracture zone is insufficient. One mined-through study indicated an observed concentration of gel in a fracture that was 15 times the injected concentration, with gel found to be hanging in stringy clumps in many fractures. The variability in gel distribution in hydraulic fractures indicates that the gel constituents are unlikely to be well mixed in groundwater.

  - Based on more extensive review of the literature, the width of a typical fracture was estimated to be much thinner than that used in the draft report (0.1 inch versus 2 inches). The impact of the reduced width of a typical fracture is that the calculated volume of fluid that can fit within a fracture is less. After an initial volume calculation using the new width, EPA found that the volume of the space within the fracture area may not hold the volume of fluid pumped into the ground during a typical fracturing event. Therefore, EPA assumes that a greater volume of fracturing fluid must "leakoff" to intersecting smaller fractures than what was assumed in the draft report, or that fluid may move beyond the idealized, hypothetical "edge of fracture zone." This assumption is supported by field observations in mined-through studies, which indicate that fracturing fluids often take a stair-step transport path through the natural fracture system.

- In the draft report, EPA approximated the edge of fracture zone concentrations considering only dilution. Based on new information and stakeholder input on the draft report, EPA does not provide estimates of concentrations beyond the point-of-injection in the final report. Developing such concentration values with the precision required to compare them to MCLs would require the collection of significant amounts of site-specific data. This data in turn would be used to perform a formal risk assessment, considering numerous fate and transport scenarios. These activities are beyond the scope of Phase I of this study.

- In Chapter 4 of the final report, EPA provides a qualitative evaluation of the fate and transport of unrecovered fracturing fluids on residual concentrations of BTEX in groundwater. EPA describes in Chapter 4 how subsurface flow would significantly disperse and dilute BTEX compounds in groundwater, minimizing potential exposure to these constituents. BTEX compounds may also interact chemically or physically with geologic material which may retard their movement and further disperse their concentrations.

See also EPA's response to comment in section III.A.1 of this document.

No data or conclusions in the final report or in any previous draft were altered to accommodate any industry parties, states, environmental groups, or others. This study was a thorough and transparent data collection and technical evaluation exercise. The report and its conclusions were prepared by career technical staff at EPA.

The study was designed based upon a transparent process including public comment on the conceptual study design which included comments from state drinking water and oil and gas agencies, industry, environmental groups, and private citizens. EPA consulted with experts in the United States Geological Survey and the Department of Energy. Consistent with principles of good science, a draft of the study was subjected to a technical peer review from hydraulic fracturing experts. The conclusions of the study were not submitted for review to any private sector parties.

2. Fluid Recovery Rates

Summary of Comments: Many commenters were concerned that a large percentage of fracturing fluid remains behind and is available to potentially migrate into USDWs, citing these concerns as a reason to continue EPA's study. Some commenters indicated that EPA was inconsistent in the recovery percentages that the Agency cited in the report. Two commenters noted that the recovery experiment that is referenced in the report only ran for 19 days and that additional fracturing fluids may be recovered after that time. Another commenter stated that one fluid recovery rate (i.e., 61 percent) should not be "indiscriminately applied to over 14,000 CBM wells."

Some commenters cited a study by three Amoco scientists in which the study found "that a significant volume of fracturing fluids is not withdrawn." These commenters explained that the scientists found that the gelling agents used in the fracturing fluids remained in the coal samples although they had been flushed with water and strong acids. The commenters argued that, since these chemicals are not fully recovered, they could "serve as continuous sources of groundwater contamination."

EPA Response: Section III.A.1 provides a discussion of processes that can limit the availability, concentration, and movement through groundwater of unrecovered fracturing fluids. EPA has ensured that the recovery percentages cited in the report are both internally consistent and consistent with the literature reviewed. Three studies on recovery rates of hydraulic fracturing fluids were reviewed in Chapter 3 of the report. Only one of these studies, Palmer et al., 1991, involved hydraulic fracturing of coalbeds (refer to footnote 1 for the study reference). Thus, the Palmer study was considered the most relevant of the three studies for the purposes of this report. The final report clarifies that the recovery rate of 61 percent was based on a 19-day flowback period. Palmer et al., 1991, predicted recovery rates as high as 82 percent over a longer recovery period.

Regarding the study by three Amoco scientists, EPA contacted one of the commenters to obtain a copy of the study to review.[6] The commenter was unable to provide the study and EPA's additional library research efforts were also unsuccessful at obtaining this study.

3. Amount of Fracturing Fluids

Summary of Comments: Some commenters were concerned about the volume of fracturing fluids used in a "typical fracturing job" and cited the following statement from the report, "Coalbed fracture treatments typically use 50,000 to 350,000 gallons of various fracturing fluids, and from 75,000 to 320,000 pounds of sand as proppant... ." Others questioned the accuracy of the quantities of fracturing fluid and proppant cited in the report, stating that these figures were more consistent with a massive hydraulic fracture. Another commenter stated that the unique properties that make many coal formations effective receptacles for methane also allow them to hold large quantities of water. This commenter stated that injection of hydraulic fracturing fluids into USDWs risks permanent contamination of these USDWs because fracturing fluids often contain large amounts of toxic chemicals.

EPA Response: EPA has clarified in the final report that more typical injection volume may be closer to a maximum of 150,000 gal/well, and a median value of 57,500 gal/well. These values are based on average injection volume data provided by Halliburton for six CBM locations.

Refer to section III.A.1 regarding factors that would influence the availability, concentration, and movement of fracturing fluids and their constituents.

4. Movement of Fracturing Fluids

Summary of Comments: Some commenters stated that unrecovered fracturing fluids will flow toward the well because of the pressure gradients. Others noted that this was only true while the well was in production. These commenters argued that once pumping stops, the aquifer will attempt to resume a normal flow pattern and the remaining hydraulic fracturing fluids will move freely within the coalbed formation.

EPA Response: Chapter 4 of the final report has been expanded to more clearly explain:

- hydraulic gradients that occur during injection versus those during fluid recovery;
- the significance of the capture zone of the production well on fracturing fluid recovery (i.e., the portion of the aquifer that contributes water to the well); and
- the movement of fracturing fluids (and what influences their movement) both inside and outside the capture zone.

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[6] Puri, R., G.E. King, and I.D. Palmer, 1991, "Damage to Coal Permeability During Hydraulic Fracturing," Society of Petroleum Engineers Proceedings from Rocky Mountain Regional Meeting and Low-Permeability Reservoirs Symposium, Denver, CO, p. 109-115, (SPE #21813).

IV. FRACTURE BEHAVIOR AND PRACTICES

A. Fracture Growth

Summary of Comments: EPA received many comments on the statements in its report that, "Vertical fracture heights in coalbeds have been measured in excess of 500 feet and lengths can reportedly reach up to 1,500 feet." Some of these commenters stated that these distances indicate the potential for communication with and contamination of USDWs. Other commenters believed that these measurements were incorrect. Some commenters also discussed whether confining layers act as barriers to vertical fracture growth.

One commenter described in detail why he believed that confining layers above and below the hydraulically fractured coal formations would also be fractured and permeated by fluids. This commenter noted that the fracture heights cited in the report exceed the thickness of the thickest coal formations identified in the report. In addition, this commenter noted that the report indicates that some of the coal seams are bounded by sandstone and conglomerate (which have different lithological properties, and therefore different fracturing properties, than shale). Further, he indicated that the report supports his position that the risk for migration of fracturing fluids into adjacent USDWs is significant because it indicates that "Stimulation fluids in coal penetrate from 50 to 100 feet away from the fracture and into the surrounding formation. In these and other cases, when stimulation ceases and production resumes, these chemicals may not be completely recovered and pumped back to the CBM well, and, if mobile, may be available to migrate through an aquifer." This commenter also noted that the report shows that many of the coal formations are located in mountainous regions such as the Rocky Mountains and Appalachian Mountains. The commenter stated that the rock formations in these regions, including the coal formations, have been subjected to intense orogenic and tectonic stress resulting in regional, systematic fractures and faults. The commenter argued that it is likely that coal formations, and other rocks above and below them, are characterized by cracks and fractures, and that because of these deformation features, rates of groundwater transport tend to be higher.

One commenter indicated that the report's description of how fractures travel is incorrect (i.e., they travel horizontally vs. vertically). This commenter added that there is some vertical expansion as the fracture moves horizontally but that this is not the primary direction of fracturing. This commenter stated that their state geologists estimate vertical fracture heights at 50 to 60 feet. Another commenter provided detailed comments on the studies that were conducted on fracture height growths. This commenter indicated that he had been involved in numerous fracture experiments (in all types of reservoirs) where the fracture height has actually been measured (using microseismic or downhole tiltmeter), as well as in mineback tests where hydraulic fractures have been excavated. Based on his experience, the fracture height has always been less than or equal to the height that would be predicted by just using stresses in the various layers (which the commenter indicated was the only factor considered in all the references used in the draft report). The commenter reported that in some cases, the differences were factors of two or three. This commenter also provided detail on factors that influence fracture height growth, such as horizontal stress in the coal, the horizontal stress in the surrounding layers, the characteristics of the layering, and the type of hydraulic fracturing fluids being pumped.

Another commenter noted that the discussion on fracture dimensions in the report was based on literature from 1993 and earlier, but acknowledged that there were "virtually no post-1993 published reports on hydraulic fracturing." The commenter recommended that EPA contact operators, service companies, and state regulatory agencies for current practices and models. Further, this commenter noted that newer data based on more sophisticated FracPro models are available for many basins. He added that, in his state, model results indicate that fracture height is "generally less than 100 feet, whereas fracture half length is typically between 150 and 700 feet." This commenter also noted that the report should state that the fracture heights have been "modeled" not "measured" because vertical fracture heights have never been fully measured in the field.

EPA Response: EPA has revised Chapter 3 to provide clarification on the characterization of fracturing behavior during hydraulic stimulations. The statement that fractures have been "measured in excess of 500 feet and lengths can reach up to 1,500 feet" has been removed because it refers to modeled estimates, rather than direct measurements. Instead, the results of 22 mined-through studies have been summarized, because they provide direct measurements of the dimensions of hydraulic fractures, as well as lower bounds on the extent of fracturing fluid movement. Chapter 3 has also been revised to better distinguish between fracture characterizations based on modeling vs. those that are directly measured.

In addition, EPA has revised Chapter 3 to clarify the issue of hydraulic barriers and barriers to fracture growth above coalbeds. EPA agrees with the commenter that when shales overlying targeted coals are extensively fractured, they may not act as barriers to hydraulic fracture growth or as hydraulic barriers. On the other hand, thick, relatively unfractured shale may present a barrier to upward fracture growth because of the stress contrast between the coalbed and the overlying shale. Deep vertical fractures can propagate vertically to shallower depths and develop a horizontal component. In the formation of these "T-fractures," the fracture tip may fill with coal fines or intercept a zone of stress contrast, causing the fracture to turn and develop horizontally, sometimes at the contact of the coalbed and an overlying formation.

B. Multiple Fractures

Summary of Comments: Some commenters raised concern over the statement in the draft report that "each well, over its lifetime is fractured several times" and urged EPA to continue to Phase II of the study. Others questioned the accuracy of EPA's statement that wells are fractured multiple times. One commenter indicated that in their state, most wells have not been re-fractured multiple times but that instead, two to four coal groups were generally fractured in each well.

EPA Response: EPA has revised the statements regarding multiple stimulations in Chapter 3. In the draft report EPA stated that "many coalbeds are refractured at sometime after the initial treatment." The text has been revised to indicate that the literature on refracturing that was reviewed pertains only to the Black Warrior Basin. EPA's extensive literature review did not find any information indicating that wells are fractured multiple times in any basin other than the Black Warrior Basin.

C. Relationship of Drinking Water Wells to Hydraulic Fracturing Activities

Summary of Comments: Some commenters were concerned about the potential for fracturing fluids to contaminate USDWs due to the high occurrence of coal reservoirs within USDWs. One commenter cited a statement from the report "if coalbeds are located within USDWs, then any fracturing fluids injected into coalbeds have the potential to contaminate the USDW." The commenter added that the report indicates that as much as 91 percent of U.S. coal reservoirs may be located within USDWs.

Two commenters indicated that hydraulic fracturing activities take place at depths far below groundwater sources used as drinking water sources. One of these commenters added that his company's records show that it conducts hydraulic fracturing at shallow depths, (i.e., less than 300 feet below ground surface), in less than one percent of all hydraulic fracturing jobs. This commenter provided this as one reason that he believed that hydraulic fracturing is unlikely to pose a threat to drinking water.

EPA Response: EPA found that 10 of the 11 coal basins, included in the study, may lie, at least in part, within USDWs. Given the concerns associated with the use of diesel fuel and the introduction of BTEX constituents into USDWs, EPA negotiated an MOA with three major hydraulic fracturing service companies for the voluntary elimination of diesel fuel in hydraulic fracturing fluids injected directly into USDWs for the purpose of CBM production. Nevertheless, even when fracturing fluids are injected directly into coalbeds located in USDWs, fracturing fluid components are likely to be significantly diluted and dispersed, as well as subject to other fate and transport processes (discussed in Chapter 4) which are likely to lower their concentrations or prevent their mobility underground. Also see the response to comment in section III.A.1.

D. Differences in State Geology

Summary of Comments: Several commenters indicated that the report did not adequately address the variability present in the different geologic formations that are subject to hydraulic fracturing, and therefore, did not address the possible impacts associated with that variability regarding regional groundwater flow and/or the occurrence and distribution of CBM resources, on assessing the potential threat of hydraulic fracturing on USDWs. One commenter indicated that to accurately represent the threats to USDWs, risk levels should be "differentiated based on modeling and actual data on similar geologic conditions."

EPA Response: EPA agrees that variability of geologic formations and regional groundwater flow are key to the assessment and understanding of the potential threat to USDWs posed by hydraulic fracturing. The study findings and conclusions are based on literature from each of the 11 major coal basins in the United States. In addition, the draft and final report contains separate attachments which discuss basin-specific geologic and hydrogeologic investigations related to each of the 11 basins. The discussions provided were intended to characterize regional coal basin methane production with respect to its effect on USDWs and to supplement the generalized information provided within the body of the report. EPA also agrees that if modeling risk levels, the variability of geologic conditions should be considered. However, such a modeling exercise is beyond the scope of the current study.

V. REGULATION OF HYDRAULIC FRACTURING PRACTICES

A. States' Authority

Summary of Comments: Several commenters recommended that EPA expand its discussion in the final report of the states' role in regulating hydraulic fracturing. Others suggested clarifying the language from the draft report regarding states' authority to regulate hydraulic fracturing. For example, one commenter indicated that EPA's statement, "States with primacy for their UIC program enforce and have the authority to place controls on any injection activities that may threaten USDW's" implies that state UIC Programs can or would regulate hydraulic fracturing. The commenter recommended that EPA add clarifying language that removes the implication that hydraulic fracturing is commonly regulated under UIC Programs.

One commenter stated that the report was inaccurate in its description of Virginia's authority to place restrictions on the depth at which hydraulic fracturing can occur. The commenter indicated that the "restrictions" are instead voluntary procedures. The commenter also clarified the purpose of these procedures.

EPA Response: EPA did not conduct a systematic review of state regulations of hydraulic fracturing and, therefore, has no basis for expanding its discussion of the state's role in the regulation of hydraulic fracturing. However, the Agency added clarifying language regarding the state's ability to regulate hydraulic fracturing. EPA also added clarifying wording to the report regarding Virginia's voluntary program.

B. Regulation of Hydraulic Fracturing under SDWA

Summary of Comments: Several commenters wanted EPA to regulate hydraulic fracturing of CBM wells under SDWA and did not believe that recommended measures such as using "water-based alternatives" instead of diesel were sufficient. One commenter stated that based on *Legal Environmental Assistance Foundation, Inc. v. U.S. E.P.A.*, 118 F.3d 1467, 1470 (11th Cir. 1997), EPA is to decide how to regulate hydraulic fracturing under SDWA, and not to determine whether "further investigation was necessary to evaluate any potential threats" before EPA acts. Another commenter was concerned whether EPA was using the presence of documented cases of "health harm from non-regulation" as the criterion for determining whether to regulate hydraulic fracturing injection activities under SDWA. This commenter argued that the purpose of the UIC Program is "to forestall and prevent such harm by isolating the injected fluids from aquifers that are or could be developed as USDWs"; and therefore, using proven harm as a regulatory threshold goes against the purpose and intent of the law.

Conversely, other commenters indicated that EPA should "recognize the need for industry to be allowed reasonable flexibility in the means that its uses to produce CBM." These commenters also indicated that under 42 U.S.C. § 300h(b)(2), Congress intended that EPA not impose restrictions through the UIC Program that interfere with or impede activities related to oil and gas development unless such restrictions are essential for preventing endangerment of drinking water sources. Another commenter specifically recommended that UIC permits not be required for hydraulic fracturing practices.

EPA Response: Based on the information collected and reviewed, EPA has determined that the injection of hydraulic fracturing fluids into CBM wells poses little or no threat to USDWs. Continued investigation under a Phase II study is not warranted at this time. The lack of confirmed incidents of drinking water well contamination due to hydraulic fracturing fluid injection from past hydraulic fracturing activities was one among many factors EPA considered. If threats to USDWs from hydraulic fracturing of CBM wells were significant, EPA would expect to have found confirmed instances of drinking water well contamination from the practice. Although thousands of CBM wells are fractured annually, EPA did not find confirmed evidence that drinking water wells have been contaminated by the injection of hydraulic fracturing fluids into CBM wells.

EPA's recent agreements with three major service companies, discussed in section II.A.2, will significantly reduce the use of diesel fuel in hydraulic fracturing fluids that are injected directly into USDWs for CBM production.

It is important to note that states with primary enforcement authority (primacy) for their UIC Programs implement and enforce their regulations, and have the authority under SDWA to place additional controls on any injection activities that may threaten USDWs. States may also have additional authorities by which they can regulate hydraulic fracturing. With the expected increase in CBM production, the Agency is committed to working with states to monitor this issue.

VI. LANGUAGE USED IN THE REPORT

A. Use of the Term "USDW"

Summary of Comments: Some commenters indicated that EPA used the term "USDW" too broadly. In particular, one commenter indicated that the report "carelessly utilizes the USDW term in the context of hydrocarbon bearing formations." This commenter added that these hydrocarbon-bearing aquifers subjected to hydraulic fracturing are unlikely to be used for drinking water, especially without treatment for two reasons: 1) the high total dissolved solids level of the waters in these formations; 2) the waters in these formations may be considered an "exempted aquifer" under SDWA because the aquifer is mineral, hydrocarbon, or geothermal energy producing, or can be demonstrated to be commercially producible. This commenter also stated that the inferences in the report, that some risks may be attributed to hydraulic fracturing, conflict with "the reality that such a formation would not be used for water supply without treatment, if it were ever to be used."

EPA Response: EPA disagrees that it has applied the term "USDW" too broadly in the report. SDWA mandates the protection of USDWs from injection activities – "if such injection may result in the presence in underground water which supplies or can reasonably be expected to supply any PWS of any contaminant, and if the presence of such contaminant may result in such system's not complying with any national primary drinking water regulation or may otherwise adversely affect the health of persons." The broad definition of a USDW by Congress was to ensure that future USDWs would be protected, even where those aquifers were not currently used as a drinking water source or could not be used without some form of water treatment such as desalination. It is also important to note that an exempted aquifer is a USDW, but is exempt from regulation.

B. Use of Scientific Terms

Summary of Comments: A few commenters provided corrections to some of the terminology used in the report. One commenter felt that there was a general misuse of geologic terminology in the report, and specifically indicated that the geologic terms "system," "formation," and "seams" should not have been used interchangeably. This commenter provided other specific clarifications or corrections to some of the discussions in the report (e.g., Section 3.1 regarding the depositional history of coal-bearing rocks in the United States).

EPA Response: EPA appreciates the careful review of the report by many of the commenters. EPA has revised some of the terminology used in the report and incorporated some of the clarifications suggested by the commenters.

C. Use of Qualifying Language

Summary of Comments: Both the commenters that supported EPA's conclusions and those who opposed it indicated that the tone of the language used throughout the report conflicted with EPA's conclusions. Commenters cited examples of this language that included the following:

- "Based on the information collected, the potential threats to USDWs posed by hydraulic fracturing ***appear to be low*** and do not justify additional study.";
- ..."the ***apparent risk*** to public health from hydraulic fracturing is not compelling enough to warrant expending resources on a phase II effort"; and
- "***the apparent threat*** to public health from hydraulic fracturing."

One of the commenters indicated that this language showed "a weak articulation of EPA's confidence in its own report." Many of the commenters who were opposed to EPA's findings, pointed to EPA's qualified statements as a reason to continue the study.

Another commenter, who supported EPA's findings, stated that the primary definition of the word, "apparent," is, "something that is clearly seen or understood, obvious, self-evident, glaring." This commenter, among others who supported the Agency's findings, recommended that EPA replace all uses of the word "apparent" when describing the threat posed to USDWs by hydraulic fracturing with words that more accurately describe the low likelihood of this threat.

EPA Response: In the final report, EPA has eliminated the use of the word "apparent" and "appears" to describe its study conclusions and has made the language more consistent with the report's results.

VII. CHAPTER-SPECIFIC COMMENTS

This summary of chapter-specific comments focuses mainly on those comments that have not been summarized within the issue-specific Sections II through VI of this document. Comments were received on almost every chapter of the document, ranging from minor editorial suggestions, to factual corrections. EPA appreciates the thorough comments that were submitted regarding the contents of the hydraulic fracturing report. The Agency has considered all comments, researched the accuracy of some comments (where necessary), and incorporated comments where appropriate.

A. Glossary

Summary of Comments: One commenter submitted recommended changes to the list of acronyms and abbreviations, and the glossary pertaining to "M"; "KCl"; "pad"; and the phrase, "wells that have been 'screened-out' cannot be used for gas production."

EPA Response: After reviewing and checking on the accuracy of the above comments, EPA incorporated changes to the glossary and list of acronyms, where appropriate.

B. Other Executive Summary Comments

Summary of Comments: EPA received many comments that were specific to the executive summary of the report, including recommendations for revising the text, tables, and figures. A few commenters suggested that the language regarding the findings and conclusions of the study needs to be clearer and stronger (e.g., qualifiers such as "appear to be low" and "persuasive evidence" weakens the conclusions). Another suggested that, in general, the executive summary and the main document need to point out that not all USDWs are currently being used nor will they ever be used as sources of drinking water. Some commenters felt that the executive summary was inappropriately long and provided suggestions for making the section shorter, including eliminating all tables from this section. Many commenters provided specific editorial comments.

A few commenters expressed concern regarding the "graphic language" in Table ES-2 *(Summary of MSDSs for Hydraulic Fracturing Fluid Additives)* used to describe the health effects of fracturing fluids, and noted that they felt it may be unnecessarily alarming, and potentially misleading to readers (i.e., it does not clarify that the health effects only pertain to some constituents that may or may not be present in the fracturing fluids). Commenters added that Table ES-2 suggests that linear gel delivery systems always contain diesel and does not indicate that fluid additives are greatly diluted. One commenter felt that the information provided in Table ES-4 *(Evidence in Support of Coal-USDW Co-Location in U.S. Coal Basins)* was too general, and believed that the information should just be presented in the more detailed sections from which it was summarized. Other commenters were concerned that the information provided in Table ES-5 *(Summary of Reported Incidents that Associate Water Quality/Quantity with Coalbed Methane (CBM) Activity)* could be misleading to the public.

One commenter felt that the executive summary figures in general were "confusing and misleading." Other commenters questioned the accuracy and clarity of Figure ES-2 *(Graphical Representation of the Hydraulic Fracturing Process in Coalbed Methane Wells)*, which depicts drinking water wells drawing down into coal seams. One commenter questioned the accuracy of the illustrations in Figures ES-3 *(Direct Fluid Injection into a USDW (Coal within USDW))* and ES-4 *(Fracture Creates Connection to USDW)* regarding the depth of the water wells and the direction of fluid migration (i.e., fracturing fluids are shown to be flowing away from the well bore toward the drinking water wells). The commenter pointed out that the descriptive text on page ES-10 conflicts with the depiction of fluid migration in Figure ES-4.

EPA Response: EPA has reviewed and considered all comments regarding the executive summary of the document. The Agency originally designed the executive summary to be a stand-alone document. Because many readers of such a document (such as Congress or the leaders of various stakeholder organizations) may have limited time to dedicate to the review of a large technical document, EPA included essential summary information, including tables and figures, in the executive summary. However, based on the comments received, EPA has pared down the executive summary by taking out most of the tables and summarizing key information from these tables in narrative form. EPA incorporated many of the specific suggestions related to the figures (e.g., decreasing the depth of drinking water wells), and in some instances, provided clarifying language to explain the figures.

C. Other Chapter 1 Comments (Introduction)

Summary of Comments: A few commenters provided comments regarding the Introduction to the hydraulic fracturing report. Comments included questions about the accuracy of the figures, and how they were depicted: groundwater flow; the relation between well depths and coal seams; and the point-of-injection for the fracturing fluids. One commenter objected to the statement that the study was "based on a high level of interest of stakeholders..." when it was the commenters' understanding that it was based only on a "handful" of complaints.

EPA Response: The statement that the study was "based on a high level of interest of stakeholders..." is an accurate statement but the term "stakeholders" was vague. To be more descriptive, Chapter 1 of the final report indicates that a reason for conducting the study was "concerns voiced by individuals who may be affected by coalbed methane development. . ." The Agency addressed each of the other comments by either incorporating suggested language or making relevant clarifications in the document language and figures.

D. Other Chapter 2 Comments (Methodology)

No substantive comments received on this chapter.

E. Other Chapter 3 Comments (Characteristics of CBM Production and HF Practices)

Summary of Comments: EPA received several comments regarding the information in Chapter 3. In particular, several commenters questioned the study's assumptions regarding recovery rates and fracture heights. A more detailed summary of the comments received on these topics can be found in sections III.C.2 and IV.A, respectively. One commenter had several specific questions regarding statements made in this chapter, including: the meaning of the term "conventional coal mines"; statements regarding the number of CBM wells in Alabama; the discussion of the origin of CBM; the statement that "coal has very little natural permeability"; contradictions between the discussion of fluids migration in this chapter compared to that shown in Figures ES-4 and 1-3; accuracy and clarity of statements regarding the rate of fluid recovery; and the statement that many CBM wells are re-fractured.

EPA Response: EPA appreciates the detailed comments that were submitted regarding Chapter 3 of the hydraulic fracturing report. The Agency made several editorial corrections and clarifications to this chapter based on these comments. A more detailed response regarding recovery rates, fracture heights, and re-fracturing of the same wells can be found in sections III.C.2, IV.A, and IV.B, respectively.

F. Other Chapter 4 Comments (HF Fluids)

Summary of Comments: Comments specific to Chapter 4 of the report included questions about the calculation of the constituents of concern at the point-of-injection, and other editorial comments and suggestions.

EPA Response: In response to comments received on Chapter 4, EPA has incorporated clarifying language regarding its calculations of BTEX compounds at the point-of-injection. Other editorial corrections and clarifications have also been incorporated. For a discussion of how EPA revised its procedure for assessing the potential effect of fracturing fluid constituents on USDWs from that presented in the draft report, refer to section III.C.1.

G. Other Chapter 5 Comments (Basin Descriptions)

Summary of Comments: Several comments were received regarding the basin descriptions, including updates from a few states on the numbers of wells in the applicable basins. One commenter suggested additional references that should be used to correct some of the statements regarding the Pottsville Formation. The other four commenters each provided specific editorial suggestions on one of the following four basins: the Central Appalachian Basin, the Northern Appalachian Basin, the Uinta Basin, and the Powder River Basin.

EPA Response: EPA has incorporated the updated well information provided by states. All other editorial comments were considered, and most were incorporated. Other basin-specific issues are discussed in section VIII of this document.

H. Other Chapter 6 Comments (Water Quality Incidents)

Summary of Comments: Several comments were received regarding the water quality incidents chapter of the report. Commenters made specific editorial suggestions, and provided clarifications about specific complaints, additional information about how their state investigates complaints, and information about state-specific hydraulic fracturing regulations. One commenter stated that the discussion of the Pottsville, Allegheny, Conemaugh, and Monongahela Groups were "oversimplified" and questioned the conflicting use of the terms "cyclothem" and "complex" when describing the depositional environments of the Allegheny Group.

A few commenters expressed concern that the descriptions of public complaints (including the information summarized in Table 6-2) are presented in the report as if the information was factual, without linking the complaints to actual findings following the state and EPA investigations. One commenter indicated that EPA does not present any data from state agencies, which suggests to the commenter that no real scientific studies were conducted. Commenters recommended that the complaints be immediately followed by a summary of the evaluation and resolution of the complaint. One recommendation was that, if kept in the report, the information be moved to an appendix.

Finally, some commenters felt that EPA was contradictory regarding the question of whether hydraulic fracturing of CBM wells threaten USDWs. For example, one commenter indicated that EPA had concluded in Chapter 6 that there is insufficient evidence to determine if there is a link between fracturing and USDW contamination. However, elsewhere in Chapter 6, EPA states that "water quality problems might be associated with some of the variety of production activities common to CBM extraction. These production activities include... methane migration through conduits created by drilling and fracturing practices..."

<u>EPA Response:</u> In response to stakeholder's comments on EPA's original study methodology, EPA compiled citizen complaints and reported incidences of CBM impacts on drinking water wells and included these accounts in Chapter 6 of the report. In the final report, EPA has clarified the rationale for including citizen complaints in its report.

The final report also clarifies that many of the reported impacts (such as impacts to water supply quantities and effects of discharge of groundwater extracted in the CBM production process) included in Chapter 6 are outside of the scope of SDWA and beyond the scope of the Phase I study. The goal of the Phase I study was to assess the potential for contamination of USDWs due to the injection of hydraulic fracturing fluids into CBM wells, and to determine based on these findings if further study is warranted. EPA also incorporated information that was provided by states regarding incident reports, and state-specific regulations. Finally, the Agency took Table 6-2 out of the document because, as suggested by some commenters, summarizing citizen complaints in a tabular format oversimplified this information, and created a potential for misinterpretation. The information in Table 6-2 is presented in detail in the main body of Chapter 6.

See also EPA's response to comment in section II.D of this document regarding other issues pertaining to water contamination incidents and citizen complaints.

I. Other Chapter 7 Comments (Conclusions and Recommendations)

<u>Summary of Comments:</u> Most comments received regarding Chapter 7 of the report also relate back to prior report chapters. Several commenters had specific suggestions or questions regarding the conclusions and recommendations section of the report. Some of these commenters agreed with the conclusions of the study, but recommended that EPA put more emphasis on the conclusions, and include information about the findings of the study earlier in the document. Specifically, commenters suggested that, at the beginning of the document, EPA include a statement clarifying that: "EPA finds no evidence of harm from hydraulic fracturing

while investigating the reported incidents that spurred the study." These commenters felt that EPA's findings that Phase II of the study is unnecessary, and that little or no public health threat is posed by hydraulic fracturing should be more strongly stated in the conclusions of the report.

Note that commenter opinions regarding Chapter 7 of the report do not reflect the overall commenter perspectives regarding the outcome and conclusions of the study. Most of the commenters expressed opinions regarding the study's conclusions, but did not state them within the context of Chapter 7.

EPA Response: EPA has reviewed all commenter suggestions regarding Chapter 7, and incorporated the majority of these comments where appropriate. Other revisions to Chapter 7, which relate back to changes in previous chapters, have been made in order to ensure internal consistency within the document.

VIII. BASIN DESCRIPTIONS

This summary of basin-specific comments focuses mainly on those comments that have not been summarized within the issue-specific Sections II through VI of this document. Many comments were received that provided minor editorial suggestions and factual corrections regarding basin descriptions. The Agency has considered all comments, researched the accuracy of some comments (where necessary), and incorporated public comments where appropriate.

A. San Juan Basin

Summary of Comments: One commenter provided suggested edits and corrections pertaining to the San Juan Basin geology, hydrology and USDW identification, and CBM production activity. This commenter also provided additional references.

EPA Response: EPA reviewed and considered all suggested edits and corrections and has incorporated revisions to the San Juan Basin descriptions. EPA also reviewed the additional references provided by the commenter, and incorporated additional pertinent information.

B. Black Warrior Basin

Summary of Comments: One commenter provided a variety of editorial comments and factual clarifications regarding the Black Warrior Basin. Examples of information the commenter questioned include: coal thickness; total dissolved solids levels; number of active Class II wells in this area; fracture height vs. length; and chemical components of fracturing fluids.

EPA Response: EPA has incorporated into the final report the majority of the commenter's suggestions regarding the description of the Black Warrior Basin.

C. Piceance Basin

Summary of Comments: One commenter provided a brief description of the activities and progress of the pilot program in the White River Dome field.

EPA Response: The final report contains the information provided by the commenter.

D. Uinta Basin

Summary of Comments: One commenter indicated that the information on the Castlegate Field is out of date. The commenter clarified that the field is currently in production, and explained why he believes that cross-contamination from the Blackhawk to the Castlegate Sandstone and Star Point Sandstone (as indicated in the report) is unlikely.

EPA Response: EPA has made revisions to the basin description based on this information.

E. Powder River Basin

Summary of Comments: *No substantive comments were submitted on this section.*

F. Central Appalachian Basin

Summary of Comments: One commenter provided clarifications and corrections regarding CBM activity, regulations, and drinking water sources in Virginia.

EPA Response: EPA has incorporated many of the commenter's clarifications into the basin description.

G. Northern Appalachian Basin

Summary of Comments: One commenter provided information on the square mileage and number of CBM wells in this basin, with associated references. This commenter, who is the individual that was interviewed for some of the information provided in this attachment, provided edits to the interview summary. Another commenter suggested several editorial corrections pertaining to the location of specific coal groups, the use of the term "group," and the use of the term "separated laterally" vs. "vertical separation."

EPA Response: EPA has incorporated all appropriate information into the basin description.

H. Western Interior Basin

Summary of Comments: This commenter questioned the accuracy of the statement that "coal seams could be coincident with a USDW" within the Cherokee Basin. The commenter discussed the aerial extent to which various coal seams in the Cherokee Basin coincide with USDWs, and recommended that EPA also review a 1997 paper entitled "Kansas coal resources and their potential for coalbed methane."

EPA Response: EPA has modified the report to indicate that "all or part of targeted coal seams could be coincident with a USDW," thereby clarifying the summary of the data provided in Table A8-2, which presents the relative depths of coal seams and USDWs.

I. Raton Basin

Summary of Comments: *No comments were submitted on this section.*

J. Sand Wash Basin

Summary of Comments: One commenter pointed out that in the Sand Wash Basin, the pilot at Craig Dome was abandoned "due to excessive water production." This commenter also believed that EPA's findings that hydraulic fracturing poses very little potential threat to USDWs does not account for proximity or overlap with natural fault lines. The commenter stated that: "if a fracture propagates into and along a fault plane, it may contaminate a USDW."

EPA Response: EPA has incorporated the commenter's information into Attachment 10 of the final report.

K. Washington Coal Regions (Pacific and Central)

Summary of Comments: *No comments were submitted on this section.*

REFERENCES

This master reference list pertains only to Chapters 1 through 7 of this document. Separate reference lists are provided for each appendix and attachment, and are provided at the end of each of these sections.

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Appendix A
Department of Energy - Hydraulic Fracturing White Paper

1.0 Introduction

The first hydraulic fracturing treatment was pumped in 1947 on a gas well operated by Pan American Petroleum Corporation in the Hugoton field.[1] The Kelpper Well No. 1, located in Grant County, Kansas was a low productivity well, even though it had been acidized. The well was chosen for the first hydraulic fracture stimulation treatment so that hydraulic fracturing could be compared directly to acidizing. Since that first treatment in 1947, hydraulic fracturing has become a standard treatment for stimulating the productivity of oil and gas wells.

Hydraulic fracturing is the process of pumping a fluid into a wellbore at an injection rate that is too high for the formation to accept in a radial flow pattern. As the resistance to flow in the formation increases, the pressure in the wellbore increases to a value that exceeds the breakdown pressure of the formation that is open to the wellbore. Once the formation "breaks-down", a crack or fracture is formed, and the injected fluid begins moving down the fracture. In most formations, a single, vertical fracture is created that propagates in two directions from the wellbore. These fracture "wings" are 180^o apart, and are normally assumed to be identical in shape and size at any point in time. In naturally fractured or cleated formations, such as gas shales or coal seams, it is possible that multiple fractures can be created and propagated during a hydraulic fracture treatment.

Fluid that does not contain any propping agent, often called "pad", is injected to create a fracture that grows up, out and down, and creates a fracture that is wide enough to accept a propping agent. The purpose of the propping agent is to "prop open" the fracture once the pumping operation ceases, the pressure in the fracture decreases, and the fracture closes. In deep reservoirs, we use man-made ceramic beads to prop open the fracture. In shallow reservoirs, sand is normally used as the propping agent. The sand used as a propping agent in shallow reservoirs, such as coal seams, is mined from certain quarries in the United States. The silica sand is a natural product and will not lead to any environmental concerns that would affect the United States Drinking Water (USDW).

The purposes of this paper are (1) to discuss the processes an engineer uses to design and pump a hydraulic fracture treatment, and (2) to provide an overview of the theories, design methods and materials used in a hydraulic fracture treatment. Currently, a discussion is taking place on the effects of hydraulic fracturing in coal seams on the USDW. Gas production from coal seams is increasing in importance in the United States. In 2000, over 6% of the natural gas production in the US was produced from coal seams, and that percentage will increase in the future. Because of the ever-increasing importance of natural gas production from coal seams, coal seam examples have been included in this technical paper.

Objectives of Hydraulic Fracturing

In general, hydraulic fracture treatments are used to increase the productivity index of a producing well, or the injectivity index of an injection well. The productivity index defines the volumes of oil or gas that can be produced at a given pressure differential between the reservoir and the well bore. The injectivity index refers to how much fluid can be injected into an injection well at a given pressure differential.

There are many different applications for hydraulic fracturing, such as:

- Increase the flow rate of oil and/or gas from low permeability reservoirs,
- Increase the flow rate of oil and/or gas from wells that have been damaged,
- Connect the natural fractures and/or cleats in a formation to the wellbore,
- Decrease the pressure drop around the well to minimize sand production,
- Decrease the pressure drop around the well to minimize problems with asphaltine and/or paraffin deposition,
- Increase the area of drainage or the amount of formation in contact with the wellbore, and
- Connect the full vertical extent of a reservoir to a slanted or horizontal well.

Obviously, there could be other uses of hydraulic fracturing, but the majority of the treatments are pumped for these seven reasons.

A low permeability reservoir is one that has a high resistance to fluid flow. In many formations, chemical and/or physical processes alter a reservoir rock over geologic time. Sometimes, these diagenetic processes restrict the openings in the rock and reduce the ability of fluids to flow through the rock. Low permeability rocks are normally excellent candidates for stimulation by hydraulic fracturing.

Regardless of the permeability, a reservoir rock can be damaged when a well is drilled through the reservoir and when casing is set and cemented in place. Damage occurs because drilling and/or completion fluids leak into the reservoir and plug up the pores and pore throats. When the pores are plugged, the permeability is reduced, and the fluid flow in this damaged portion of the reservoir may be substantially reduced. Damage can be severe in naturally fractured reservoirs, like coal seams. To stimulate damaged reservoirs, a short, conductive hydraulic fracture is often the desired solution. As such, hydraulic fracturing works very well in many damaged, coal seam reservoirs.

In many cases, especially for low permeability formations, damaged reservoirs and horizontal wells in a layered reservoir, the well would be "uneconomic" unless a successful hydraulic fracture treatment is designed and pumped. Thus, the engineer in charge of the economic success of such a well, must (1) design the optimal fracture treatment, and then (2) go to the field to be certain the optimal treatment is pumped successfully.

Candidate Selection

The success or failure of a hydraulic fracture treatment often depends on the quality of the candidate well selected for the treatment. Choosing an excellent candidate for stimulation often ensures success, while choosing a poor candidate will normally result in economic failure. To select the best candidate for stimulation, the design engineer must consider many variables. The most critical parameters for hydraulic fracturing are formation permeability, the *in-situ* stress distribution, reservoir fluid viscosity, skin factor, reservoir pressure, reservoir depth and the condition of the wellbore. The skin factor refers to whether the reservoir is already stimulated or, perhaps is damaged. If the skin factor is positive, the reservoir is damaged and could possibly be an excellent candidate for stimulation.

The best candidate wells for hydraulic fracturing treatments will have a substantial volume of oil and gas in place, and will have a need to increase the productivity index. Such reservoirs will have (1) a thick pay zone, (2) medium to high pressure, (3) *in-situ* stress barriers to minimize vertical height growth, and (4) either be a low permeability zone or a zone that has been damaged (high skin factor). For coalbed methane reservoirs, the ideal candidate, in addition to the 4 factors listed above, will be a thick coal seam containing both (1) a large volume of sorbed gas and (2) abundant coal cleats to provide permeability.

Reservoirs that are not good candidates for hydraulic fracturing are those with little oil or gas in place due to thin reservoirs, low reservoir pressure, or small aerial extent. Reservoirs with extremely low permeability may not produce enough hydrocarbons to pay all the drilling and completion costs even if successfully stimulated; thus, such reservoirs would not be good candidates for stimulation. In coal seam reservoirs, the number, thickness and location of the coal seams must be considered when deciding if the coals can be completed and stimulated economically. If the coal seams are too thin or too scattered up and down the hole, the coals may not be ideal candidates for stimulation by hydraulic fracturing.

Developing Data Sets

For most petroleum engineering problems, developing a complete and accurate data set is often the most time consuming part of solving the problem. For hydraulic fracture treatment design, the data required to run both the fracture design model and the reservoir simulation model can be divided into two groups. One group lists the data that can be "controlled" by the engineer. The second group reflects data that must be measured or estimated, but cannot be controlled.

The primary data that can be controlled by the engineer are the well completion details, treatment volume, pad volume, injection rate, fracture fluid viscosity, fracture fluid density, fluid loss additives, propping agent type, and propping agent volume. The data that must be measured or estimated by the design engineer are formation depth, formation permeability, *in-situ* stresses in the pay zone, *in-situ* stresses in the surrounding layers, formation modulus, reservoir pressure, formation porosity, formation compressibility, and the thickness of the reservoir. There are actually three (3) thickness that are important to the design engineer: the gross thickness of the reservoir; the net thickness of the oil or gas producing interval; and the permeable thickness that will accept fluid loss during the hydraulic fracture treatment.

The most critical data for the design of a fracture treatment are, roughly in order of importance, (1) the *in-situ* stress profile, (2) formation permeability, (3) fluid loss characteristics, (4) total fluid volume pumped, (5) propping agent type and amount, (6) pad volume, (7) fracture fluid viscosity, (8) injection rate, and (9) formation modulus. Since most engineers have more work to do than time to do the work, the design engineer should focus most of his/her time on the most important parameters. In hydraulic fracture treatment design, by far, the two most important parameters are the *in-situ* stress profile and the permeability profile of the zone to be stimulated and the layers of rock above and below the target zone.

In new fields or reservoirs, most operating companies are normally willing to spend money to run logs, cut cores and run well tests to determine important factors such as the *in-situ* stress and the permeability of the major reservoir layers. By using such data, along with fracture treatment records and production records, accurate data sets for a given reservoir in a given field can normally be compiled. These data sets can be used on subsequent wells to optimize the fracture treatment designs. It is normally not practical to cut cores and run well tests on every well. Thus, the data obtained from cores and well tests must be correlated to log parameters so the logs on subsequent wells can be used to compile accurate data sets.

To design a fracture treatment, most engineers use pseudo 3-dimensional (P3D) models. Full 3-D models exist; however, the use of full 3-D models is currently limited to supercomputers and research organizations. To use a P3D model, the data must be input by reservoir layer. Fig. 1 illustrates the profiles of important input data required by a P3D model. For the situation in

Fig. 1, the fracture treatment would be initiated in the sandstone reservoir. The fracture would typically grow up and down until a barrier is reached to prevent vertical fracture growth. In many cases, thick marine shale will be a barrier to vertical fracture growth. In some cases, coal seams will prevent fractures from growing vertically. Many coal seams are highly cleated, and when the fracture fluid enters the coal seam, it remains contained within the coal seam. In thick, highly cleated coal seams, the growth of the hydraulic fracture will normally be limited to the coal seam.

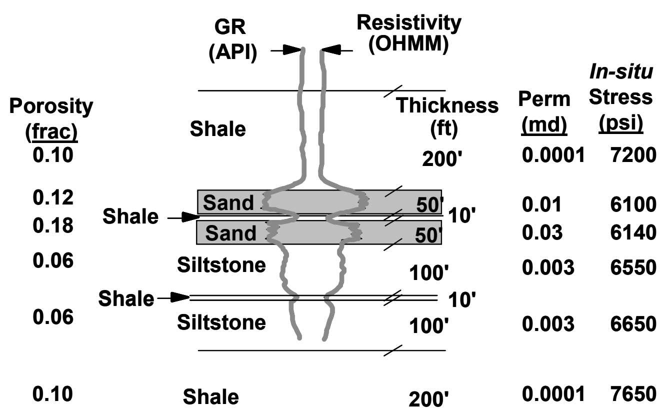

Fig. 1 – Typical input data for a P3D model.

The data used to design a fracture treatment can be obtained from a number of sources, such as drilling records, completion records, well files, open hole geophysical logs, cores and core analyses, well tests, production data, geologic records, and other public records, such as publications. In addition, service companies provide data on their fluids, additives and propping agents. Table 1 illustrates typical data needed to design a fracture treatment and possible sources for the data.

Table 1 – Sources of Data

Data	Units	Sources
Formation Permeability	md	Cores, Well Tests, Correlations, Production Data
Formation Porosity	%	Cores, Logs
Reservoir Pressure	psi	Well Tests, Well Files, Regional Data
Formation Modulus	psi	Cores, Logs, Correlations
Formation Compressibility	psi	Cores, Logs, Correlations
Poisson's Ratio		Cores, Logs, Correlations
Formation Depth	ft	Logs, Drilling Records
In-situ Stress	psi	Well Tests, Logs, Correlations
Formation Temperature	°F	Logs, Well Tests, Correlations
Fracture Toughness	psi - $\sqrt{\text{in}}$	Cores, Correlations
Water Saturation	%	Logs, Cores
Net Pay Thickness	Ft	Logs, Cores
Gross Pay Thickness	Ft	Logs, Cores, Drilling Records
Formation Lithology		Cores, Drilling Records, Logs, Geologic Records
Wellbore Completion		Well Files, Completion Prognosis
Fracture Fluids		Service Company Information
Fracture Proppants		Service Company Information

<u>Fracture Treatment Optimization</u>

The goal of every design engineer should be to design the optimum fracture treatment for each and every well. In 1978, Holditch *et al.*[2] wrote a paper concerning the optimization of both the propped fracture length and the drainage area (well spacing) for low permeability gas reservoirs. Fig. 2 illustrates the methodology used to optimize the size of a fracture treatment [3,4]. Fig. 2 clearly shows the following:

- As the propped length of a fracture increases, the cumulative production will increase, and the revenue from hydrocarbon sales will increase,
- As the fracture length increases, the incremental benefit ($ of revenue per foot of additional propped fracture length) decreases,
- As the treatment volume increases, the propped fracture length increases,
- As the fracture length increases, the incremental cost of each foot of fracture ($ of cost per foot of additional propped fracture length) increases, and

- When the incremental cost of the treatment is compared to the incremental benefit of increasing the treatment volume, an optimum propped fracture length can be found for every situation.

Additional economic calculations can be made to determine the optimum fracture treatment design. However, in all cases, the design engineer must consider the effect of the fracture upon flow rates and recovery, the cost of the treatment, and the investment guidelines of the operator of the well.

Field Considerations

After the optimum fracture treatment has been designed, it must be pumped into the well successfully. A successful field operations requires planning, coordination and cooperation of all parties. Treatment supervision and the use of quality control measures will improve the successful application of hydraulic fracturing. Safety is always the primary concern in the field. Safety begins with a thorough understanding by all parties on their duties in the field. A safety meeting is always held to review the treatment procedure, establish a chain of command, be sure everyone knows his/her job responsibilities for the day, and to establish a plan for emergencies. The safety meeting should also be used to discuss the well completion details and the maximum allowing injection rate and pressures, as well as the maximum pressures to be held as backup to an annulus. All casing, tubing, wellheads, valves, and weak links, such as liner tops, should be thoroughly tested prior to rigging up the fracturing equipment. Mechanical failures during a treatment can be costly and dangerous. All mechanical problems should be repaired prior to pumping the fracture treatment.

Prior to pumping the treatment, the engineer-in-charge should conduct a detailed inventory of all the equipment and materials on location. The inventory should be compared to the design and the prognosis. After the treatment has concluded, the engineer should conduct another inventory of all the materials left on location. In most cases, the difference in the two inventories can be used to verify what was mixed and pumped into the wellbore and the hydrocarbon bearing formation.

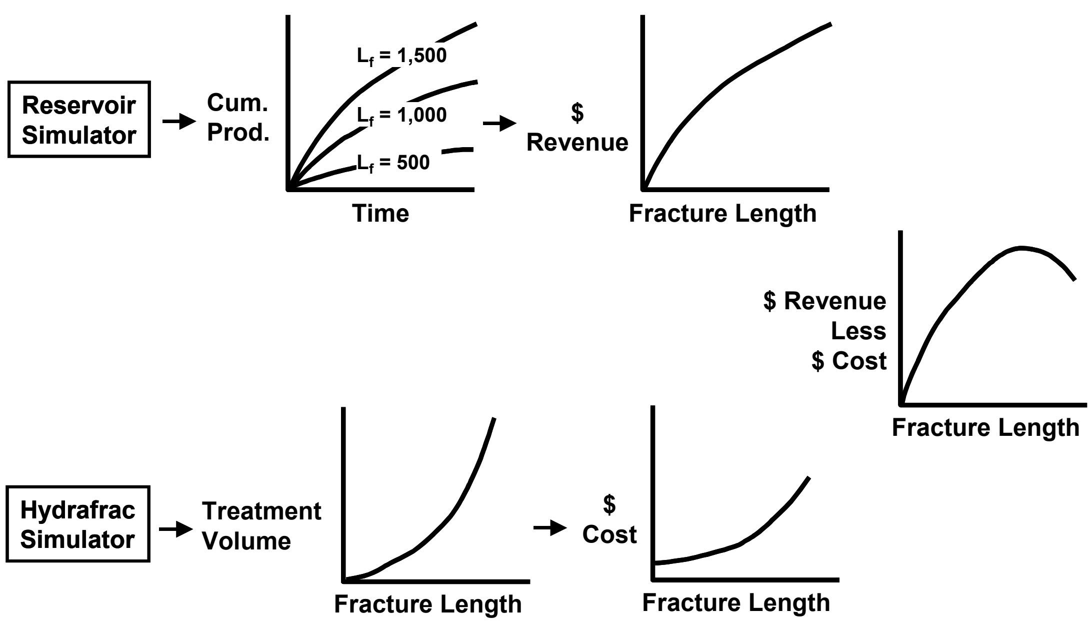

Fig. 2 – Fracture treatment optimization process.

In addition to an inventory, samples of the base fracturing fluid (usually water) should be taken and analyzed. Typically, a water analysis is done on the base fluid to determine the minerals present and the type of bacteria in the water. The data from the water analysis can be used to select the additives required to mix the viscous fracture fluid required to create a wide fracture and to transport the propping agent into the fracture. Table 2 shows the typical compositions for mix waters used in different fracturing situations. In addition to testing the water, samples of the additives used during a treatment and the fracture fluid after all additives have been added should be taken during the job and saved for future analyses, if required.

Table 2 – Fracturing Fluids and Conditions for Their Use

Base Fluid	Fluid Type	Main Composition	Used For
Water Based	Linear Fluids	Gelled Water, GUAR< HPG, HEC, CMHPG	Short Fractures, Low Temperatures
	Crosslinked Fluids	Crosslinker + GUAR, HPG, CMHPG, CMHEC	Long Fractures, High Temperatures
Foam Based	Water Based Foam	Water and Foamer + N_2 or CO_2	Low Pressure Formations
	Acid Based Foam	Acid and Foamer + N_2	Low Pressures, Water Sensitive Formations
	Alcohol Based Foam	Methanol and Foamer + N_2	Low Pressure Formations With Water Blocking Problems
Oil Based	Linear Fluids	Oil, Gelled Oil	Water Sensitive Formations, Short Fractures
	Crosslinked Fluids	Phosphate Ester Gels	Water Sensitive Formations, Long Fractures
	Water External Emulsions	Water + Oil + Emulsifier	Good For Fluid Loss Control

Formation temperature is one of the main factors concerning the type of additives required to mix the optimum fracturing fluid. In deep, hot reservoirs (>250°F), more additives are required than in shallow, low temperature reservoirs. Since most coal seams are very shallow, fewer additives are normally required to mix the optimum fracture fluid.

2.0 Fracture Mechanics

Fracture mechanics has been part of mining engineering and mechanical engineering for hundreds of years. No one is more interested in underground rock fractures than a miner working in an underground mine. In petroleum engineering, we have only used fracture mechanics theories in our work for about 50 years. Much of what we use in hydraulic fracturing theory and design has been developed by other engineering disciplines many years ago. However, certain aspects, such as poroelastic theory, are unique to porous, permeable underground formations. The most important parameters are *in-situ* stress, Poisson's ration, and Young's modulus.

In-situ Stresses

Underground formations are confined and under stress. Fig. 3 illustrates the local stress state at depth for an element of formation. The stresses can be divided into 3 principal stresses. In Fig. 3, σ_1 is the vertical stress, σ_2 is the maximum horizontal stress, while σ_3 is the minimum horizontal stress, where $\sigma_1 > \sigma_2 > \sigma_3$. This is a typical configuration for coalbed methane reservoirs. However, depending on geologic conditions, the vertical stress could also be the intermediate (σ_2) or minimum stress (σ_3). These stresses are normally compressive and vary in magnitude throughout the reservoir, particularly in the vertical direction (from layer to layer). The magnitude and direction of the principal stresses are important because they control the pressure required to create and propagate a fracture, the shape and vertical extent of the fracture, the direction of the fracture, and the stresses trying to crush and/or embed the propping agent during production.

A hydraulic fracture will propagate perpendicular to the minimum principal stress (σ_3). If the minimum horizontal stress is σ_3, the fracture will be vertical and, we can compute the minimum horizontal stress profile with depth using Eq. 1.

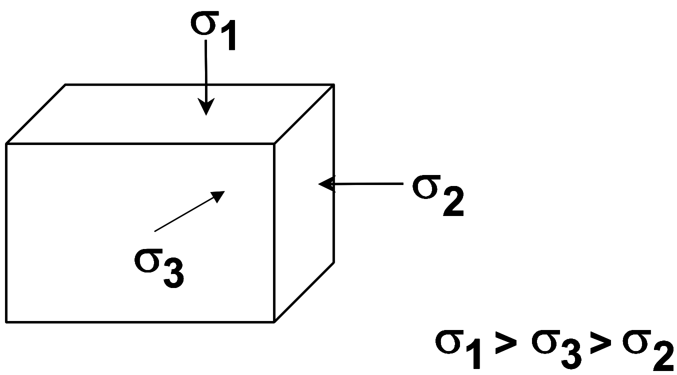

Fig. 3 – Local in-situ stress at depth.

$$\sigma_{min} \cong \frac{\nu}{1-\nu}\left(\sigma_{ob} - \alpha\,\sigma_p\right) + \alpha\,\sigma_p + \sigma_{ext} \qquad \text{Eq. 1}$$

Where:

σ_{min}	=	the minimum horizontal stress (*in-situ* stress)
ν	=	Poissons' ratio
σ_{ob}	=	overburden stress
α	=	Biot's constant
σ_p	=	reservoir fluid pressure or pore pressure
σ_{ext}	=	tectonic stress

Poisson's ratio can be estimated from acoustic log data or from correlations based upon lithology. For coal seams, the value of Poisson's ratio will range from 0.2 – 0.4. The overburden stress can be computed using density log data. Normally, the value for overburden pressure is about 1.1 psi per foot of depth. The reservoir pressure must be measured or estimated. Biot's constant must be less than or equal to 1.0 and typically ranges from 0.5 to 1.0. The first two (2) terms on the right hand side of Eq.1 represent the horizontal stress resulting from the vertical stress and the poroelastic behavior of the formation. The tectonic stress term is important in many areas where plate tectonics or other forces increase the horizontal stresses.

Poroelastic theory can be used to determine the minimum horizontal stress in tectonically relaxed areas.[8,9] Poroelastic theory combines the equations of linear elastic stress-strain theory for solids with a term that includes the effects of fluid pressure in the pore space of the reservoir rocks. The fluid pressure acts equally in all directions as a stress on the formation material. The "effective stress" on the rock grains is computed using linear elastic stress-strain theory. Combining the two sources of stress results in the total stress on the formation, which is the stress that must be exceeded to initiate fracturing.

In many areas, however, the effects of tectonic activity must be included in the analyses of the total stresses. To measure the tectonic stresses, injection tests are conducted to measure the minimum horizontal stress. The measured stress is then compared to the stress calculated by the poroelastic equation to determine the value of the tectonic contribution.

<u>Basic Rock Mechanics</u>

In addition to the *in-situ* or minimum horizontal stress, other rock mechanical properties are important when designing a hydraulic fracture. Poisson's ratio is defined as "the ratio of lateral expansion to longitudinal contraction for a rock under a uniaxial stress condition".[10] The value of Poisson's ratio is used in Eq. 1 to convert the effective vertical stress component into an effective horizontal stress component. The effective stress is defined as the total stress minus the pore pressure.

The theory used to compute fracture dimensions is based upon linear elasticity. To apply this theory, the modulus of the formation is an important parameter. Young's modulus is defined as "the ratio of stress to strain for uniaxial stress".[10] The modulus of a material is a measure of the stiffness of the material. If the modulus is large, the material is stiff. In hydraulic fracturing, a stiff rock will result in more narrow fractures. If the modulus is low, the fractures will be wider. The modulus of a rock will be a function of the lithology, porosity, fluid type, and other variables. Table 3 illustrates typical ranges for modulus as a function of lithology.

Table 3. Typical Ranges of Young's Modulus for Various Lithologies

Lithology	Young's Modulus
Soft Sandstone	2-5 x 10^6 psi
Hard Sandstone	6-10 x 10^6 psi
Limestone	8-12 x 10^6 psi
Coal	0.1-1 x 10^6 psi
Shale	1-10 x 10^6 psi

Because coal is highly cleated, the modulus of the coal seam *in-situ* may be very low. In very low modulus, highly cleated coal seams, it is likely that most fractures will be wide and short, that is, not penetrating far into the formation from the well bore.

Fracture Orientation

A hydraulic fracture will propagate perpendicular to the least principle stress (Fig. 3). In some shallow formations the least principal stress is the overburden stress; thus, the hydraulic fracture will be horizontal. Nielsen and Hansen published a paper where horizontal fractures in coal seam reservoirs were documented [11]. In reservoirs deeper than 1000 ft or so, the least principal stress will likely be horizontal; thus, the hydraulic fracture will be vertical. The azimuth orientation of the vertical fracture will depend upon the azimuth of the minimum and maximum horizontal stresses. Lacy and Smith provided a detailed discussion of fracture azimuth in SPE Monograph 12.[12]

Injection Tests

The only reliable technique for measuring *in-situ* stress is by pumping into a reservoir, creating a fracture, and measuring the pressure at which the fracture closes [13]. The well tests used to measure the minimum principal stress are as follows: *in-situ* stress tests; step-rate/flow back tests; mini-fracture tests; and step-down tests. For most fracture treatments, mini-fracture tests and step-down tests are pumped ahead of the main fracture treatment. As such, accurate data are normally available to calibrate and interpret the pressures measured during a fracture treatment. *In-situ* stress tests and step-rate/flow back tests are not run on every well. However, it is common to run such tests in new fields or new reservoirs to help develop the correlations required to optimize fracture treatments for subsequent wells.

An *in-situ* stress test (or micro-frac) can be either an injection-falloff test or an injection-flow back test. The *in-situ* stress test is conducted using small volumes of fluid (a few barrels), injected at low injection rates (gals/min), normally using straddle packers to minimize well bore storage effects, into a small number of perforations (1-2 ft). The objective is to pump a thin fluid (water or nitrogen) at a rate barely sufficient to create a small fracture. Once the fracture is open, then the pumps are shut down, and the pressure is recorded and analyzed to determine when the fracture closes. Thus, fracture closure pressure is synonymous with *in-situ* stress and with minimum horizontal stress. When the pressure in the fracture is greater than the fracture closure pressure, the fracture is open. When the pressure in the fracture decreases below the fracture closure pressure, the fracture is closed. Fig. 4 illustrates a typical wellbore configuration for conducting an *in-situ* stress test. Fig. 5 shows typical data that are measured. Multiple tests are conducted to ensure repeatability. The data from any one of the injection-falloff tests can be analyzed to determine when the fracture closes.

Fig. 6 illustrates how one such test can be analyzed to determine *in-situ* stress.

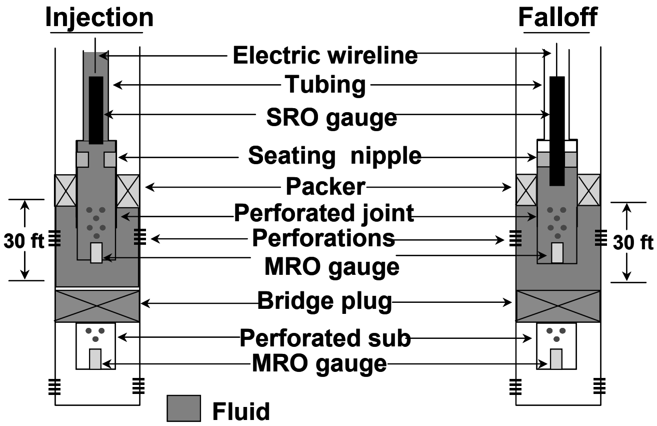

Fig. 4 – Cased hole test configuration.

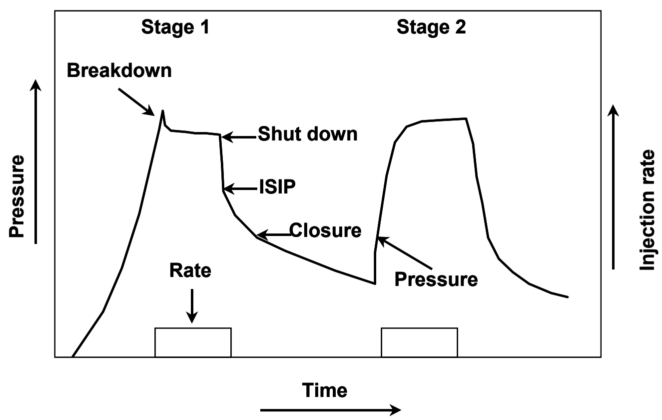

Fig. 5 – Typical stress test pump-in/shut-in.

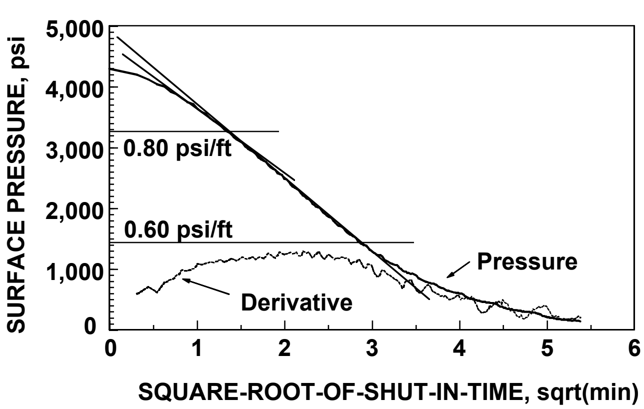

Fig. 6 – Closure pressure analysis.

Mini-fracture tests are run to reconfirm the value of *in-situ* stress in the pay zone and to estimate the fluid loss properties of the fracture fluid. A mini-fracture test is run using fluid similar to the fracture fluid that will be used in the main treatment. Several hundred barrels of fracturing fluid are normally pumped at fracturing rates. In coal seams, because the fracture height will usually be small, the mini-fracture test will often be eliminated or pumped with only a small volume of fracturing fluid. The purpose of the injection is to create a fracture that will be of similar height to the one created in the main fracture treatment. After the mini-fracture has been created, the pumps are shut down and the pressure decline is monitored. The pressure decline can be used to estimate the fracture closure pressure and the total fluid leak-off coefficient. Data from mini-fracture treatments can be used to alter the design of the main fracture treatment if the data determined during the mini-fracture test is substantially different that the data used to design the main fracture treatment.

For an injection-falloff test to be conducted successfully, it is necessary to have a clean connection between the wellbore and the created fracture. The purpose of *in-situ* stress tests and mini-fracture tests are to determine the pressure in the fracture when the fracture is open, and the pressure when the fracture is closed. If there is excess pressure drop near the wellbore, due to poor connectivity between the wellbore and the fracture, the interpretation of *in-situ* stress test data can be difficult. In coal seam reservoirs, due to the highly cleated nature of the coal, multiple fractures that follow tortuous paths are often created during injection tests.[14] When these tortuous paths are created, the pressure drop in the "near-wellbore" region can be very high, which complicates the analyses of the pressure falloff data. As such, *in-situ* stress test data and data from mini-fracture tests in coal seams are very difficult to measure and interpret.

The design engineer needs data from well tests to design the optimum fracture treatment. It is common for an operator to spend a lot of money and time running injection tests to determine values of *in-situ* stress, formation permeability, and leak-off coefficient. Fracture treatment theory is well grounded in science and engineering and, in most cases, data are collected from logs, cores and well tests to assure that designs are as accurate as possible.

3. Fracture Propagation Models

The first fracture treatments were pumped just to see if a fracture could be created and if sand could be pumped into the fracture. In 1955, Howard and Fast[15] published the first mathematical model that an engineer could use to design a fracture treatment. The Howard and Fast model assumed the fracture width was constant everywhere, allowing the engineer to compute fracture area based upon fracture fluid leakoff characteristics of the formation and the fracturing fluid.

2D Fracture Propagation Models

The Howard and Fast model was a two-dimensional (2D) model. In the following years, other 2D models were published.[16-19] When using a 2D model, the engineer fixes one of the dimensions (normally the fracture height), then calculates the width and length of the fracture. With experience and accurate data sets, 2D models can be used with confidence because the design engineer can accurately estimate the created fracture height beforehand.

Figs. 7 and 8 illustrate two of the most common 2D models used in fracture treatment design. The PKN geometry (Fig. 7) is normally used when the fracture length is much greater than the fracture height, while the KGD geometry (Fig. 8) is used if fracture height and length are similar [20]. Either of these two models can be used successfully to design hydraulic fractures. The key is to use models to make decisions. The design engineer must always compare actual results with the predictions from model calculations. By "calibrating" the 2D model with field results, the 2D models can be used to make design changes and improve the success of stimulation treatments.

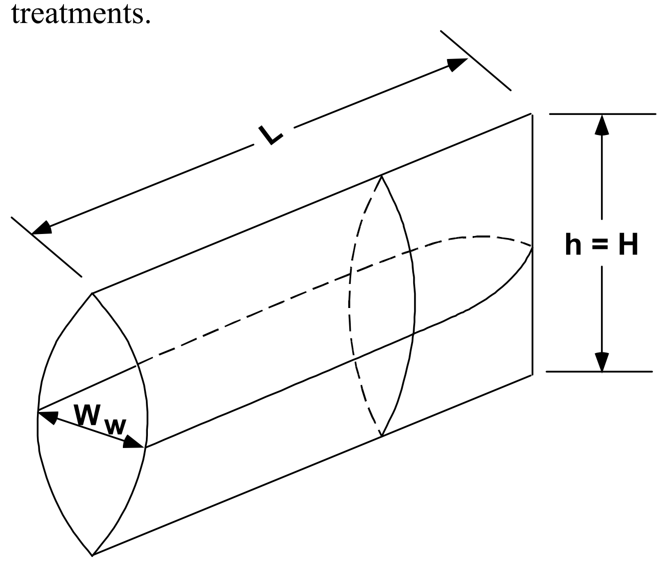

Fig. 7 – PKN geometry.

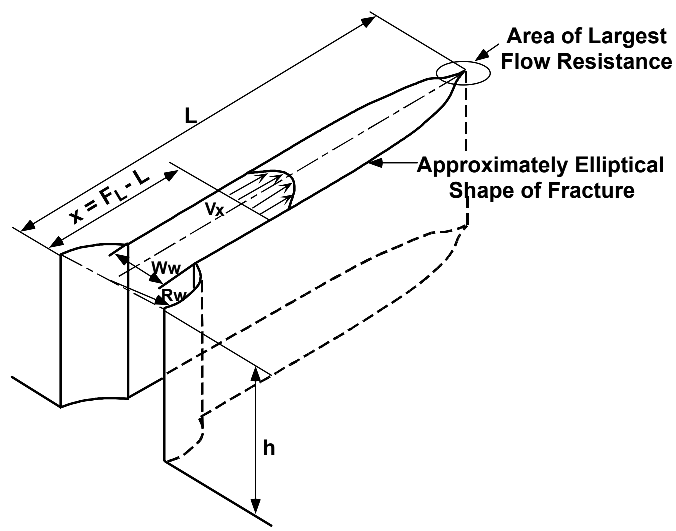

Fig. 8 – KGD geometry.

If the correct value of fracture height is used in a 2D model, the model will give reasonable estimates of created fracture length and width, provided, of course, that other parameters, such as *in-situ* stress, Young's modulus, formation permeability and total leakoff coefficient are also entered correctly. Engineers had to use 2D models for years due to the lack of computing power. Today, with high-powered computers available to most engineers, Pseudo 3-Dimensional (P3D) models are used by most fracture design engineers. P3D models are better than 2D models for most situations because the P3D model computes the fracture height, width and length distribution using the data for the pay zone and all the rock layers above and below the perforated interval.

<u>*3D Fracture Propagation Models*</u>

Clifton[21] provides a detailed explanation of how 3-Dimensional fracture propagation theory is used to derive equations for programming 3D models, as well as P3D models. Figs. 9 and 10 illustrate typical results from a P3D model. P3D models give more realistic estimates of fracture geometry and dimensions, which can lead to better designs and better wells. P3D models are used to compute the shape of the hydraulic fracture as well as the dimensions.

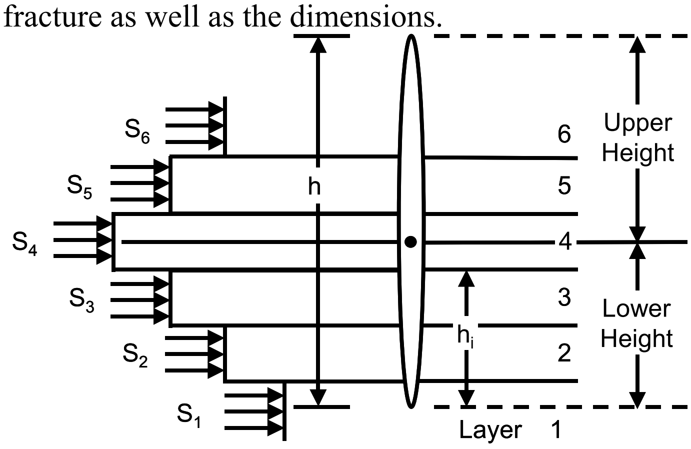

Fig. 9 – Width from a P3D model.

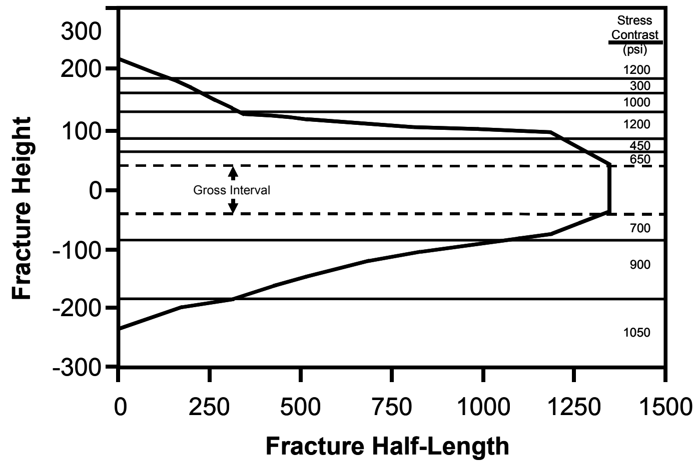

Fig. 10 – Width and height from P3D model.

4. Fracturing Fluids and Additives

To create the fracture, a fluid is pumped into the wellbore at high rate to increase the pressure in the wellbore at the perforations to a value greater than the breakdown pressure of the formation. The breakdown pressure is generally believed to be the sum of the *in-situ* stress and the tensile strength of the rock. Once the formation is broken down, and the fracture is created, then the fracture can be propagated at a pressure called the fracture propagation pressure. The fracture propagation pressure is equal to the sum of the *in-situ* stress, plus the net pressure drop, plus the near wellbore pressure drop. The net pressure drop is equal to the pressure drop down the fracture due to viscous fluid flow in the fracture. The near wellbore pressure drop can be a combination of the pressure drop of the viscous fluid flowing through the perforations and/or the pressure drop due to tortuosity between the wellbore and the propagating fracture. Thus, the fracturing fluid properties are very important in the creation and propagation of the fracture.

<u>Properties of a Fracturing Fluid</u>

The ideal fracturing fluid should be compatible with the formation rock, compatible with the formation fluid, generate enough pressure drop down the fracture to create a wide fracture, be able to transport the propping agent in the fracture, break back to a low viscosity fluid for clean up after the treatment, and be cost effective. The family of fracture fluids available consist of water base fluids, oil base fluids, acid base fluids and foam fluids. Table 2 lists the types of fracturing fluids that are available and the general use of each type of fluid. For most reservoirs, water base fluids with appropriate additives will be the best fluid. In some cases, foam generated using N_2 or CO_2 can be used to successfully stimulate shallow, low-pressure zones. When water is used as the base fluid, the water should be tested for quality. Table 4 presents generally accepted levels of water quality for use in hydraulic fracturing.

Table 4 - Acceptable Levels for Mix Water

pH	6-8
Iron	< 10 ppm
Oxidizing Agents	None
Reducing Agents	None
Carbonate*	< 300 ppm
Bicarbonate*	< 300 ppm
Bacteria	None
Cleanliness	Reasonable
*Higher Carbonate/Bicarbonate Content Will Require Further Pilot Testing on Gel Break, and Crosslinking	

The viscosity of the fracture fluid is important. The fluid should be viscous enough (normally 50–1000 cp) to create a wide fracture (normally 0.2–1.0 in) and transport the propping agent into the fracture (normally 10s to 100s of feet). The density of the fluid is also important. Water based fluids have densities near 8.4 ppg. Oil base fluids, although never used to fracture treat coal seam reservoirs, will have densities that are 70-80% of the water based fluids. Foam fluids can have densities that are 50% or less those of water based fluids. The density affects the surface injection pressure and the ability of the fluid to flow back after the treatment. In low pressure reservoirs, low density fluids, like foam, can be used to assist in the fluid clean up.

A fundamental equation used in all fracture models is that the fracture volume is equal to the total volume of fluid injected minus the volume of fluid that leaks off into the reservoir. The fluid efficiency is the percentage of fluid that is still in the fracture at any point in time, when compared to the total volume injected at the same point in time. The concept of fluid loss was used by Howard and Fast to determine fracture area [15]. If too much fluid leaks off, the fluid has a low efficiency (say 10-20%) and the created fracture volume will be only a fraction of the total volume injected. However, if the fluid efficiency is too high (say 80-90%), the fracture will not close rapidly after the treatment. Ideally, a fluid efficiency between 40-60% will provide an optimum balance between creating the fracture and having the fracture close down after the treatment.

In most low permeability reservoirs, fracture fluid loss and efficiency is controlled by the formation permeability. In high permeability formations, a fluid-loss additive must be added to the fracture fluid to reduce leak-off and improve fluid efficiency. In highly cleated coal seams, the leak-off can be extremely high, with efficiencies down in the 10-20% range. To fracture treat these highly cleated coal seams, the treatment must often be pumped at high injection rates using fluid loss additives. In general, the objective of most fracture treatments in coal seams is to create a short, wide fracture to connect the coal cleat system to the well bore vs. creating long hydraulic fractures that penetrate deeply into the coal seam. Therefore, water with very few additives, pumped at medium to high injection rates is commonly used to stimulate coal seam reservoirs.

Fracture Fluid Additives

Typical additives for a fracture fluid have been described in detail by Ely [22]. Typical additives for a water based fluid are briefly described below.

- Polymers – used to viscosify the fluid
- Crosslinkers – used to change the viscous fluid to a pseudo-plastic fluid
- Biocides – used to kill bacteria in the mix water
- Buffers – used to control the pH of the fracture fluid
- Surfactants – used to lower the surface tension
- Fluid loss additives – used to minimize fluid leak-off into the formation
- Stabilizers – used to keep the fluid viscous at high temperature
- Breakers – used to break the polymers and crosslink sites at low temperature

Additional information on additives is presented in Table 5.

Table 5 – Summary of Chemical Additives

Type of Additive	Function Performed	Typical Products
Biocide	Kills bacteria	Gluteridehyde carbonate
Breaker	Reduces fluid viscosity	Acid, oxidizer, enzyme breaker
Buffer	Controls the pH	Sodium bicarb., fumaric acid
Clay stabilizer	Prevents clay swelling	KCl, NH CL, KCl substitutes
Diverting agent	Diverts flow of fluid	Ball sealers, rock salt, flake boric-acid
Fluid loss additive	Improves fluid efficiently	Diesel, particulates, fine sand
Friction reducer	Reduces the friction	Anionic copolymer
Iron Controller	Keeps iron in solution	Acetic & citric acid
Surfactant	Lowers surface tension	Fluorocarbon, Nonionic
Gel stabilizer	Reduces thermal degradation	MEOH, sodium thiosulphate

The owner of the oil or gas well normally does not own the equipment or the additives required to pump a fracture treatment. The operator will hire a service company to pump the fracture treatment. Each service company has their own research department for developing fracture fluids and additives. Each service company obtains their additives from various suppliers. As such, there is no set of rules one can use to select the proper additives for a fracture fluid, without first consulting with the service company that will mix and pump the fluid into the well. Many times, pilot tests of the fracture fluids must be conducted to be certain all the additives will work properly at the temperature in the reservoir and for the duration of the treatment.

All operating and service companies are concerned with protecting the environment and the USDW. As such, research is being conducted in developing "green additives" to use in hydraulic fracturing, especially in shallow formations like coal seam reservoirs. It costs a lot of money to handle additives and dispose of fracturing fluids that are either left over after the treatment or produced back from the well bore. The development of new, green additives will be a new technology that will benefit all parties.

5. Propping Agents and Fracture Conductivity

Propping agents are required to "prop-open" the fracture once the pumps are shut down and the fracture begins to close. The ideal propping agent will be strong, resistant to crushing, resistant to corrosion, have a low density, and readily available at low cost.[23] The products that best meet these desired traits are silica sand, resin-coated sand, and ceramic proppants.

Types of Propping Agents

Silica sand is obtained from sand mining operations. There are several sources in the

United States and a few outside the US. The sand must be tested to be sure it has the necessary compressive strength to be used in any specific situation. Generally, sand is used to prop open fractures in shallow formations. For coal seam reservoirs, sand is usually the best choice for a propping agent and virtually every fracture treatment in a coal seam reservoir uses sand. Sand is much less expensive per pound than the resin-coated sand or the ceramic proppants.

Resin-coated (epoxy) sand is stronger than sand and is used where more compressive strength is required to minimize proppant crushing. Some resins can be used to form a consolidated sand pack in the fracture, which will help to eliminate proppant flow back into the wellbore. Resin coated sand is more expensive than sand.

Ceramic proppants consist of sintered bauxite, intermediate strength proppant (ISP), and light weight proppant (LWP). The strength of the proppant is proportional to its density. Also, the higher strength proppants, like sintered bauxite, cost more than ISP and LWP. Ceramic proppants are used to stimulate deep (>8,000 ft) wells where large values of *in-situ* stresses will apply large forces on the propping agent.

Factors Affecting Fracture Conductivity

The fracture conductivity is the product of propped fracture width and the permeability of the propping agent, as illustrated in Fig. 11. The permeability of all the propping agents, sand, resin-coated sand, and the ceramic proppants, will be 200+ darcies when no stress has been applied to the propping agent. However, the conductivity of the fracture will be reduced during the life of the well because of increasing stress on the fracture, stress corrosion affecting the proppant strength, proppant crushing, proppant embedment into the formation, and damage due to gel residue or fluid loss additives.

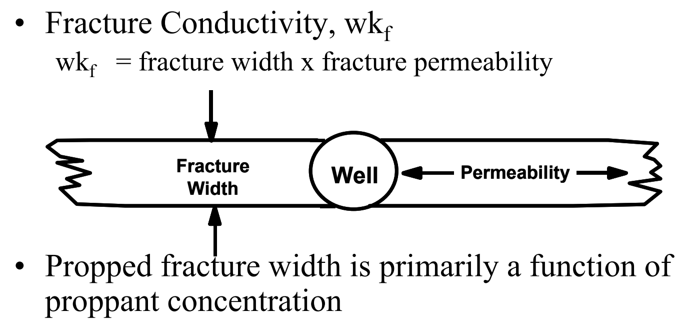

Fig. 11 – Definition of fracture conductivity.

The effective stress on the propping agent is the difference between the *in-situ* stress and the flowing pressure in the fracture, as illustrated in Fig. 12. As the well is produced, the effective stress on the propping agent will normally increase because the value of the flowing bottom hole pressure will be decreasing. However, as can be seen by examining Eq. 1, the *in-situ* stress will decrease with time as the reservoir pressure declines. This phenomenon of decreasing *in*-situ stress as the reservoir pressure declines was proven conclusively by Salz.[8] In shallow coal seam reservoirs, the effective stress on the propping agent is always low and does not normally affect the fracture conductivity.

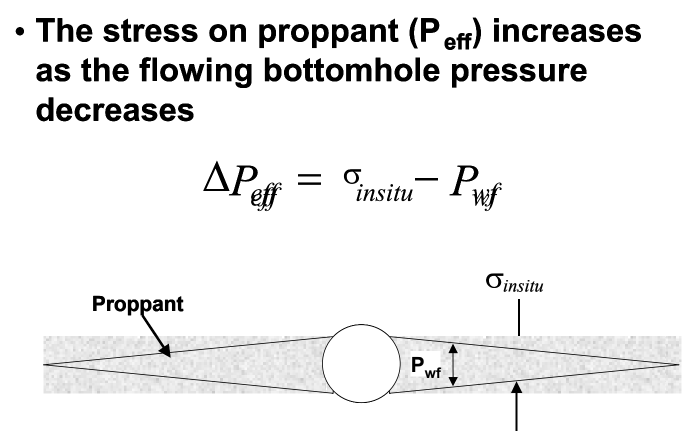

Fig. 12 – Effective stress on proppant.

Fig. 13 illustrates the differences is fracture conductivity vs. increasing effective stress on the propping agent for a variety of commonly used propping agents. The data in Fig. 13 clearly show that for shallow wells, where the effective stress is less than 4000 psi, sand can be used to create high conductivity fractures. As the effective stress increases to larger and larger values, then the higher strength, more expensive propping agents must be used to create a high conductivity fracture.

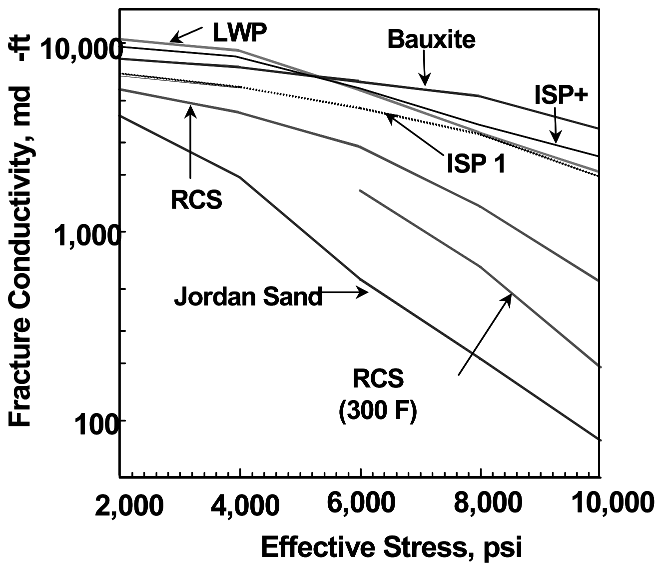

Fig. 13 – Effect of stress on conductivity.

6. Fracture Treatment Design

Data Requirements

In **Section 1** of this paper, the data required by the engineer to design a hydraulic fracture treatment was discussed. The data were divided into two groups: (1) data that must be measured or estimated and (2) data that can be controlled by the design engineer. The primary data that can be controlled by the engineer are the well completion details, treatment volume, pad volume, injection rate, fracture fluid viscosity, fracture fluid density, fluid loss additives, propping agent type, and propping agent volume.

As stated earlier, the most important data are (1) the *in-situ* stress profile, (2) formation permeability, (3) fluid loss characteristics, (4) total fluid volume pumped, (5) propping agent type and amount, (6) pad volume, (7) fracture fluid viscosity, (8) injection rate, and (9) formation modulus. The two most important parameters are the *in-situ* stress profile and the permeability profile of the zone to be stimulated and the layers of rock above and below the target zone.

There is a structured methodology followed by the engineer to design, optimize, execute, evaluate and re-optimize the fracture treatments in any reservoir. The first step is always the construction of a complete and accurate data set. Table 1 lists the sources for the data required to run fracture propagation and reservoir models. Notice that the design engineer must be capable of analyzing logs, cores, production data, well test data, and digging through well files to obtain all the information needed to design and evaluate a well that is hydraulically fracture treated.

Design Procedures

To design the optimum treatment, the engineer must determine the effect of fracture length and fracture conductivity upon the productivity and the ultimate recovery from the well. As in all engineering problems, sensitivity runs need to be made to evaluate uncertainties, such as formation permeability and drainage area. In coal seam reservoirs, uncertainties can also exist in variables such as the gas content and the desorption rate. The production data obtained from the reservoir model should be used in an economics model to determine the optimum fracture length and conductivity. Then a fracture treatment must be designed using a P3D fracture propagation model to achieve the desired length and conductivity at minimum cost. The most important concept is to design a fracture using all data and appropriate models that will result in the optimum economic benefit to the operator of the well.

A P3D hydraulic fracture propagation model should be run to determine what needs to be mixed and pumped into the well to achieve the optimum values of propped fracture length and fracture conductivity. The base data set should be used to make a base case run. Then, the engineer determines which variables are the most uncertain. Many times, the values of *in-situ* stress, modulus, permeability, fluid loss coefficient, for example, are not known with certainty and have to be estimated. The design engineer acknowledges these uncertainties and makes sensitivity runs with the P3D model to determine the effect of these uncertainties on the design process. As databases are developed, the number and magnitude of the uncertainties will diminish.

In effect, the design engineer should fracture treat the well many times on his or her computer screen. Making these sensitivity runs will (1) lead to a better design and (2) educate the design engineer on how certain variables affect the ultimate values of both the created and the propped fracture dimensions. Such designs will be comprehensive, will consider uncertainties, and will be developed using professional processes.

Fracturing Fluid Selection

A critical decision by the design engineer is the selection of the fracture fluid for the treatment. Economides *et al.* [24] developed a flow chart that can be used to select the category of fracture fluid on the basis of factors such as reservoir temperature, reservoir pressure, the expected value of fracture half-length, and a determination if the reservoir is water sensitive. Their fluid selection flow chart for a gas well is presented in Fig. 14.

Most productive coal seam reservoirs are less than 5000 ft deep. The permeability in highly cleated coal seams decreases with increasing depth and overburden stress. At depths greater than about 5000 ft, in most cases, the coal seam does not have enough permeability to be economically developed.

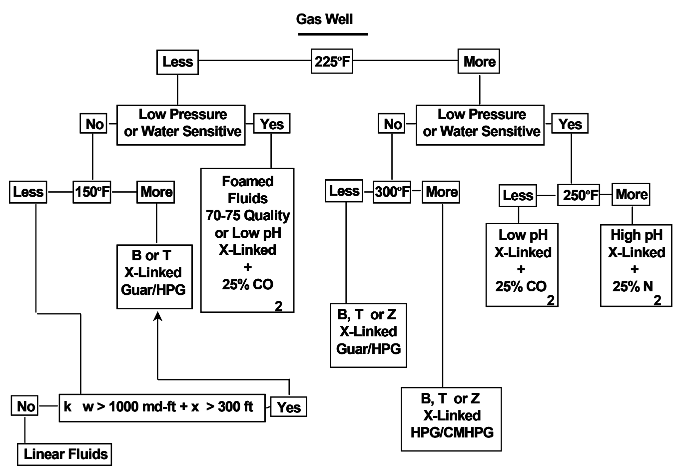

Fig. 14 – Selecting a fracture fluid.

Because most productive coal seams are shallow, low temperature reservoirs, then the choice of fracturing fluid (according to Fig. 14) will be (1) N_2 foam for low pressure reservoirs, (2) linear water based fluids if all you need is a short, low conductivity fracture, or (3) cross-linked gel if you need a wide or long fracture. Holditch *et al.*[14] discussed the criteria for selecting a fracturing fluid in the Gas Research Institute's Coal Seam Stimulation Manual.

For thick highly cleated coals, a crosslinked fluid should be used to create wide fractures and place as much proppant as possible in the fractures close to the wellbore. The purpose of the treatment is to link up the cleats to the wellbore using the hydraulic fracture and the proppant. The fluid should use the minimum amount of gel possible and breaker should be used to minimize damage to the fracture, and to assist in cleanup.

If the fracture is intended to connect up several thin coal seams that are vertically scattered up and down the wellbore, then coil tubing can be used to selectively stimulate each coal seam. Fig. 15 illustrates how coil tubing can be used to stimulated multiple intervals, one at a time.

- Single or multiple fracturing stimulation using coiled tubing as a conduit for both the isolation and the treatment.

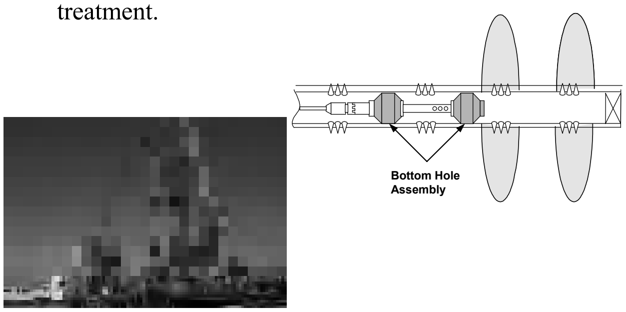

Fig. 15 – Fracturing using coil tubing.

In low-pressure coal seams, N_2 foam can be used as the fracture fluid. Foamed fracture fluids will create wide fractures, can transport the propping agent, and are easier to clean up than fluids that do not contain N_2.

Propping Agent Selection

Economides *et al.* [24] also produced a flow chart for selecting propping agents. Their chart is included as Fig. 16. Because most productive coal seams are shallow, sand is always used as the propping agent. In certain cases, where proppant flow back becomes a problem, then resin-coated sand is sometimes used. Special care must be used to design such treatments, because at low temperature, it may be difficult to get the resin to set and to create the consolidated sand pack needed to prevent proppant flow back.

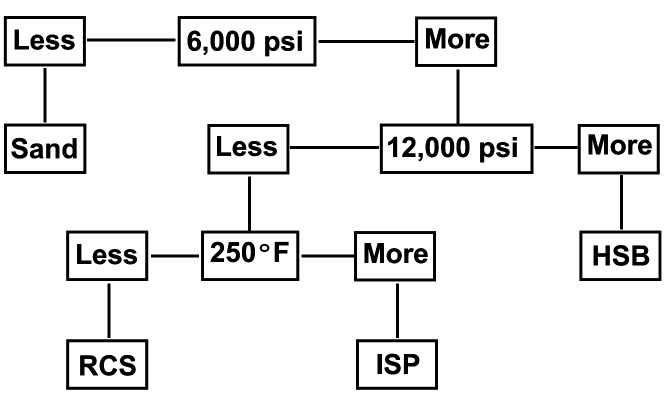

Fig. 16 - Proppant selection based on closure pressure.

To determine the optimum fracture conductivity, the design engineer should use the dimensionless conductivity (Cr) concept published by Cinco-Ley [25].

$$C_r \quad \frac{P_i \; K \; L_f}{wk_f} \qquad \text{Eq. 2}$$

where w is the fracture width (ft), k_f is the proppant permeability (md), k is the formation permeability (md), and L_f is the fracture half-length. To minimize the pressure drop down the fracture, the value of Cr should be approximately equal to ten (10).

For example, in a coal seam, if the formation permeability is 25 md, and the optimum fracture half-length is 50 ft, then the optimum fracture conductivity would be 3,927 md-ft. The engineer needs to design the treatment to create a fracture wide enough, and pump proppants at concentrations high enough to achieve the high conductivity required to optimize the treatment.

Some engineers tend to compromise fracture length and conductivity in an often-unsuccessful attempt to prevent damage to the formation around the fracture. Holditch[26] showed that substantial damage to the formation around the fracture can be tolerated as long as the optimum fracture length and conductivity are achieved. Ideally, the design engineer can create the optimum fracture length and conductivity while minimizing damage to the formation. If the opposite occurs, that is, the formation is not damaged, but the fracture is not long enough or conductive enough, then the well performance will be disappointing.

The operator of the well should always evaluate the risks such as mechanical risks, product price risks and geologic risks. Uncertainties in the input data can be evaluated by making sensitivity runs using both the reservoir models and the fracture propagation models. One of the main risks in hydraulic fracturing is that the entire treatment will be pumped and/or paid for (i.e. the money is spent), but for whatever reason, the well does not produce at the desired flow rates nor recovers the expected cumulative recovery. Many times, mechanical problems with the well or the surface equipment cause the treatment to fail. Other times, the reservoir does not respond as expected.

To evaluate the risk of mechanical or reservoir problems, the design engineer can use 100% of the costs on only a fraction of the revenue in the economic analyses. For example, say one (1) in every five (5) fracture treatments in a certain formation is not successful. Then one can use 80% of the expected revenue and 100% of the expected costs to determine the optimum fracture length. An illustration of how such an analyses can alter the desired fracture length is presented in Fig. 17.

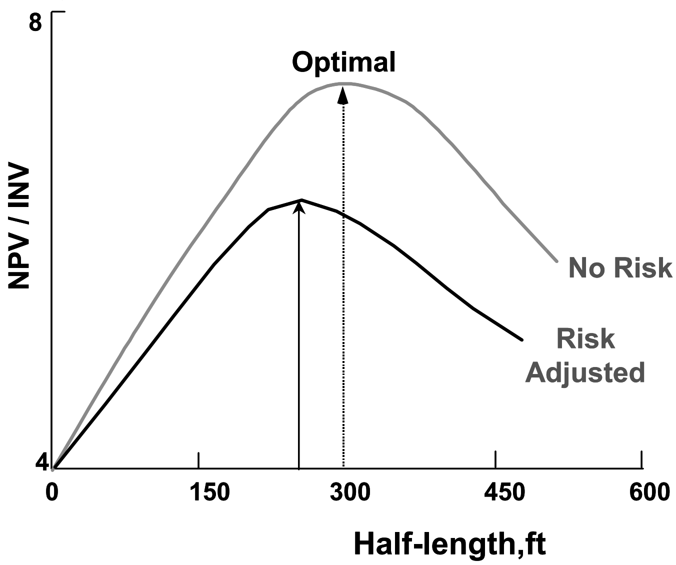

Fig. 17 – Economic analysis.

Finally, after the optimum, risk adjusted fracture treatment has been designed, it is extremely important to be certain the optimum design is pumped correctly into the well. For this to occur, the design engineer and the service company should work together to provide quality control before, during and after the treatment is pumped. The best engineers tend to spend sufficient time in the office to design the treatment correctly, then go to the field to help supervise the field operations (or provide on-site advice to the supervisor).

7. Post-Fracture Well Behavior

The original fracture treatments in the 1950's were designed to increase well productivity. These treatments were normally pumped to remove damage in moderate to high permeability wells. McGuire and Sikora[27] and Prats[28] published equations that were used for many years to design fracture treatments that resulted in desired folds of increase in the productivity index of a well. The productivity index of an oil well is

$$J = \frac{q_o}{\left(p_e - p_{wf}\right)} \qquad \text{Eq. 3}$$

and for a gas well is

$$J = \frac{q_g \quad \mu\, z}{\left(p_e^2 - p_{wf}^2\right)} \qquad \text{Eq. 4}$$

J is the productivity index in terms of barrels per psi per day or mcf per psi squared per day. The viscosity and compressibility are included in the equation for productivity index of a gas well, because they are pressure dependent.

Assuming J is the productivity index for a fractured well at steady state flow, and Jo is the productivity index of the same well under radial flow conditions, Prats[28] found that

$$\frac{J}{J_o} = \frac{\ln\left(\frac{r_e}{r_w}\right)}{\ln\left(\frac{r_e}{0.5\,L_f}\right)} \qquad \text{Eq. 5}$$

for a well containing an infinite conductivity fracture whose fracture half-length is L_f. Prats found that a well with a fracture half-length of 100 ft will produce as if the well had been drilled with a 100 ft diameter drill bit. In other words, the hydraulic fracture, if conductive enough, acts to extend the wellbore and stimulate flow rate from the well. If the dimensionless fracture conductivity, Cr (Eq. 2), is equal to 10 or greater, the hydraulic fracture will essentially act as if it is an infinately conductive fracture.

In coal seam reservoirs, the gas diffuses through the coal into the cleat system. If the cleat system is poorly developed and the permeability of the coal is low (<<1md), then the coal reservoir will probably not be economic to produce because it is almost impossible to create long, conductive fractures in thin coal seams. Thus, most commercial coal seam reservoirs are highly cleated, moderate permeability (5md<k<100md) reservoirs. As such, short, conductive fractures are required and large volumes of fluids are not needed to stimulate highly cleated coal seam reservoirs. The object of a hydraulic fracture in a highly cleated coal seam is to connect the cleat system with the well bore using the hydraulic fracture fluids and proppants.

8.0 Fracture Diagnostics

Fracture diagnostics involves analyzing the data before, during and after a hydraulic fracture treatment to determine the shape and dimensions of both the created and propped fracture. Fracture diagnostic techniques have been divided into several groups.[29]

Group 1 – Direct far field techniques

Direct far field methods are comprised of tiltmeter fracture mapping and microseismic fracture mapping techniques. These techniques require delicate instrumentation that has to be emplaced in boreholes surrounding and near the well to be fracture treated. When a hydraulic fracture is created, the expansion of the fracture will cause the earth around the fracture to deform. Tiltmeters can be used to measure the deformation and to compute the approximate direction and size of the created fracture. Surface tiltmeters are placed in shallow holes surrounding the well to be fracture treated and are best for determining fracture orientation and approximate size. Downhole tiltmeters are placed in vertical wells at depths near the location of the zone to be fracture treated. As with surface tiltmeters, downhole tiltmeter data can be analyzed to determine the orientation and dimensions of the created fracture, but are most useful for determining fracture height. Tiltmeters have been used on an experimental basis to map hydraulic fractures in coal seams.[11]

Microseismic fracture mapping relies on using a downhole receiver array of accelerometers or geophones to locate microseisms or micro-earthquakes that are triggered by shear slippage in natural fractures surrounding the hydraulic fracture. The principle of microseismic fracture mapping[29] is illustrated in Fig. 18. In essence, noise is created in a zone surrounding the hydraulic fracture. Using sensitive arrays of instruments, the noise can be monitored, recorded, analyzed and mapped.

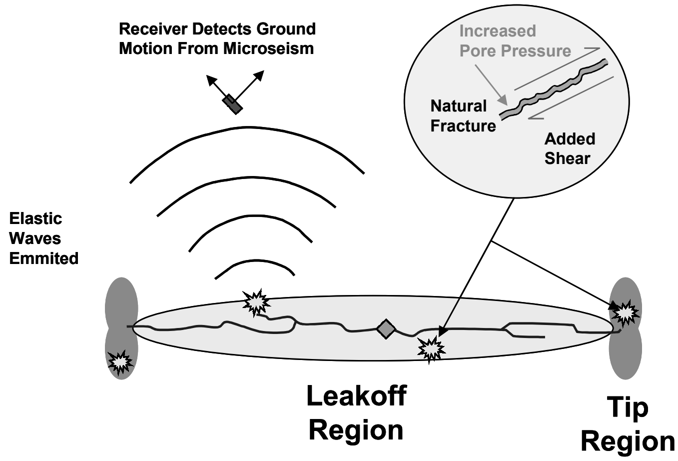

Fig. 18 – Principle of microseismic fracture mapping.

Tiltmeters have been used extensively in the oil and gas industry for more than 10 years, although it has only been recent that the technology has been available to look at fractures at depths greater than 4,000ft. Current surface tiltmeter technology can see below 10,000ft. Microseismic monitoring has traditionally been too expensive to be used on anything but research wells, but its cost has dropped dramatically in the past few years, so although still expensive (on the order of $50,000 to $100,000), it is being used more commonly throughout the industry. As with all monitoring and data collection techniques, however, the economics of marginal wells makes it difficult to justify any extra expense. If the technology is used at the beginning of the development of a field, however, the data and knowledge gained are often used on subsequent wells, effectively spreading out the costs.

Group 2 – Direct near-wellbore techniques

Direct near-wellbore techniques are run in the well that is being fracture treated to locate or image the portion of fracture that is very near (inches) the wellbore. Direct near-wellbore techniques consist of tracer logs, temperature logging, production logging, borehole image logging, downhole video logging, and caliper logging. If a hydraulic fracture intersects the wellbore, these direct near-wellbore techniques can be of some benefit in locating the hydraulic fracture.

However, these near-wellbore techniques are not unique and can not supply information on the size or shape of the fracture once the fracture is 2-3 wellbore diameters in distance from the wellbore. In coal seams, where multiple fractures are likely to exist, the reliability of these direct near-wellbore techniques are even more speculative. As such, very few of these direct near-wellbore techniques are used on a routine basis to look for a hydraulic fracture.

Group 3 – Indirect fracture techniques

The indirect fracture techniques consist of hydraulic fracture modeling of net pressures, pressure transient test analyses, and production data analyses. Because the fracture treatment data and the post-fracture production data are normally available on every well, the indirect fracture diagnostic techniques are the most widely used methods to determine the shape and dimensions of both the created and the propped hydraulic fracture.

The fracture treatment data can be analyzed with a P3D fracture propagation model to determine the shape and dimensions of the created fracture. The P3D model is used to history match the fracturing data, such as injection rates and injection pressures. Input data, such as the *in-situ* stress and permeability in key layers of rock can be varied (within reason) to achieve a history match of the field data.

Post-fracture production and pressure data can be analyzed using a 3D reservoir simulator to estimate the shape and dimensions of the propped fracture. Values of formation permeability, fracture length and fracture conductivity can be varied in the reservoir model to achieve a history match of the field data.

The main limitation of these indirect techniques is that the solutions are not very unique and require as much fixed data as possible. For example, if the engineer has determined the formation permeability from a well test or production test prior to the fracture treatment, so that the value of formation permeability is known and can be fixed in the models, the solution concerning values of fracture length become more unique. Most of the information in the literature concerning post-fracture analyses of hydraulic fractures has been derived from these indirect fracture diagnostic techniques.

Limitations of fracture diagnostic techniques

Warpinski discussed many of these same fracture diagnostic techniques.[30] Table 6, from Warpinski's paper, lists certain diagnostic techniques and their limitations. In general, fracture diagnostics is expensive and only used in research wells. Fracture diagnostic techniques do work and can provide important data when entering a new area or a new formation. However, in coal seam wells, where costs must be minimized to maintain profitability, fracture diagnostic techniques are rarely used and are generally cost prohibitive.

Table 6 – Limitations of Fracture Diagnostic Techniques

Parameter	Technique	Limitation
Fracture Height	Tracer logs	Shallow depth of investigation; shows height only near the wellbore
Fracture Height	Temperature logs	Difficult to interpret; shallow depth of investigation; shows height only near wellbore
Fracture Height	Stress profiling	Does not measure fracture directly; must be calibrated with *in-situ* stress tests
Fracture Height	P3D models	Does not measure fracture directly; estimates vary depending on which model is used
Fracture Height	Microseismic	Optimally requires nearby offset well; difficult to interpret; expensive
Fracture Height	Tiltmeters	Difficult to interpret; expensive and difficult to conduct in the field
Fracture Length	P3D models	Length inferred, not measured; estimates vary greatly depending on which model is used
Fracture Length	Well testing	Large uncertainties depending upon assumptions and lack of prefracture welltest data
Fracture Length	Microseismic	Optimally requires nearby offset well; difficult to interpret; expensive
Fracture Length	Tiltmeters	Difficult to interpret; expensive and difficult to conduct in the field
Fracture Azimuth	Core techniques	Expensive to cut core and run tests; multiple tests must be run to assure accuracy
Fracture Azimuth	Log techniques	Requires open hole logs to be run; does not work if natural fractures are not present
Fracture Azimuth	Microseismic	Analysis intensive; expensive for determination of azimuth
Fracture Azimuth	Tiltmeters	Useful only to a depth of 5000 ft; requires access to large area; expensive

9.0 Nomenclature

CMHPG	=	Carboxymethylhydroxypropyl-guar
HEC	=	Hydroxyelthycellulose
HPG	=	Hydroxypropylguar
ISIP	=	Instantaneous shut-in pressure
ISP	=	Intermediate strength proppant
k	=	Formation permeability, md
KCL	=	Potassium chloride
KGD	=	Kristonovich, Geertsma, Daneshy
L_f	=	Fracture half-length, ft
LWP	=	Light weight proppant
MEOH	=	Methanol
MRO	=	Memory readout gauge
NH_4CL	=	Ammonium chloride
PKN	=	Perkins, Kern, Nordgren
RSC	=	Resin coated sand
SRO	=	Surface Readout gauge
wk_f	=	Fracture conductivity, md-ft
α	=	Biot's constant
ν	=	Poissons' ratio
σ_{ext}	=	Tectonic stress
σ_{min}	=	Minimum horizontal stress (*in-situ* stress)
σ_{ob}	=	Overburden stress
σ_p	=	Reservoir fluid pressure or pore pressure
σ_1	=	Vertical (overburden) stress
σ_2	=	Minimum horizontal stress
σ_3	=	Maximum horizontal stress

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Appendix B

Quality Assurance Plan: Evaluation of Impacts to Underground Sources of Drinking Water by Hydraulic Fracturing of Coalbed Methane Reservoirs

The U.S. Environmental Protection Agency (EPA), bases environmental protection efforts on the best available scientific information and sound science. The credibility of the resulting policy decision depends, to a large extent, on the strength of the scientific evidence on which it is based. Sound science can be described as organized investigations and observations conducted by qualified personnel using documented methods and leading to verifiable results and conclusions (SETAC, 1999).

This Quality Assurance Plan for data collection and evaluation describes the procedures the Agency used for a systematic and well-documented, graded approach to realizing the goal for the "Evaluation of Impacts to Underground Sources of Drinking Water by Hydraulic Fracturing of Coalbed Methane Reservoirs." The goal of Phase I of EPA's hydraulic fracturing study was to assess the potential for contamination of USDWs due to the injection of hydraulic fracturing fluids into CBM wells and to determine based on these findings, whether further study is warranted. This Quality Assurance Plan (developed following the guidelines of EPA publication 240/B-01/003) guides the production of a set of data and scientific findings that are sound, with conclusions supported by the data.

1.0 Project Management

This section of the Quality Assurance Plan addresses the basic area of project management, including the project history and objectives, and roles and responsibilities of the participants.

1.1 Project and Task Organization

Overall project management was provided by the EPA's Office of Water, Groundwater and Drinking Water (OGWDW), Groundwater Protection Division. Data was gathered by an EPA OGWDW contractor.

The contractor compiled the gathered data into a draft summary report, reviewed the draft report, and submitted the draft report to EPA and other federal agencies for review. After the contractor addressed comments from EPA and other federal agencies, EPA submitted the draft report to a Peer Review Panel for their comments (see Table B-1 for a list of the members of the Peer Review Panel). Following receipt of comments from the Peer Review Panel, EPA and its contractors responded to those comments. The availability of the report for stakeholder review and comment was announced in the *Federal Register* on August 28, 2002.

Table B-1: Peer Review Panel

Name	Affiliation	Education	Experience
Morris Bell	Engineer, Colorado Oil and Conservation Commission	Engineering Degree, University of Oklahoma	Closely involved with coalbed methane development in the San Juan and Raton Basins. Has investigated water well complaints and directed projects to test water wells. Worked for Amoco as a production engineer, drilling and completing tight gas wells. Also worked as a consultant, specializing in the completion and evaluation of coalbed methane wells.
Peter E. Clark	Associate Professor, Dept. of Chemical Engineering and Material Science, University of Alabama	Ph.D., University of Oklahoma State University	Specializes in complex fluid flows and hydraulic fracturing. Has taught several courses in the Chemical Engineering, Mineral Engineering, Engineering Mechanics, and Civil Engineering Departments. These courses included fluid mechanics, petroleum rock and fluids, well completion, drilling, and natural gas engineering.
David Hill	Manager, Engineering Resources, Gas Technology Institute (GTI)	MBA, Northwestern University; BS, Marietta College, Petroleum Engineering	Expertise includes unconventional reservoirs (e.g., coalbed methane, gas shales, tight sands); hydraulic fracturing; and reservoir evaluation in technical, managerial, and marketing aspects of technology development, deployment, and commercialization. Has authored and co-authored over 40 articles about oil- and gas-related research and development, and field-based operations.

Table B-1: Peer Review Panel

Name	Affiliation	Education	Experience
Buddy McDaniel	Technical Advisor for Production Enhancement Technology, Halliburton	B.S., Chemical Engineering, University of Oklahoma	Specializes in applications for highly deviated and horizontal wellbores and understanding of reservoir response to fracturing applications. Conducted research related to laboratory measurement of fracture conductivity of proppants under simulated reservoir conditions. Was actively involved in design and application of hydraulic fracturing treatments in soft chalks, deviated and horizontal wellbores, gas storage wells, geothermal wells, and conventional hydrocarbon reservoirs.
Jon Olson	Asst. Professor, Dept. of Petroleum and Geosystems Engineering, University of Texas at Austin	Ph.D., Stanford University, Applied Earth Sciences	Worked in the areas of fracture mechanics and coal geology and has published several papers on these subjects. Was employed by Mobil Exploration for several years as research engineer in the areas of rock mechanics, structural geology, and well performance.
Ian Palmer	Senior Petroleum Engineer, BP Amoco	Ph.D., University of Adelaide in Australia	Has worked extensively in coalbed methane extraction, including fracture design and prediction, rock mechanisms of coal, and openhole cavity completions. Also developed hydraulic fracturing models.
Norm Warpinski	Distinguished Member of Technical Staff, Sandia Laboratories	Ph.D., University of Illinois, Mechanical Engineering	Authority on hydraulic fracturing, geomechanics, poroelasticity, in situ stresses, and production mechanisms. Has expertise ranging from theoretical modeling and laboratory testing to field and in situ mineback experiments. Serves as project manager and lead scientist for a program to develop hydraulic fracture diagnostic technology for use in industry fracturing applications. Has published extensively on subject of hydraulic fracturing.

1.2 Problem Definition and Background

Hydraulic fracturing is a half century-old technology used in oil and natural gas production. The hydraulic fracturing process uses very high hydraulic pressures to initiate a fracture. A hydraulically induced fracture acts as a conduit in the rock or coal formation that allows the oil or coalbed methane to travel more freely from the rock pores (where the oil or methane is trapped) to the production well that can bring it to the surface.

After a well is drilled into a reservoir rock that contains oil, natural gas, and water, every effort is made to maximize the production of oil and gas. One way to improve or maximize the flow of fluids to the well is to connect many pre-existing fractures and flow pathways in the reservoir rock with a larger fracture. This larger, man-made fracture starts at the well and extends out into the reservoir rock for as much as several hundred feet. To create or enlarge fractures, a thick fluid, typically water-based, is pumped into the coal seam at a gradually increasing rate and pressure. Eventually the coal seam is unable to accommodate the fracturing fluid as quickly as it is injected. When this occurs, the pressure is high enough that the coal fractures along existing weaknesses within the coal. Along with the fracturing fluids, sand (or some other propping agent or "proppant") is pumped into the fracture so that the fracture remains "propped" open even after the high fracturing pressures have been released. The resulting proppant-containing fracture serves as a conduit through which fracturing fluids and groundwater can more easily be pumped from the coal seam.

To initiate coalbed methane production, groundwater and some of the injected fracturing fluids are pumped out (or "produced" in the industry terminology) from the fracture system in the coal seam. As pumping continues, the pressure eventually decreases enough so that methane desorbs from the coal, flows toward, and is extracted through the production well.

EPA is conducting a study to assess the potential for contamination of underground sources of drinking water (USDWs) due hydraulic fracturing fluid injection into coalbed methane wells. The study focuses on hydraulic fracturing used specifically for enhancing coalbed methane production. EPA, through its contractors and subcontractors, gathered information on the hydraulic fracturing process and requested comment from the public on contamination allegedly due to hydraulic fracturing practices. In this Phase I effort, EPA did not incorporate new, scientific fact finding, but used existing sources of information, and consolidated pertinent data in a summary report to serve as the basis for the study. EPA decided if additional research was required based on the findings from this effort.

1.3 Project and Task Description

The purpose of this project is to assist EPA in assessing the potential for contamination of USDWs from the injection of hydraulic fracturing fluids into coalbed methane wells, and to determine based on these findings if further study is warranted. EPA will use the information from this study in any regulatory or policy decisions regarding hydraulic fracturing. The first step in investigating the potential for hydraulic fracturing to affect the quality of USDWs was to define mechanisms by which contamination could occur.

EPA defined two hypothetical mechanisms by which hydraulic fracturing of coalbed methane wells could potentially impact USDWs:

1. Direct injection of fracturing fluids into a USDW in which the coal is located, or injection of fracturing fluids into a coal seam that is already in hydraulic communication with a USDW (e.g., through a natural fracture system).

2. Creation of a hydraulic connection between the coalbed formation and an adjacent USDW.

The objective of the project is to consider these two mechanisms, based on existing literature and data, when evaluating whether hydraulic fracturing fluid injection into coalbed methane wells could contaminate USDWs.

Information was collected regarding the geology and hydrogeology of the coalbed methane production regions, the processes used to hydraulically fracture coalbed methane production wells, and the fluids used in the fracturing process. EPA also evaluated water supply incidents possibly related to hydraulic fracturing of coalbed methane production wells. EPA relied on currently available literature and data as the primary source of information for project efforts.

1.4 Quality Objectives and Criteria

To ensure that findings are valid, the following quality assurance questions will be addressed for all sources of data:

- What was the purpose of the study?
- Whose data are they?
- What is their source?
- Are the data reliable?
- Is the interpretation biased?

This Quality Assurance Plan establishes a set of guidelines and general approaches to assess available data and information in a clear, consistent, and explicit manner. Data collection and review according to this process will make conclusions more transparent, and thus more readily understood and communicable to stakeholders.

The objectives of the systematic expert review of data and information are transparency, avoidance of bias, validity, replicability, and comprehensiveness. Following a data and information review protocol can ensure a common understanding of the task and adherence to a systematic approach. The components of this Quality Assurance Plan are as follows:

- Specification of the hypotheses to be addressed;
- Justification of the expertise represented in the expert investigators team;
- Specification of the methods to be used for identification of relevant studies, assessment of evidence of the individual studies, and interpretation of the entire body of available evidence (WHO, 2000);
- Review process; and
- Communication of findings.

Revisions to the Quality Assurance Plan may be necessary as new aspects of the task emerge during the study development process.

1.5 Special Training and Certification

To provide authoritative assessments of data and information, it is important to rely on expert investigators to evaluate the evidence, draw conclusions on the existence of actual and/or potential hazard, and estimate the magnitude of the associated risk. The team of expert investigators, that evaluated the evidence associated with this study, possesses the following qualifications:

- Formal training in basic scientific principles applicable to the project;
- Basic knowledge of the subject or the body of technical information pertaining to it;
- Experience in scientific review of technical data and information;
- Ability to use descriptive and analytical tools appropriately;
- Ability to design studies to test hypotheses;
- Ability to communicate results accurately to decision-makers and stakeholders; and
- Experience coordinating multiple tasks and disciplines to ensure timely and accurate delivery of study components.

The above-listed qualifications ensure that the project team was able to fulfill the objectives of this project.

1.6 Documents and Records

Documents produced for the project and submitted to EPA included the draft and final summary reports (hard copy and digital format). Information and records included in the data report package following completion of the project included:

- Maps (hard copies);
- Scientific literature (hard copies);
- Books (hard copies);
- Database search results (hard copies);
- Logbooks (hard copies); and
- Site visit notes and photographs (hard copies).

All the above-listed materials are maintained by the EPA OGWDW.

2.0 Data Generation and Acquisition

Processes and methods used to collect the data and information must be clear, explicit, and based on valid practice. It is important to adhere to a rigorous and thorough approach to the processes of data collection and data logging.

In Phase I, EPA did not incorporate new, scientific fact finding, but instead used existing sources of information, and consolidated pertinent data in a summary report to serve as the basis for the study. EPA decided if additional research is required based on the findings from this effort. As such, this Quality Assurance Plan does not cover areas of sampling process design, sampling methods, sample handling and custody, analytical methods, quality control, instrument/equipment testing, inspection, and maintenance, instrument/equipment calibration and frequency, and inspection/acceptance of supplies and consumables.

2.1 Non-Direct Measurements

All information summaries and conclusions developed during the course of this project were based on non-direct measurements. Available literature and data were used as the primary source of information for the summary report. An extensive literature search was conducted using the Engineering Index and GeoRef on-line reference databases. Searches will be guided by subject topics and key words within the following areas:

- Hydrogeology of the coalbed methane basins;
- Hydraulic fracturing practices;
- Fracture behavior;
- Hydraulic fracturing fluids and additives; and
- Information regarding water quality incidents.

All search results were printed, catalogued, and surveyed for pertinent journal articles, books and conference proceedings that may contain information meeting the specific data needs of the summary report. Most pertinent articles were acquired from the University of Texas Library in Austin, Texas, as this library's holdings include an extensive collection of oil and gas-related publications. References from the articles were researched and documents relevant to the study were acquired. All papers collected for the study were archived by topic for future reference.

To verify facts extracted from the literature, state regulatory agencies, geological surveys, gas companies, service companies and other relevant organizations were contacted by telephone. Dated telephone logs were used to document all communications. Personal conversations with the employees of the various organizations yielded additional information in the form of literature, figures and maps. These were collected and referenced in conjunction with literature identified in the literature searches.

Internet-based searches were used to locate additional information. Relevant web sites were located using various search engines such as Google™, Yahoo®, and Alta Vista®. More specialized search engines, such as those provided on state geological survey web sites, also were searched. All relevant web sites were logged and referenced appropriately. Efforts were made to acquire the most recent literature. EPA offered state drinking water agencies and the public an opportunity to provide information to EPA on any impacts to groundwater believed to be associated with hydraulic fracturing by a request for public comment. Submissions were reviewed by EPA staff for information pertinent to this report. In addition, a request to provide information and comments regarding incidents of public and private well impacts that could potentially be associated with hydraulic fracturing was published in the July 30, 2001 *Federal Register* (*Federal Register*: July 30, 2001; Volume 66; Number 146; Page 39395-39397).

Details on specific methods used to collect information for each of the major report chapters is included in Chapter 2 of this report.

2.2 Data Management

Gathered information and data was managed to facilitate finding any one piece of gathered data. To achieve this goal, the following data management procedures were used:

- All telephone interviews were recorded in labeled log books;
- All scientific literature, published maps, existing water quality data, conference proceedings, and trade journal articles were filed by coal basin;
- Material safety data sheets and product literature were filed separately;
- Trip folders (to contain notes and photographs) were generated for each site visit;
- Computer database searches were filed separately; and
- Internet websites were referenced in the summary report.

Most data was stored in hard copy format. Wherever possible, data was stored digitally on compact disc.

3.0 Assessment and Oversight

The quality assurance review process provides a means to examine if the results and conclusions are verifiable. The review process results in a determination of whether the conclusions are directly supported by the data or evidence gathered and can be independently validated by others. This quality assurance review process is hierarchical and includes four review levels:

- Weighted emphasis on data based on source;
- Cross referencing of data sources when possible;
- EPA and other federal agencies review; and
- Review by a Peer Review Panel.

EPA's review was accomplished by the Work Assignment Manager in conjunction with other EPA headquarter offices and with other EPA Underground Injection Control regional offices involved with coalbed methane or hydraulic fracturing. Other federal agencies asked to review work products produced by this project, included the United States Geological Survey and the Department of Energy.

EPA assembled a peer review panel consisting of experts in hydraulic fracturing or associated subjects. The panelists provided comments to EPA regarding the sources of data used in the study, the data themselves, and the conclusions drawn from those data.

Comments were requested to assist the investigators in making the study as sound as possible and to ensure that the study met EPA standards for objectivity, evidence, and responsiveness to the study charge. Reviewer comments and objections were preserved and made a part of the record for the study. Issue papers were written containing detailed explanations of responses to comments and objections. Reasons for proceeding or not proceeding with the study were clearly explained.

4.0 Data Validation and Usability

This section describes activities that occurred after the initial collection of data. These activities determined whether or not the gathered data were useful and helpful to the project.

4.1 Data Review, Verification, and Validation

Subsequent to the data logging process, those reports potentially providing useful information underwent a selection process to evaluate quality of the information and usefulness to the study. Systematic evaluation of the validity of individual studies, data, and information included assessment of the following:

- Source of the data and information;
- Qualitative review of the literature;
- Qualitative review of data and information collected;
- Scientific strength of the data and information;
- Geographical, geological, geochemical, spatial, and temporal relevance;
- Relevance to determining baseline conditions;
- Validity of extrapolation to the scope of the study;
- Characteristics of associations, plausibility, alternative explanations;
- Consistency and specificity of the results;
- Scientific uncertainties, limitations, and confounding variables; and

- Other evaluation parameters, as appropriate.

A scale or rating of the data and information with respect to a level of proof required to support conclusions is specifically not proposed as part of this quality assurance process. Establishing a specific level of scientific evidence required to justify a subsequent conclusion would generate significant controversy. Instead, expert judgment was used to evaluate and weigh available data and information.

A variety of technical methods and tools were utilized to sort through the pertinent information and decipher the meaning of the data. These data analysis methods may include:

- Quantitative review of selected data and information collected;
- Tabulating valid data and information;
- Constructing geologic cross sections;
- Evaluating current and historical site operations;
- Review of consistencies between studies;
- Review of sources of discrepancies between studies and information; and
- Other methods/tools as appropriate.

All assumptions were explicitly documented, the basis for the use of any models explained, lack of evidence noted, and scientific uncertainties described as precisely as possible.

4.2 Reconciliation with User Requirements

This sub-section describes how the gathered and validated data and information were used to meet the requirements of this project and EPA.

4.2.1 Drawing Conclusions

Drawing conclusions from evaluated, analyzed, and summarized data and information involve judgment as to whether observations are consistent with the study hypotheses/objectives, or, whether some alternative is suggested. The expert investigators drew upon all evaluated and appropriately summarized data and information; however, no checklist or formula was applied to arrive at conclusions. Instead, critical scientific reasoning and judgment was used to draw conclusions. The process of scientific reasoning and judgment was made explicit by describing and documenting how investigators:

- Assessed completeness of data and information;
- Accounted for lack of evidence and limitations, and impacts on the conclusions;
- Assessed and accounted for bias in original data and/or information;
- Used applicable guidelines and rationales;
- Used any ranges of estimates to arrive at conclusions, where appropriate and;
- Incorporated assumptions into assessments and accounted for the implications of those assumptions in their conclusions.

Conclusions were drawn within the boundaries of the data and the scope of the study. Lack or absence of evidence was addressed. The relative strength or weakness of available information to support conclusions, limitations on where a conclusion may apply, and alternative interpretations of data, was recognized. Any qualification on the use of the data and factors that contribute to uncertainty was conveyed.

Much of the information obtained from public response to the *Federal Register* Notice or from other sources cannot be confirmed through review of peer-reviewed publications or other data sources. However, the information was reviewed and contrasted to evaluate the extent of complaints received and any trends in the complaints within and between individual coalbed methane production basins.

4.2.2 Communication of Findings

This Quality Assurance Plan is reflected in the communication of scientific findings in a clear, accurate, and complete manner to interested parties. Investigators communicated:

- The body of technical information that was considered;

- The manner for evaluating, and drawing conclusions from, collected data and information; and

- Conclusions that address the hypotheses/objectives, supported by the results of data evaluation and analysis.

The use of presentation tools such as charts, diagrams, and computer-generated displays was based on sufficient, valid, and defensible data.

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Attachment 1
The San Juan Basin

The San Juan Basin covers an area of about 7,500 square miles across the Colorado/New Mexico line in the Four Corners region (Figure A1-1). It measures roughly 100 miles long in the north-south direction and 90 miles wide. The Continental Divide trends north-south along the east side of the basin, and land surface elevations within the basin range from 5,100 feet on the western side to over 8,000 feet in the northern part.

The San Juan Basin is the most productive coalbed methane basin in North America. Coalbed methane production in the San Juan Basin totaled over 800 billion cubic feet (Bcf) in 1996 (Stevens et al., 1996). This number rose to 925 Bcf in 2000 (GTI, 2002). The coals of the Upper Cretaceous Fruitland Formation range from 20 to over 40 feet thick. Total net thickness of all coalbeds ranges from 20 to over 80 feet throughout the San Juan Basin. Coalbed methane production occurs primarily in coals of the Fruitland Formation, but some coalbed methane is trapped within the underlying and adjacent Pictured Cliffs Sandstone, and many wells are completed in both zones. Coalbed methane wells in the San Juan Basin range from 550 to 4,000 feet in depth, and about 2,550 wells were operating in 2001 (CO Oil and Gas Conservation Commission and NM Oil Conservation Division, 2001).

1.1 Basin Geology

The San Juan Basin is a typical asymmetrical, Rocky Mountain basin, with a gently dipping southern flank and a steeply dipping northern flank (Figure A1-2) (Stone et al., 1983). The Fruitland Formation is the primary coal-bearing unit of the San Juan Basin and the target of most coalbed methane production. Geologic cross sections showing generalized relationships between the Fruitland Formation and adjacent are shown in A1-4 through A1-6. The Fruitland coals are thick, with individual beds up to 80 feet thick. The Fruitland Formation is composed of interbedded sandstone, siltstone, shale, and coal. The stratigraphy of the Fruitland Formation is predictable throughout the basin, as follows:

- The thickest coalbeds are always found in the lower third of the formation;
- Pictured Cliffs Sandstone occurs immediately below the formation;
- Sandstone content is greater in the lower half; and
- Siltstone and shale predominate in the upper half (Choate et al., 1993).

The San Juan Basin may be subdivided into three unique regions, based on similar geologic, hydrologic, and production characteristics (Figure A1-7). These regions are denoted as Area 1, Area 2, and Area 3, and are described in more detail below (Kaiser and Ayers, 1994).

Area 1 consists of the northwestern quarter of the basin. Area 1 is characterized by the thickest (>20 feet) and highest-rank coal deposits in the San Juan Basin (Ayers et al., 1994). Most wells produce more than 1,000 cubic feet per day and several wells produce more than 15,000 cubic feet per day. Almost 90 percent of total methane production from the Fruitland Formation comes from three fields in a region of Area 1 known as the "Fairway" (Young et al., 1991; Ayers et al., 1994). Area 1 is an area of active recharge and in most places is hydrostatically over-pressured (greater than 0.50 pounds per square inch per foot). Wells in Area 1 usually produce moderate to large volumes of water, some of which meet the quality criteria of less than 10,000 milligrams per liter (mg/L) total dissolved solids (TDS) for an underground source of drinking water (USDW) (Kaiser et al., 1994).

Area 2 (the west-central region of the San Juan Basin) is hydrostatically under-pressured (0.30 to 0.50 pounds per square inch per foot) and is an area of regional groundwater discharge. Coalbeds are usually 7 to 15 feet thick, and occur primarily in northwest-trending belts that extend to the southwestern margin of the basin. Methane production from wells can be more than 100 thousand cubic feet per day, and a few wells produce 200 to 500 thousand cubic feet per day. Methane gas is produced water-free in this area as a consequence of the hydrostratigraphy and trapping mechanisms (Kaiser and Ayers, 1994). Additionally, Kaiser and Ayers (1994) suggest that water may be less mobile in the hydrophilic and low permeability coals. The Fruitland Formation in this area where it is under-pressured generally shows the presence of saline-type waters (Kaiser et al., 1994) that most likely have TDS concentrations greater than 10,000 mg/L, which does not meet the criteria for a USDW.

Area 3, the eastern region of the San Juan Basin, is hydrostatically under-pressured, and features low permeability and low hydraulic gradient, which suggests slow water movement within most of the aquifer. Only a few coalbed gas wells are located in this part of the basin, and they produce up to 8,000 cubic feet of methane per day, with little or no water content (Kaiser and Ayers, 1994). Produced waters from the Fruitland Formation in most of Area 3 have a high-salinity, resembling seawater (Kaiser and Ayers, 1994) in which TDS are too high to meet the water quality criteria of a USDW. However, along the southern margin of Area 3, TDS concentrations are less than 10,000 mg/L (Kaiser et al., 1994).

1.2 Basin Hydrology and USDW Identification

Tertiary sandstones and Quaternary alluvial deposits are present at the surface over much of the basin interior. These serve as the primary drinking water aquifers in the basin (Figure A1-2), and produced 55 million gallons per day in 1985 (Wilson, 1986). Cretaceous sandstones are an important source of water on the basin's periphery (Choate et al., 1993). The Paleocene Ojo Alamo Sandstone yields as much as 30 gallons per minute of potable water (Hale et al., 1965) and is mentioned as one of the primary drinking water aquifers of the region (Brown and Stone, 1979). Cleats and larger fractures in the Fruitland coals and the presence of interbedded permeable sandstones make the Fruitland Formation an aquifer and source of drinking water along the northern margin of the basin where TDS in the groundwater are less than 10,000. In most of Area 1, both the Fruitland Formation and the underlying upper Pictured Cliffs Sandstone act as a single hydrologic unit (Kaiser et al., 1994). The Fruitland and upper Pictured Cliffs Sandstone aquifer is underlain and confined by the low-permeability main Pictured Cliffs Formation and is overlain and partly confined by the Kirtland shale, which is up to 1,000 feet thick in the central basin. Overlying the Kirtland Formation is the Ojo Alamo Sandstone, (Figures A1-4, A1-5 and A1-6) which has been suggested as a possible source of groundwater for the municipality of Bloomfield (Stone et al., 1983). At Bloomfield, the coal and gas bearing Fruitland is separated from the Ojo Alamo aquifer by the Kirtland shale.

In the northern part of the basin, the Fruitland Formation and the underlying upper Pictured Cliffs Sandstone can be considered a single hydrogeologic unit on a regional scale because they exhibit the same hydraulic head and water quality characteristics and are the source of both the water and gas in the Pictured Cliffs sand tongues (Ayers and Zellers, 1994; Ayers et al., 1994). At the local scale, however, the two formations appear to exhibit poor hydraulic continuity, as evidenced by areas of over-pressuring (greater than 0.5 pounds per square inch per foot), abrupt changes in potentiometric surface (Figure A1-8), and upward flow (Kaiser et al., 1994). Discrete flow within individual units here is likely due to pinch out of thick, laterally extensive coal seams and truncation and offset of the beds by faults.

In general, groundwater is recharged along the Fruitland outcrops at the elevated, west, northern, and northwestern margins of the basin, and lateral flow converges primarily from the northeast and southeast toward upward discharge to the San Juan River valley (Kaiser et al., 1994). In the north, the Fruitland and upper Pictured Cliffs Sandstone aquifer system is confined by the overlying Kirtland shale, but it is poorly confined by the Kirtland in the central and southern portions of the basin. Water from the Fruitland discharges in the western part of the basin and migrates upward across the Kirtland shale into the Animas and San Juan Rivers (Stone et al., 1983). Generalized groundwater movement in the Fruitland system is shown in cross-section and plan view in Figures A1-9 and A1-10 (Kaiser and Swartz, 1988). The results of groundwater flow modeling for the entire basin (Kaiser et al., 1994) are shown in Figure A1-11.

In most of Area 1, the Fruitland system produces water containing less than 10,000 mg/L TDS, the water quality criteria for a USDW. Groundwater is usually freshest at the outcrop in recharge areas. The water dissolves salts and mixes with formation water as it flows, and the groundwater becomes increasingly saline as distance from the recharge source increases. The presence of low-salinity water at given locations in the San Juan Basin usually marks close proximity to the recharge source or the most permeable flow paths and implies a dynamic, active aquifer system (Kaiser et al., 1994). Figure A1-12 shows the chloride concentration of groundwater in the Fruitland Formation, and indicates that water nearest the northern recharge areas has a low dissolved solids and chloride content. Kaiser et al. (1994) reported that wells in the northern part of Area 1 produced water containing from 180 to 3,015 mg/L TDS. This was found to be the case over large portions of Area 1, especially within freshwater plumes resulting from areas of high permeability or fracture trends (Kaiser and Swartz, 1990; Oldaker, 1991).

Kaiser et al. (1994) conducted a water-quality sampling program in the San Juan Basin. Analyses taken from Fruitland coal wells in Area 1 show that the majority of wells (16 of 27 wells) produce water containing less than 10,000 mg/L TDS, (Figures A1-13a and A1-13b), although some nearby wells thought to be in less permeable zones produce water with higher TDS concentrations up to 23,000 mg/L (Kaiser et al., 1994). The boundary between waters with more and less than 10,000 TDS has not been published. Another group of wells throughout the same area was also sampled, but these wells were completed (constructed) in the adjacent and underlying Pictured Cliffs Sandstone bodies, which are in hydrologic communication with the Fruitland system (Kaiser et al., 1994).

Although from the above information it would seem that the Fruitland would be classified a USDW, the following additional information about disposal of brackish water produced along with the methane would seem to indicate that most of the water in the Fruitland would not meet the TDS criteria for USDW. Coalbed methane wells in the San Juan Basin produced from 0 to over 10,500 gallons of water per day, which contain from less than 300 mg/L TDS to over 25,000 mg/L (Kaiser et al., 1994; Kaiser and Ayers, 1994). Brackish water of various TDS concentrations and brine are produced in the over-pressured Area 1 of the basin while virtually no water is produced from coalbed methane wells in Areas 2 and 3 of the basin. Cox (1993) reported "Water disposal in the San Juan basin is a significant, long-term issue." In 1992, coalbed methane wells produced over 5 million gallons of water per day, and production was expected to increase to over 7.5 million gallons per day by 1995 (Cox, 1993). Produced water is disposed of by means of evaporation ponds, or, more commonly, by Class II injection into deeper zones such as the Entrada and Bluff sandstones, Morrison Formation, and Mesa Verde sandstone (Kaiser and Ayers, 1994). The authors estimated that injection wells cost up to $2 million each and Cox (1993) reported that 51 of them had been constructed in the basin by 1993.

Area 2 is primarily an area of groundwater discharge. The Fruitland coals and Pictured Cliffs Sandstone in Area 2 are in hydraulic communication and behave as a single aquifer. The aquifer is under-pressured (less than 0.50 pounds per square inch per foot), transmits groundwater from the northeast and southeast, and eventually discharges to the

Animas and San Juan rivers. The TDS of most samples from Area 2 ranges from 10,000 to 16,000 mg/L (Kaiser et al., 1994).

The Fruitland system in most of Area 3 contains slow-moving water with salinity approximately equal to that of seawater, greater than 25,000 mg/L TDS, (Kaiser and Ayers, 1994). In Area 3, the Fruitland and Pictured Cliffs are separate, confined aquifers. In the southeastern one-third of Area 3, the Kirtland shale is absent because of Tertiary-age erosion, and the Fruitland and Ojo Alamo Sandstone could be in hydraulic communication with one another (Figure A1-6). In this area Tertiary rocks, including the Ojo Alamo, are mapped by the United States Geological Survey (Figure A1-14) as an aquifer having water with TDSs ranging from 500 to 1,000 mg/L (Lyford, 1979).

At the basin's southern margin in Area 3, downward flow occurs from the Ojo Alamo through the Kirtland shale to the poorly confined Fruitland aquifer through which it then moves southward to outcrops at a lower elevation and northward to the San Juan River Valley (Kaiser et al., 1994) (Figure A1-11). Twenty-four of 26 water samples from the Fruitland/Pictured Cliffs aquifer system in the south margin of the basin reported by Kaiser and Swartz (1994) had less than 9,000 mg/L TDS (Figure A1-13e & A1-13f). Groundwater in the Fruitland Formation at the southern margin of the basin has less than 10,000 mg/L TDS because most recharge there comes from above the Kirtland formation, rather than from southward throughput from the Fruitland Formation.

1.3 Coalbed Methane Production Activity

Coalbed methane production occurs primarily in coals of the Fruitland Formation. However, some methane is absorbed in the underlying and adjacent Pictured Cliffs Sandstone, therefore many wells are completed in both zones. About 2,550 wells were operating in the San Juan Basin in 2001 (CO Oil and Gas Conservation Commission and NM Oil Conservation Division, 2001). All wells are vertical wells that range from about 500 to 4,000 feet in depth, and were drilled using water or water-based muds. Almost every well has been fracture-stimulated, using either conventional hydraulic fracturing in perforated casing or cavitation cycling in open holes (Palmer et al., 1993b). Total gas production was 925 Bcf in 2000 (GTI, 2002).

Cavitation cycling is a fracturing method unique to a small area of the north-central San Juan Basin called the "Sweet Spot," or Fairway, of Area 1 (Figure A1-15). Almost half of all San Juan wells are located within the Fairway area and utilize open-hole completions (no casing across the production interval) and cavitation cycling. Cavitation cycling is used in this area because coals are: 1) very thick (individual coals over 40 feet thick); 2) hydrostatically over-pressured (0.5 to 0.7 pounds per square inch per foot); and 3) relatively more permeable than the rest of the basin (and coals in other basins) (Palmer et al., 1993b). This method uses several mechanisms to link the wellbore to the coal fracture system. Cavitation cycling:

- Creates a physical cavity in the coals of the open-hole section (up to 10 feet in diameter);

- Propagates a self-propping, vertical, tensile fracture that extends up to 200 feet away from the wellbore (parallel to the direction of least stress); and

- Creates a zone of shear stress-failure that enhances permeability in a direction perpendicular to the direction of least stress (Palmer et al., 1993a; Khodaverian and McLennan, 1993) (Figure A1-16).

Cavitation is accomplished by applying pressure to the well using compressed air or foam, and then abruptly releasing the pressure. The over-pressured coal zones provide a pressure surge into the wellbore (a “controlled blowout”), and the resulting stress causes dislodgement of coal chips and carries the chips up the well. These cycles of pressure and blowdown are repeated many times over a period of hours or days, and the repeated, alternating stress-shear failure in the coal formation creates effects that extend laterally from the wellbore (Kahil and Masszi, 1984). The resulting vertical fracture is tensile in origin, that is, it results from a “pulling” force rather than the compressive forces that create conventional hydraulic fractures. Because the fracture is tensile in origin, the height of the fracture does not usually extend out of the target coal seam (Logan et al., 1989).

Wells outside the Fairway area utilize cased-hole, perforated completions that employ conventional hydraulic fracturing (Holditch, 1990). Palmer et al. (1993a) reported that hydraulic fracturing in the San Juan Basin uses between 55,000 to 300,000 gallons of stimulation and fracturing fluids and between 100,000 to 220,000 pounds of sand proppant. In the San Juan Basin, geologic conditions in conjunction with fracturing techniques usually produce vertical fractures much longer than they are high, for example, up to 400 feet radially and less than 150 feet high (e.g., Colorado 32-7 No. 9 well, La Plata County, CO; Mavor et al., 1991). The primary reasons for the controlled height of San Juan coalbed fractures are the thickness and close spacing of coal seams (obviating the need for excessive height), and the presence and petro-physical properties of the overlying Kirtland shale (which prevents inadvertent fracture excursion out of the Fruitland) (Jeu et al., 1988; Logan et al., 1989; Palmer and Kutas, 1991). Holditch (1993) reports that where the coal seam is not overlain by shale, hydraulic fractures in the San Juan Basin can grow into overlying beds.

Fassett (1991) found that coalbed methane could migrate into overlying USDWs near the northern outcrop, in areas where confining shale layers are absent. Because of these factors, hydraulic fracturing in the San Juan Basin may indirectly impact overlying USDWs near the Fruitland outcrop at the basin margins, where USDWs are in closer proximity and the Kirtland shale may be eroded. Near the northern and northwestern recharge zones, groundwater usually contains less than 3,000 mg/L TDS (Kaiser et al., 1994; Cox et al., 1995).

Fracturing and stimulation fluids utilized in the northern San Juan Basin include (Figure A1-17 and Table A1-1):

- Hydrochloric acid (12% to 28% HCl);
- Plain water;
- Slick water (water mixed with solvent);
- Linear gels (water and a thickener such as guar-gum or a polymer);
- Cross-linked gels with breakers (gels with additives to prevent fluid leak-off from the fracture, and "breaker" chemicals to reduce viscosity so that the gel can be produced back from the well after treatment); and
- Nitrogen and CO_2 foam (75 percent gas, 25 percent water or slick water, plus a foaming agent) since about 1992 (Harper et al., 1985; Jeu et al., 1988; Holditch et al., 1989; Palmer et al., 1993a; Choate et al., 1993)

Oilfield service companies supply the stimulation fluid used to fracture the well as part of the service. The chemical composition of many fracturing fluids may be proprietary, and EPA was unable to find complete chemical analyses of any fracturing fluids in the literature. Table A1-1 presents some data from the literature concerning the general chemical makeup of common San Juan fracturing fluids (Economides and Nolte, 1989; Penny et al., 1991). In addition, most gel fluids utilize a breaker compound (usually borate or persulfate compounds or an enzyme, at 2 pounds/1,000 gallons) to allow post-treatment thinning and easier recovery of gels from the fracture (e.g., Jeu et al., 1988; Palmer et al., 1993a; Pashin and Hinkle, 1997).

Many of the compounds listed in Table A1-1 are quite hazardous in their undiluted form. However, these compounds are substantially diluted prior to injection. Coalbed methane development by fracturing, and stimulation in the San Juan Basin are regulated by the Colorado Oil and Gas Conservation Commission and the New Mexico Oil and Gas Board. Based on an analysis of current regulations, neither agency regulates the type or amount of fluids used for fracturing (Colorado State Oil and Gas Board Rules and Regulations 400-3, 2001; New Mexico Energy, Minerals and Natural Resources Department, Oil Conservation Division Regulations Title 19, Chapter 15, 2001).

About half of the coalbed methane wells in Area 1 are located in the Fairway zone and feature "cavitation-cycling" completions (Palmer et al., 1993a) (Figure A1-15). Therefore, about half of the wells in Area 1 have probably been stimulated using conventional fracture treatments. Based on the well density of Area 1 in 1990 (Figure A1-18) compared to the 2001 well population (2,550 wells), it is estimated that between 700 and 1,000 coalbed methane wells have been fracture-stimulated in the USDW of Area 1.

It has been shown that methane can migrate from gas wells into aquifers along the northern margin of the basin, but this condition was remediated with improved gas well construction (Cox et al., 1995). In addition, wells completed in other aquifers in the outcrop area have been shown to produce water chemically and isotopically similar to Fruitland wells, implying communication between the formations (Cox et al., 1995).

1.4 Summary

Coalbed methane development and hydraulic fracturing in some of the northern portions of the San Juan Basin take place within a USDW. The waters of the Fruitland-upper Picture Cliffs aquifer and producing zone in Area 1 usually contain less than 10,000 mg/L TDS. Most waters in the northern half of Area 1 contain less than 3,000 mg/L, and wells near the outcrop produce water that contains less than 500 mg/L.

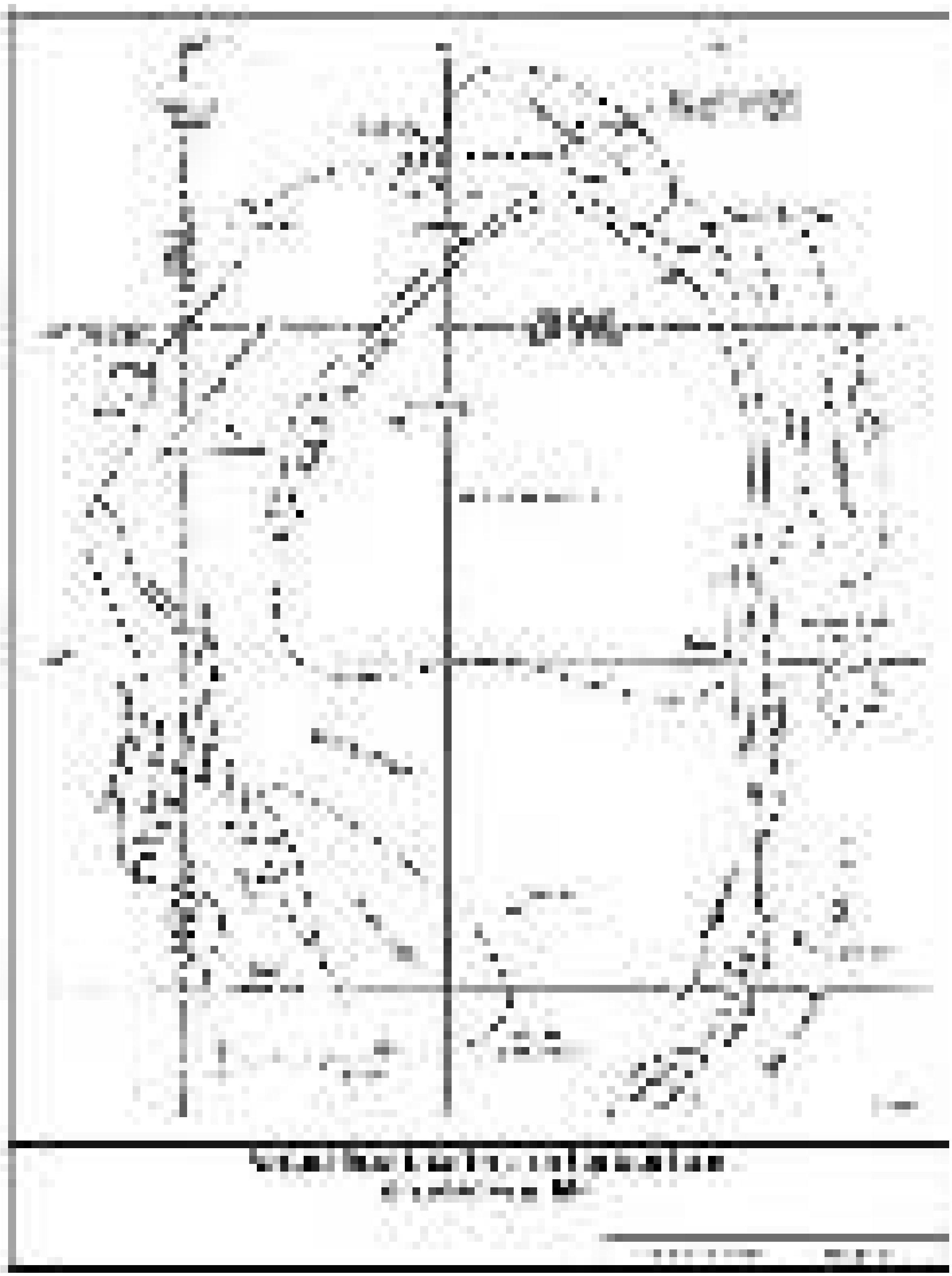
Each fracture stimulation treatment may inject, on average, approximately 55,000 to 300,000 gallons of stimulation and fracturing fluid per treatment. There are no state controls on the type, composition, or volume of fracturing fluid employed in each well or treatment. In contrast to conventional gas formations, the anisotropic nature of fracture permeability, the volume of treatment fluids employed, and the height and proppant distribution in coalbed fractures may prevent the effective recovery of fracturing fluids during subsequent production.

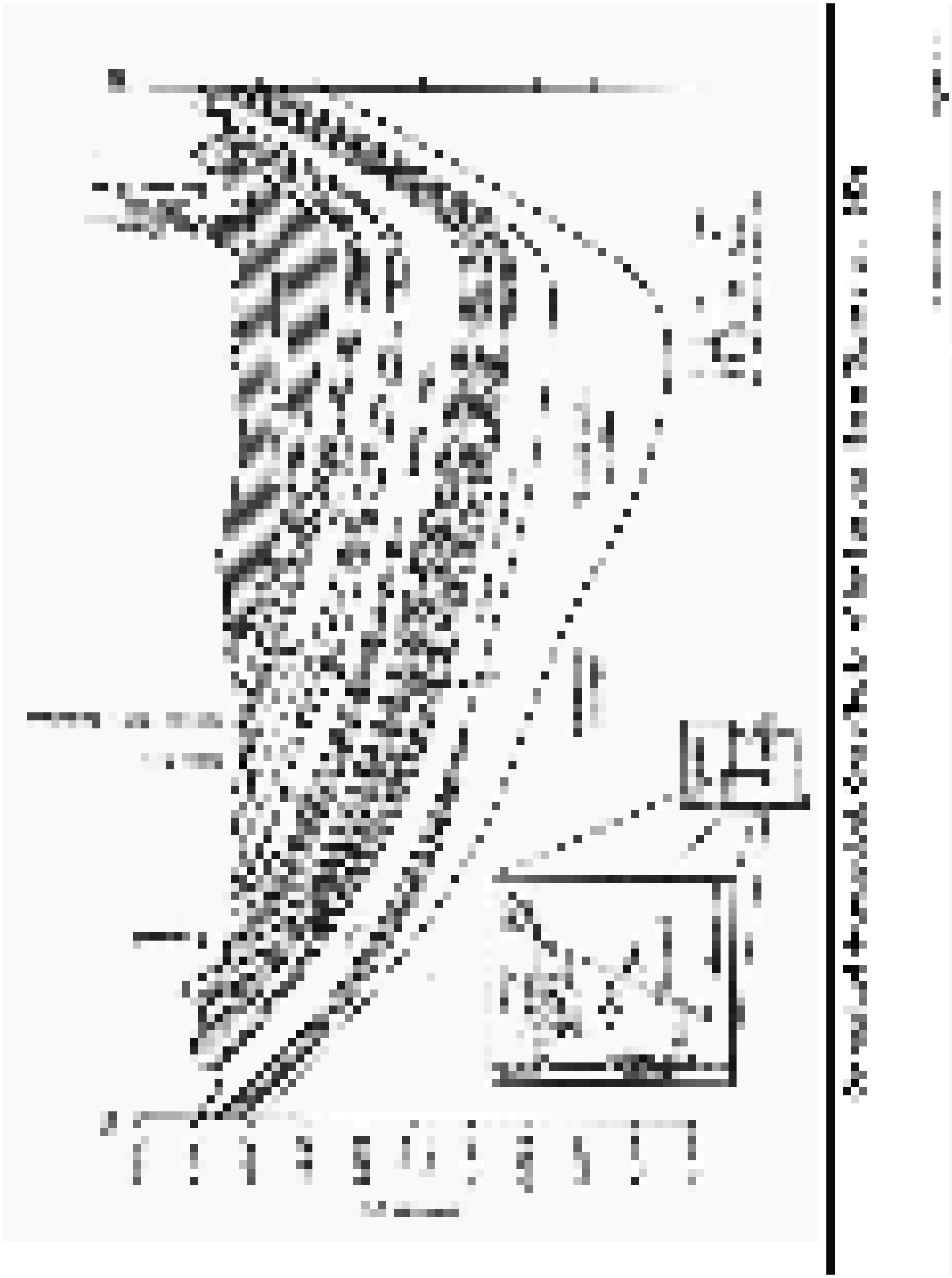
The potential for fracturing to cause or allow degradation of water in aquifers adjacent to the producing zones seems relatively remote in the currently active gas producing fields, but the potential for such degradation varies in different parts of the basin. It has been shown that methane can migrate from gas wells into aquifers along the northern margin of the basin, but this condition was corrected with improved gas well construction. There is little potential for fracturing to create communication between the Fruitland-upper Picture Cliffs aquifer and the Ojo Alamo aquifer over much of the basin because the aquifers are separated by the poorly permeable Kirkland shale. However, the Kirkland varies greatly in thickness and forms a leaky hydrogeologic barrier when it is thinner. In the eastern part of the basin, the Kirkland Formation has been eroded and the Ojo Alamo lies disconformably and directly upon the Fruitland Formation, potentially allowing fracturing to cause hydraulic communication between the saline waters of the Fruitland and the fresh waters (500 to 1,000 mg/L) of the Ojo Alamo.

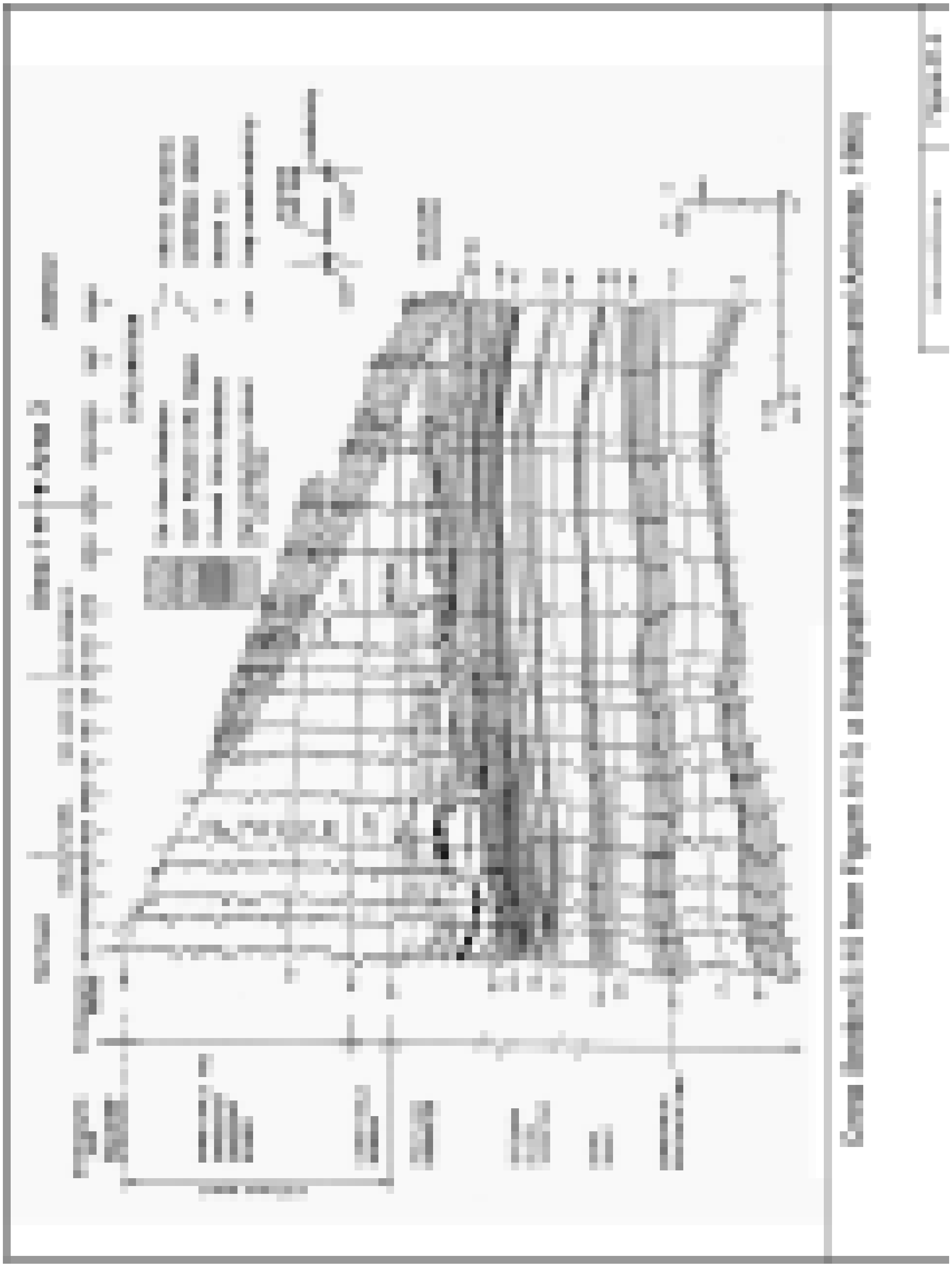
Table A1-1. Chemical Components of Typical Fracture/Stimulation Fluids Used for San Juan Coalbed Methane Wells

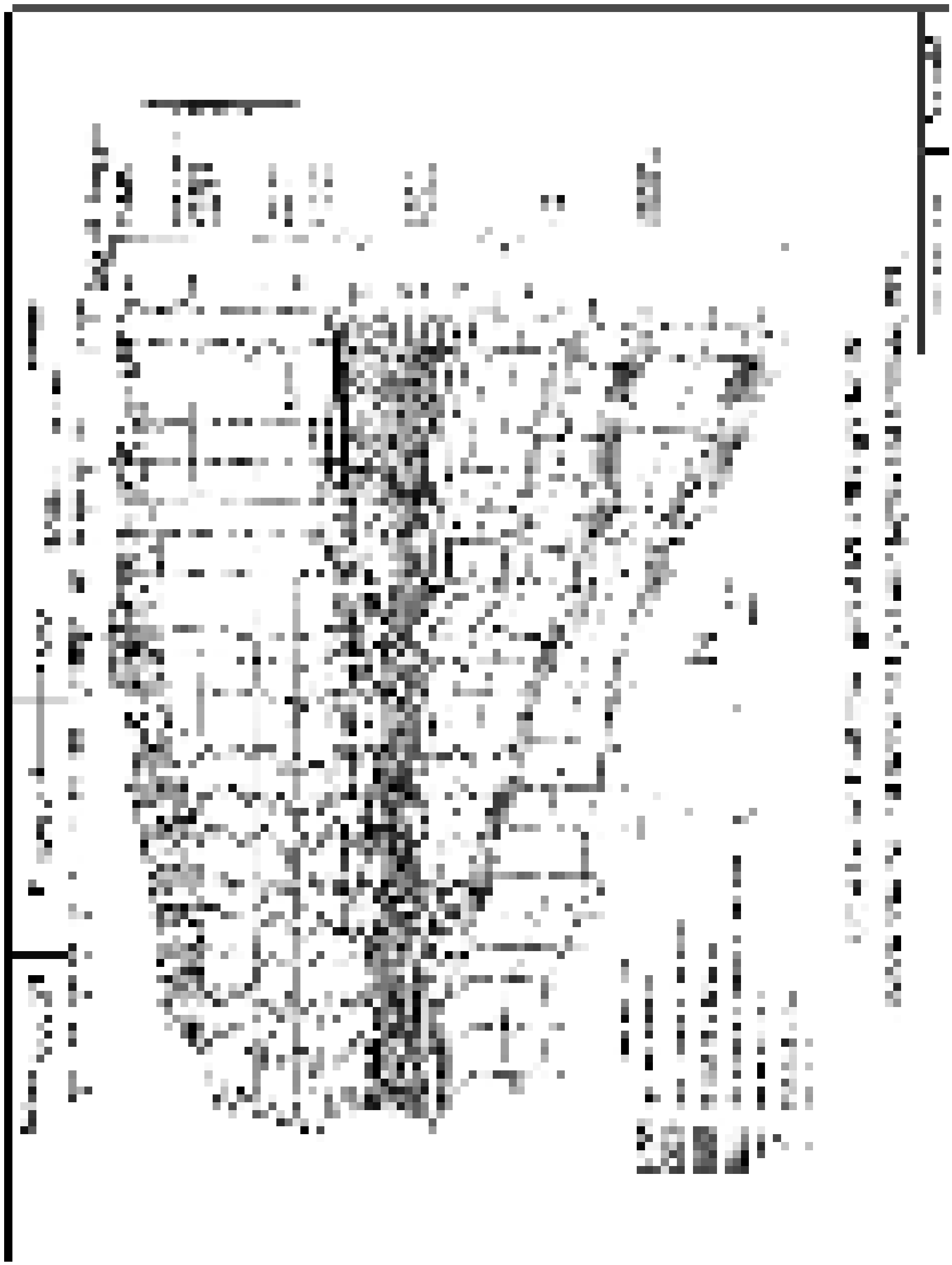
Type of Stimulation Fluid	Composition	pH
Hydrochloric acid	12% to 28% HCl water solution	<1-3
"Slick" water	miscible or immiscible solvent as viscosity reducer (% unknown)	NA
Diesel oil	NA	NA
Nitrogen and CO_2 foam	75 % gas, 25 % water or slick water, plus a foaming agent)	NA
Gels[1]		
R-F	3% resorcinol, 3% formaldehyde, 0.5% KCl, 0.4% $NaHCO_3$	6.5
Pfizer Flocon 4800	0.4% xanthan, 154 ppm Cr^{3+} (as $CrCl_3$), 0.5% KCl	4.0
Marathon MARCIT	1.4% polyacrylamide (HPAM), 636 ppm Cr^{3+} (as acetate), 1% NaCl	6.0
DuPont LuDox SM	10% colloidal silica, 0.7% NaCl	8.2
CPAM crosslinked with Pfizer Floperm 500	0.4% cationic polyacrylamide (CPAM), 1520 ppm glyoxal 2% KCl	7.3
Drilling Specialties HE-100 Crosslinked	0.3% HPAM-AMPS, 100 ppm Cr^{3+} (as acetate), 2% KCl	5.0
Dowell YF-230	Hydroxypropylguar (HPG) x-linked with borate, persulfate with amine	NA

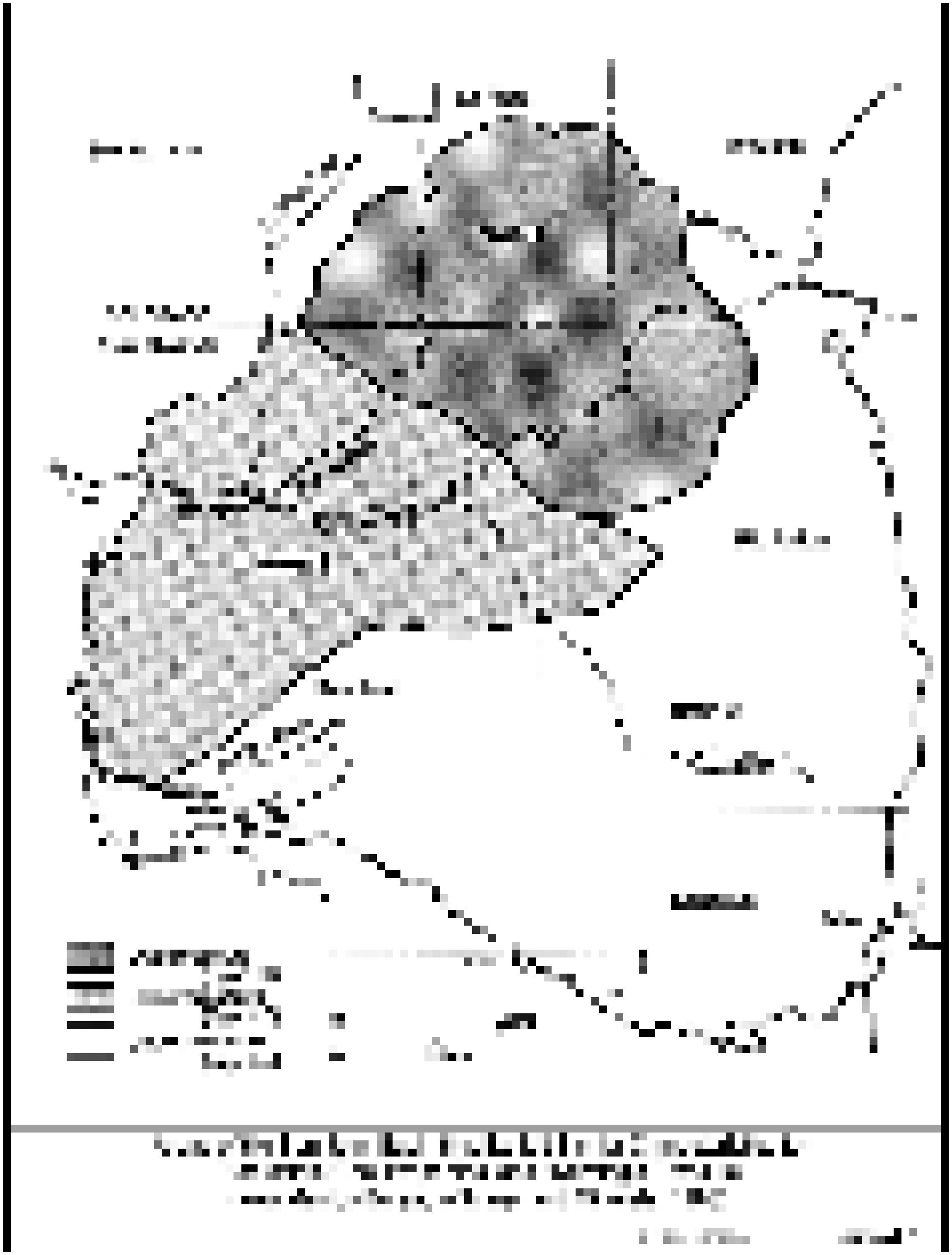
[1] Gels are typically mixed at a ratio of 40 lbs. per 1000 gal. water; compositions shown are "as mixed".

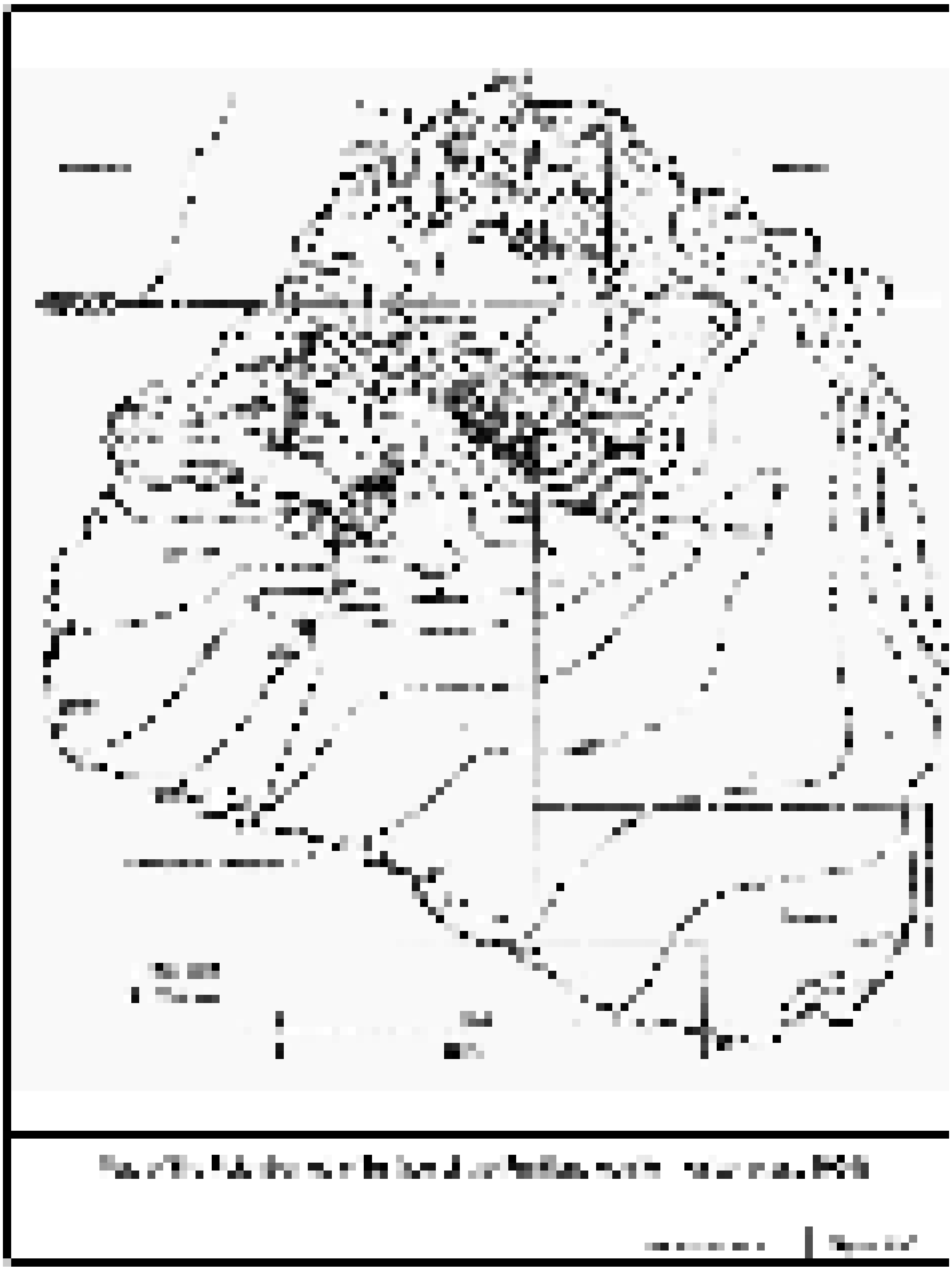


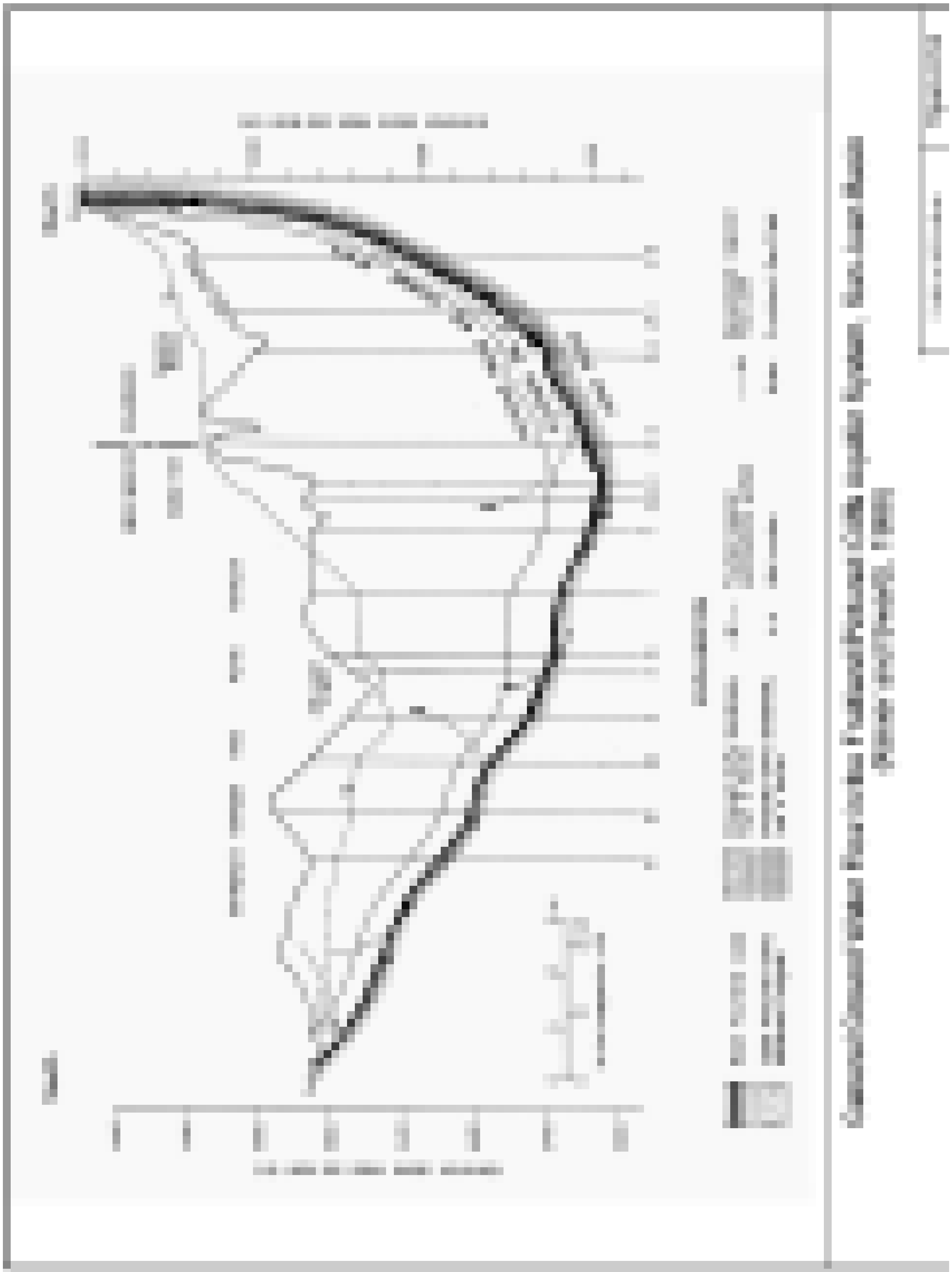


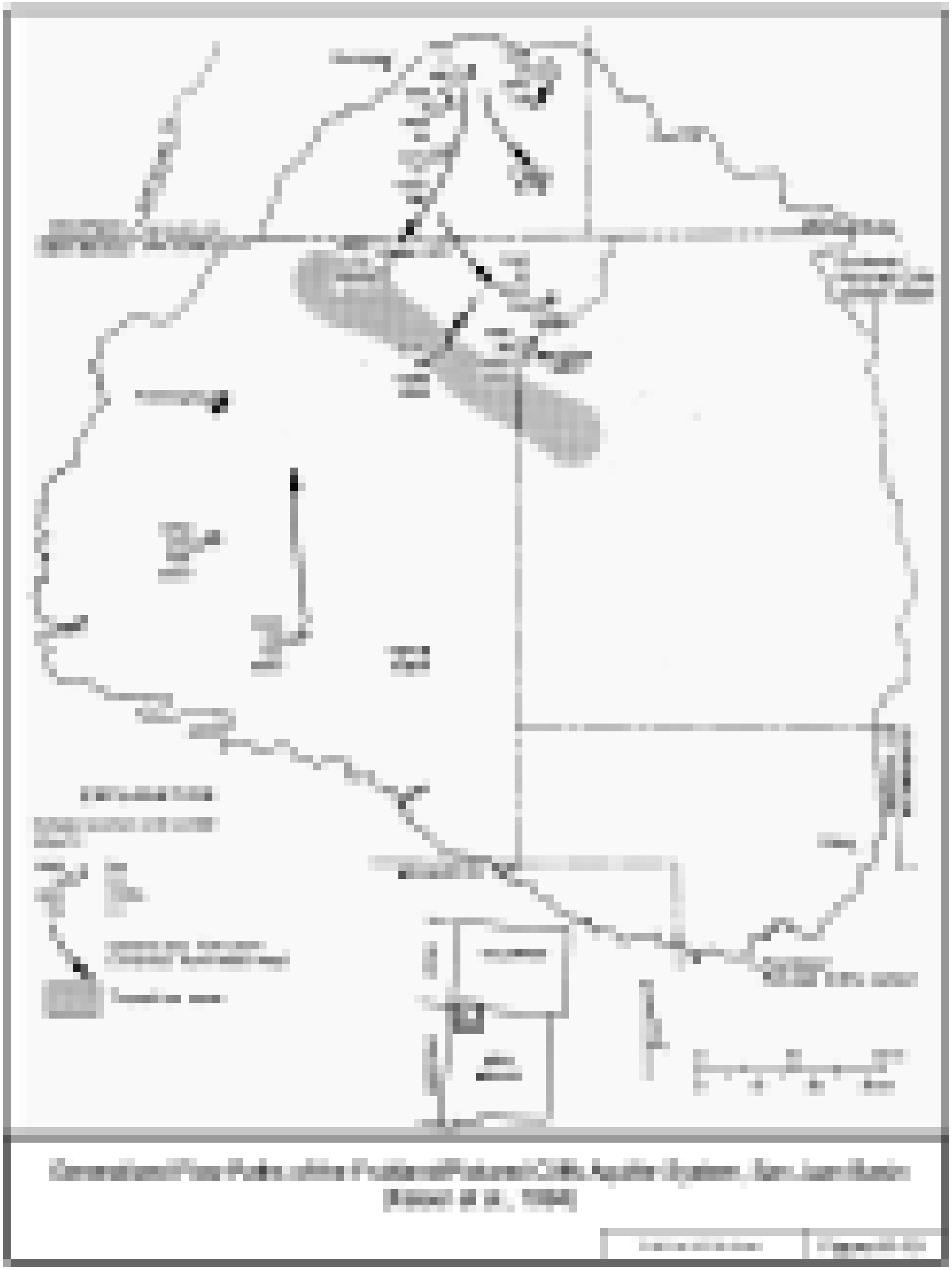


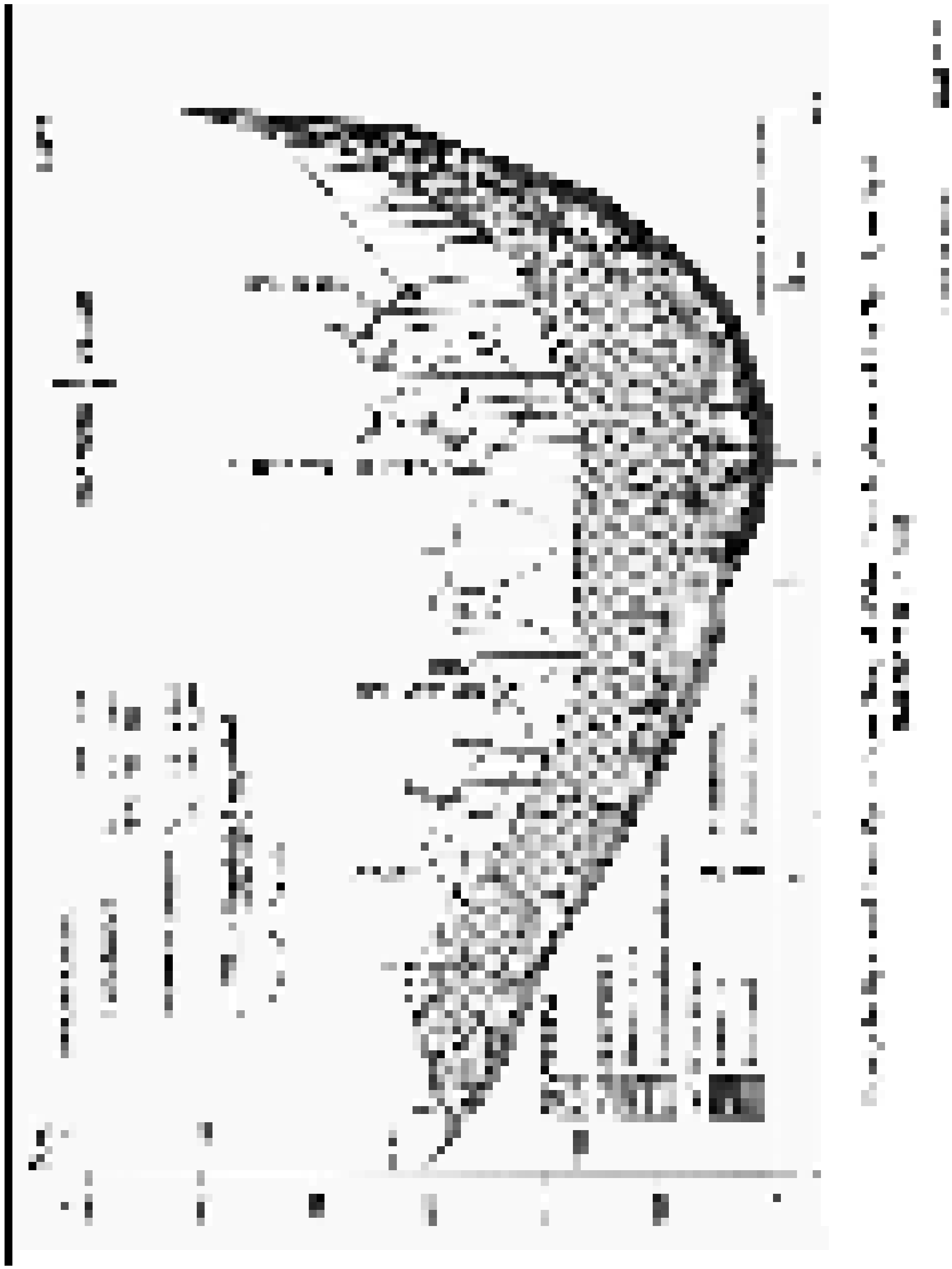


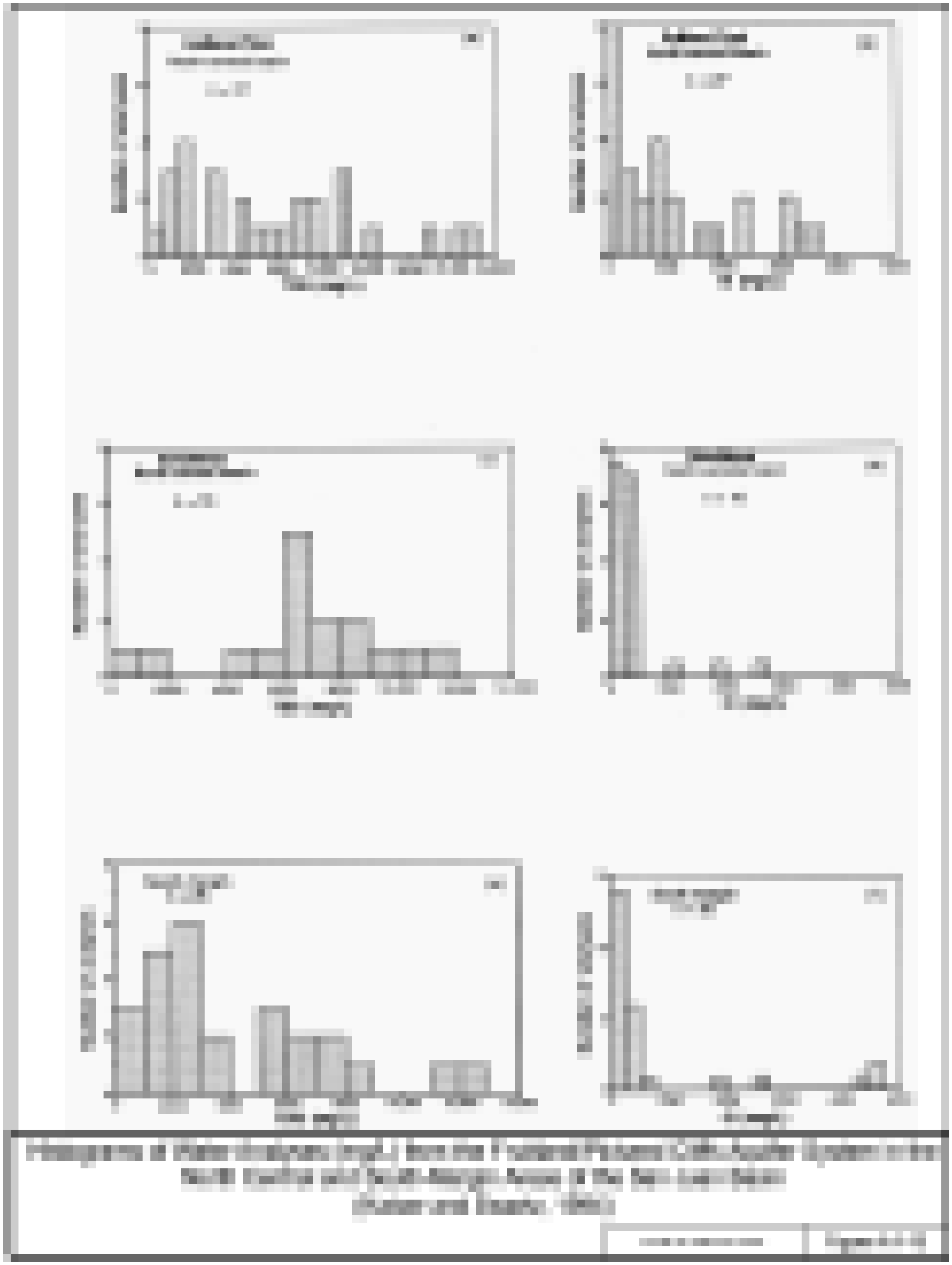


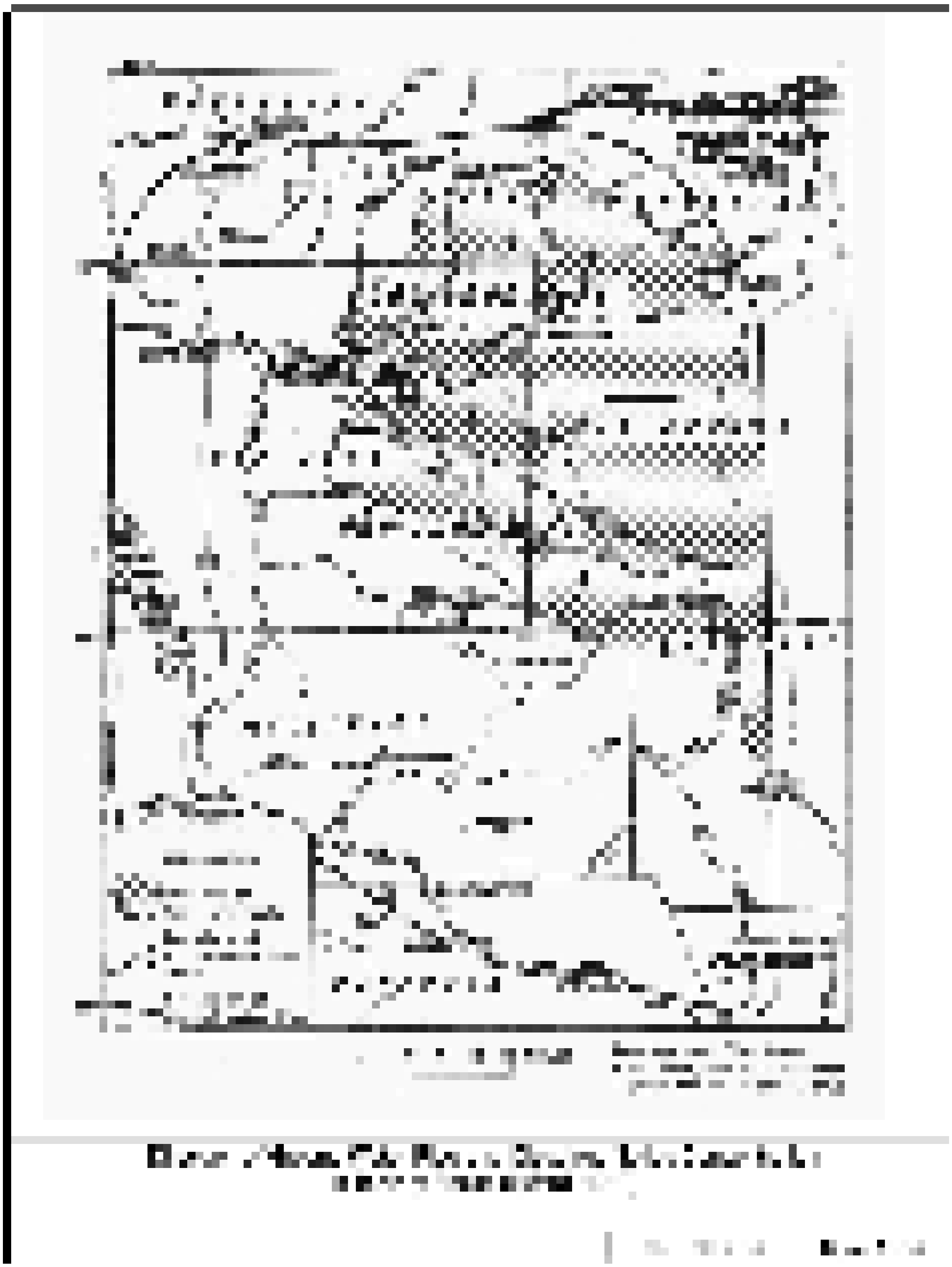


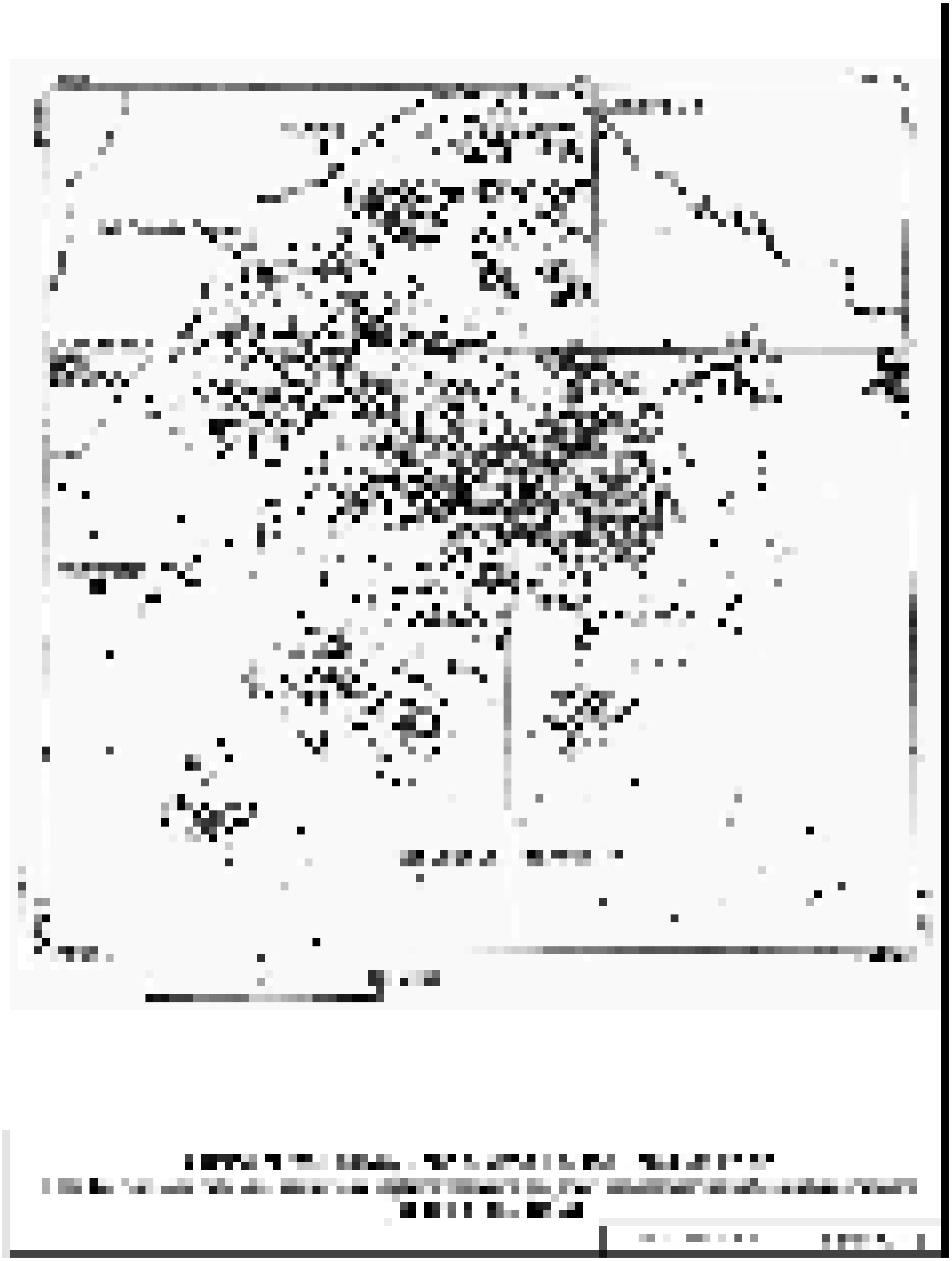


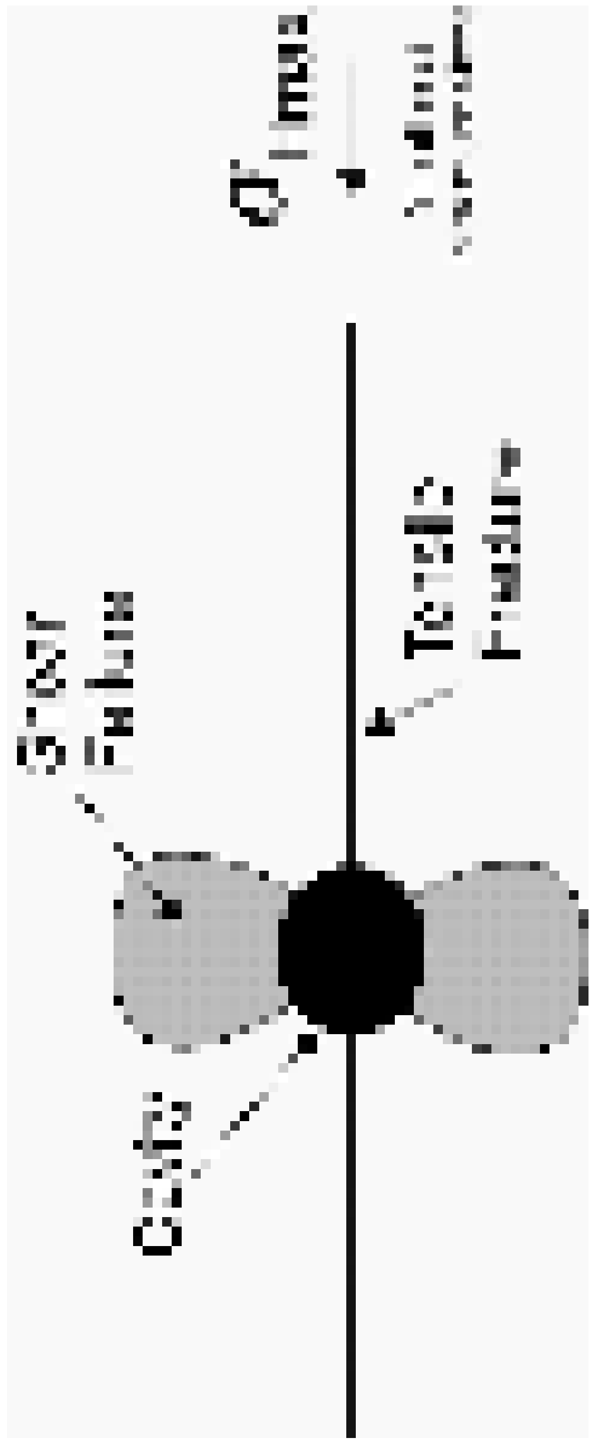








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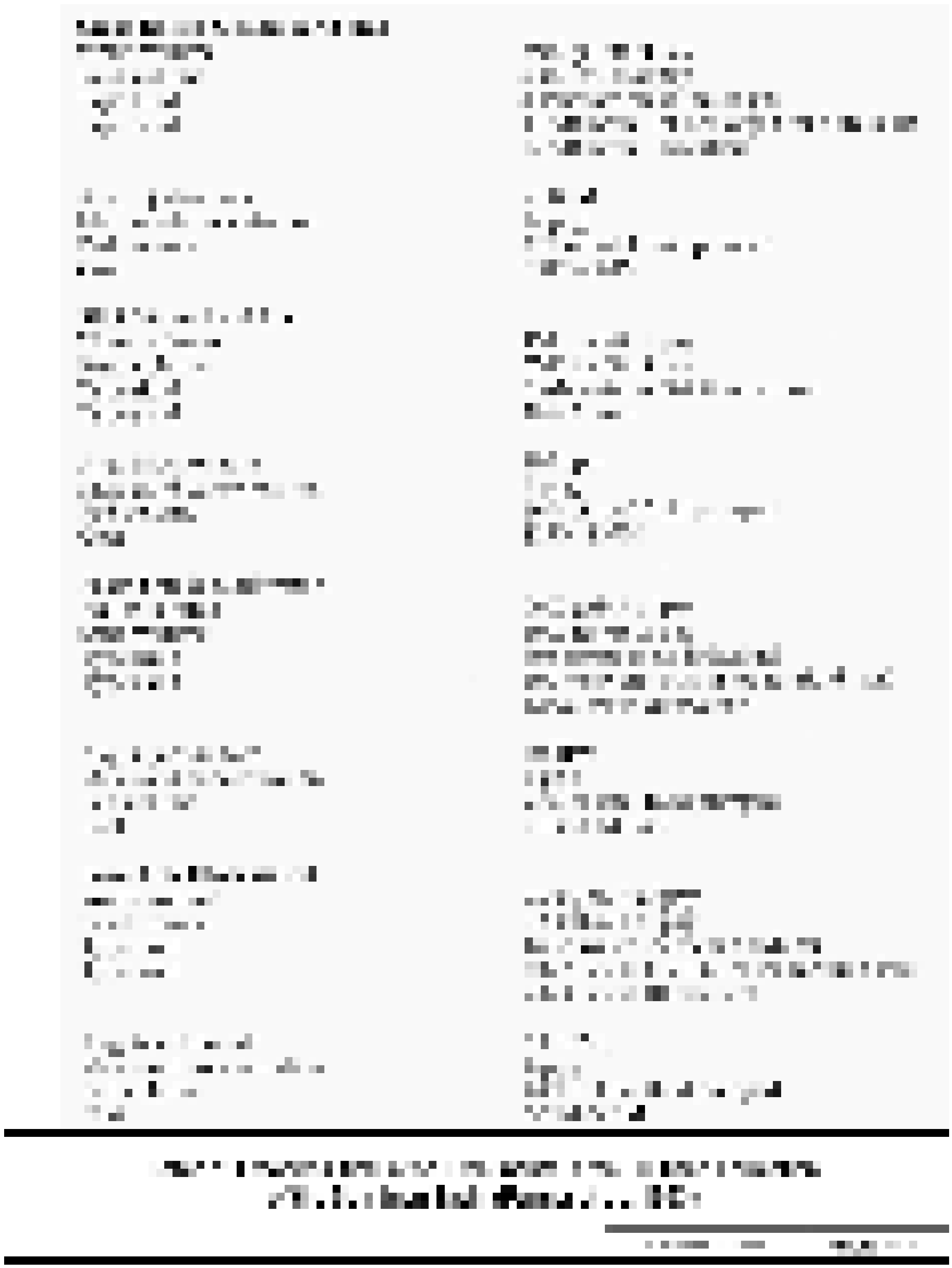
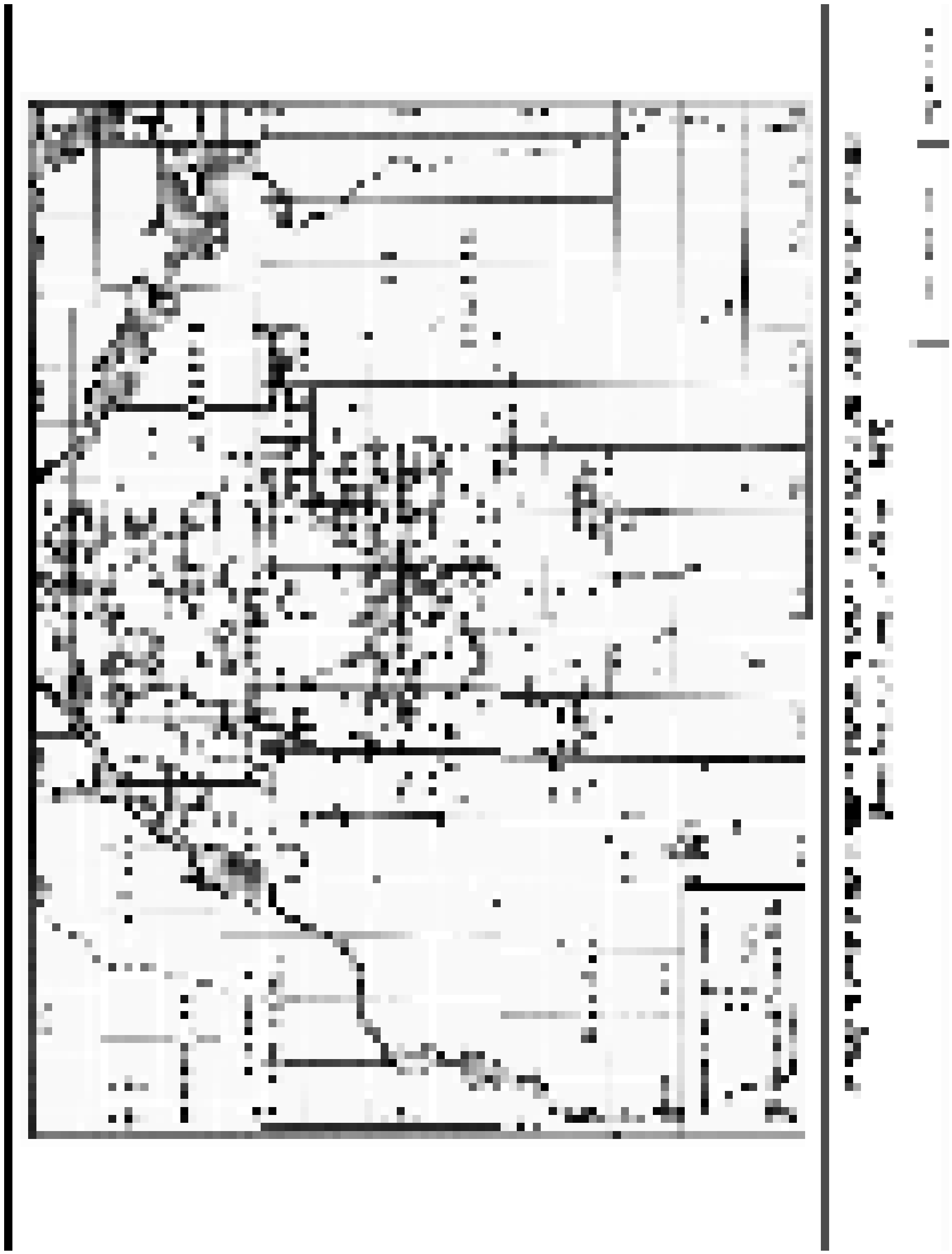

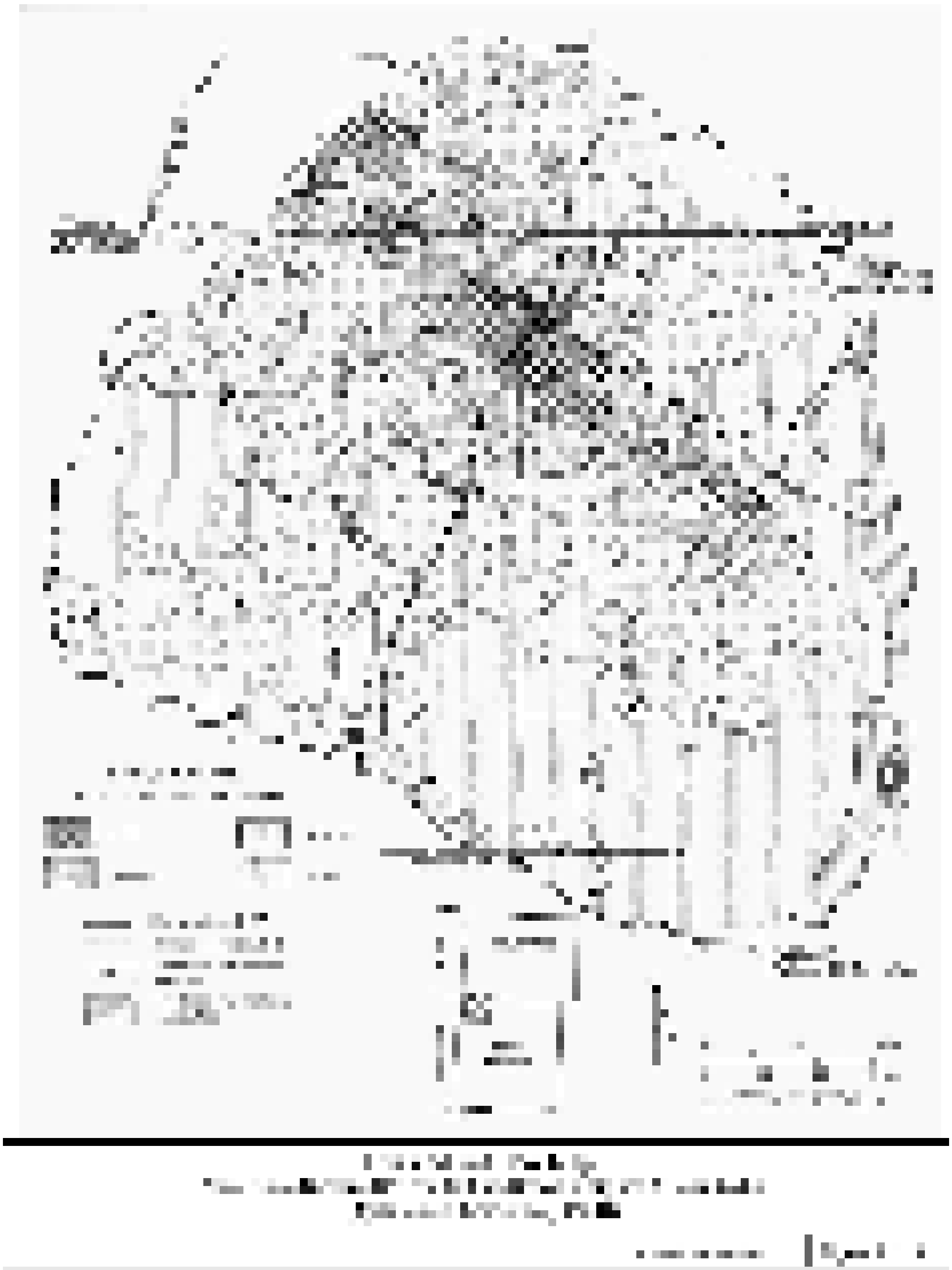

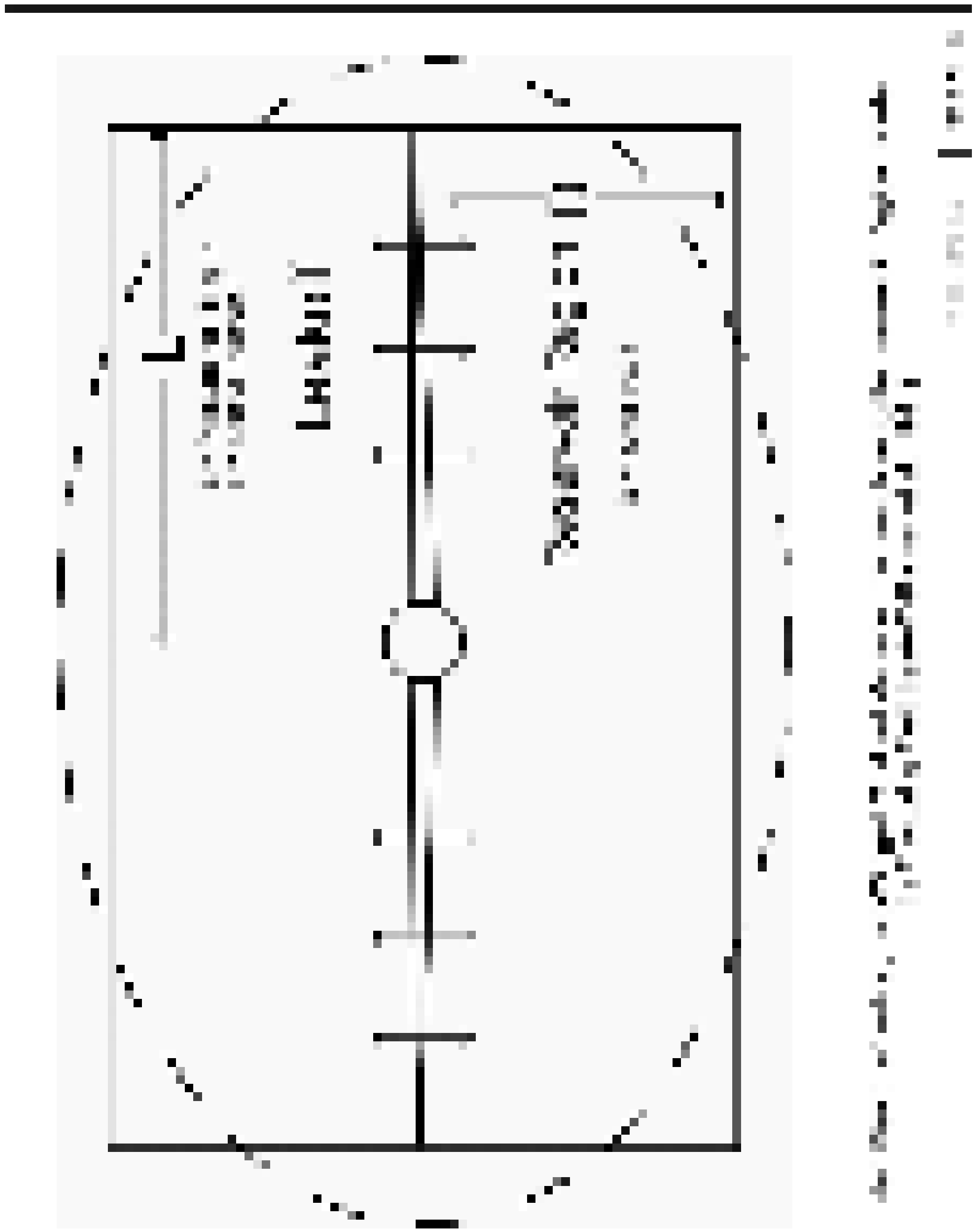

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Attachment 10
The Sand Wash Basin

The Sand Wash Basin is in northwestern Colorado and southwestern Wyoming. It is part of the Greater Green River Basin, which includes the Washakie Basin, the Great Divide (Red Desert) Basin, and the Green River Basin (Figure A10-1). These sub-basins are separated by uplifts caused by deformation of the basement rock. The Cherokee Arch, an anticlinal ridge that runs east to west along the Colorado/Wyoming border, separates the Sand Wash Basin from the adjacent Washakie Basin. The Greater Green River Basin, in total, covers an area of approximately 21,000 square miles. The Sand Wash Basin covers approximately 5,600 square miles, primarily in Moffat and Routt Counties of Colorado.

Coalbed methane resources in the Sand Wash Basin have been estimated at 101 trillion cubic feet (Tcf). Approximately 90 percent of this resource is within the Williams Fork Formation (Kaiser et al., 1993). Despite this ample resource, economic viability of recovery of the gas is limited by the presence of large volumes of water in most coalbeds. Presently, there appears to be no commercial production (GTI, 2002); however, approximately 120 permits for drilling within Moffat County were issued between February 2000 and August 2001 (Colorado Oil and Gas Commission, 2001). It is not clear exactly how many of these permits were related to coalbed methane exploration and production.

10.1 Basin Geology

The geologic history of the Sand Wash Basin is relatively complex, characterized by periods of deposition followed by deformation related to tectonic activity. This activity has impacted depositional patterns, coal occurrence and maturity, and hydrology (Tyler and Tremain, 1994). A very thorough discussion of the geologic history of the Sand Wash Basin is available in Tyler and Tremain (1994).

The coal-bearing formations in the region include the Iles, Williams Fork, Fort Union, and the Wasatch Formations (Figure A10-2). These formations were deposited, from bottom to top, during the Upper Cretaceous, Paleocene and upper Paleocene periods. The total thickness of the coal seams in these formations can measure up to 150 feet (Quarterly Review, 1993). Basement rock formations in the Sand Wash Basin can be as deep as 17,000 feet (Tyler and Tremain, 1994). A map of the coal and geologic features is presented in Figure A10-3a and a conceptual cross-section is presented in Figure A10-3b.

The Sand Wash Basin was near the western edge of the Western Interior Seaway that spreads across what is now central North America during the Upper Cretaceous (Figure

A10-4). During the late Cretaceous the seaway retreated to the northeast. Intermontane basins developed during the Laramide, and coal-bearing fluvial-lacustrine sediments were deposited (Quarterly Review, 1993). The coal in the Sand Wash Basin was formed from peat deposited in swamps along a broad coastal plain. Sediments that eroded from nearby uplift formations covered the peat beds (Tyler and Tremain, 1994). The alternating deposition of organic material and sands was repeated many times creating layers of coal interbedded with layers of sandstone and other sedimentary rocks that filled the basin.

Cretaceous or Mesaverde Group coal in the Sand Wash Basin ranges in rank from sub-bituminous along the basin margins to high volatile A bituminous coal in the deeper parts of the basin. These ranks are indicative of moderately mature to well-developed mature coal formed under high pressure and high heat. Within the Mesaverde Group, the most important potential coalbed methane resource in the basin (Kaiser et al., 1993), the coal ranks from sub-bituminous along the basin margins to medium volatile bituminous in the basin center (Kaiser et al., 1993). The methane in these coals formed both biogenically (by bacterial action on organic matter), and thermogenically (under high temperature). The average gas content of 261 coal samples collected during two studies was 147 standard cubic feet of methane per ton of coal (Boreck et al., 1977; Tremain and Toomey, 1983). Some samples from the Sand Wash Basin have been found to contain as much as 540 standard cubic feet of methane feet per ton. Gas content has generally been found to increase somewhat with depth. At depths of less than 1,000 feet, gas content is typically less than 20 standard cubic feet per ton, which has been taken to indicate that gas probably leaked out of the shallow coalbeds into the atmosphere. Analysis of gas samples has indicated that the gas is typically 90 percent methane, the remainder being mostly nitrogen and carbon dioxide (Scott, 1994). Carbon dioxide content ranges from 1 to more than 25 percent (Scott, 1994).

Of all the coal-bearing formations, the Upper Cretaceous Williams Fork is the most significant unit because it contains the thickest and most extensive coalbeds. The Williams Fork Formation is within the Mesaverde Group that also includes the Almond Formation along the Wyoming state line (Tyler and Tremain, 1994). The Almond Formation is shown (Figure A10-2) as a separate formation overlying the Williams Fork (Tyler and Tremain, 1994), but is also reported (Kaiser et al., 1993) to be a lateral equivalent of the upper Williams Fork Formation found in the southern Sand Wash Basin. For more information relative to this apparent conflict see Kaiser et al. (1993, p. 29). The coal-bearing Williams Fork Formation outcrops along the southern and eastern margins of the basin, and may be deeper than 8,000 feet in the deepest part of the basin (Figure A10-3b). The coals are interbedded with sandstones and shale. The thickest total coal deposits in the Williams Fork Formation, up to 129 feet, are centered near Craig, CO. This total is made up of several separate coalbeds up to 25 feet thick interbedded with sedimentary rock.

Stratigraphically above the Williams Fork Formation, the Paleocene Fort Union Formation, which includes sandstone, siltstone, shale, and coal, is also a potentially productive zone for coalbed methane production. The Fort Union outcrops at the Elkhead Mountains east of the basin and along the southern and western parts of the basin. The bottom of the Fort Union Formation is about 7,000 feet below the surface. Net coal thickness can be up to 80 feet with as many as nine individual beds. Individual beds up to 50 feet thick have been identified.

The Wasatch Formation includes beds of shale and sandstone and minor amounts of coal. It can extend as deep as 2,000 feet below the surface. The Wasatch Formation has not been targeted for coalbed methane development because of the small quantity of coal.

10.2 Basin Hydrology and USDW Identification

Regional groundwater flow in the Sand Wash Basin is from east to west and to the northwest towards the center of the basin. Water enters the aquifers at the exposed outcrops along the southern and eastern margins of the basin and moves northwestward. Vertical movement of groundwater, including potential artesian conditions, is dependent on local geologic conditions. Kaiser and Scott (1994) summarized their extensive investigation of groundwater movement within the Fort Union and Mesaverde Group. The Mesaverde Group is a highly transmissive aquifer. The coalbeds along with associated sandstone beds within the group may be the most permeable part of the aquifer. The Williams Fork Formation contains sandstone beds that are reported to be excellent aquifers (Brownfield, 2002). Lateral flow within the Fort Union Formation is slower, in part, owing to less permeable fluvial sandstones in the unit.

Total dissolved solids (TDS) concentrations of groundwater in the Mesaverde Group were investigated by Kaiser and Scott (1994) (Figure A10-5). They found that chloride concentrations ranged from 290 milligrams per liter (mg/L) in the eastern area of the basin near the outcrops where water enters the aquifers, to more than 26,000 mg/L in the central part of the basin. Calcium showed a similar pattern of distribution with the lowest concentrations near the outcrops, increasing toward the basin center. Calcium concentrations ranged from 10 mg/L to over 2900 mg/L. Based on the chloride and calcium concentrations presented by Kaiser and Scott (1994), the water in the aquifers near the recharge areas at the basin margins meets the water quality criteria for an underground source of drinking water (USDW) of less than 10,000 mg/L, but the water in the deeper central part of the basin does not (Figure A10-5). The mapped outcrop area (Figure A10-3a) of the Mesaverde Group indicates that the coal seam lies within a USDW where it is relatively shallow and close to the eastern and southern margins of the basin.

10.3 Coalbed Methane Production Activity

Coalbed methane resources in the Sand Wash Basin have been estimated at 101 Tcf. Approximately 90 percent of this gas is in the Williams Fork Formation (Kaiser et al., 1993). Approximately 24 Tcf of coalbed methane are located at depths less than 6,000 feet below ground surface (Kaiser et al., 1994). Despite this ample resource, economic viability of recovery of the gas is limited by the presence of large volumes of water in most coalbeds. Exploration in the 1980s and 1990s led to limited commercial use of the resource. Records from the Colorado Oil and Gas Commission indicate that approximately 31 million cubic feet of coalbed methane was produced in Moffat County during 1995 (Colorado Oil and Gas Commission, 2001). From 1996 to 1999 (the last year that data are available), no further gas was produced in this County (Colorado Oil and Gas Commission, 2001). However, Colorado Oil and Gas Commission records indicate that approximately 120 permits for drilling within Moffat County were issued during the period from February 2000 through August 2001 (Colorado Oil and Gas Commission, 2001). It is not clear exactly how many of these permits were related to coalbed methane exploration and production, but a handful of the permits were issued to gas companies, and the permits are listed as targeting known coalbeds within specific methane producing formations (Colorado Oil and Gas Commission, 2001).

At Craig Dome in Moffat County, Cockrell Oil Corporation drilled a 16-well development for exploration in the Williams Fork Formation. According to the Colorado Geological Survey, Craig Dome is located along the Cedar Mountain fault system (Colorado Geological Survey, 2002). The wells were abandoned a short time later because of excessive water. The Colorado Geological Survey indicated that the fault system may act as a conduit for anomalously high water migration from the outcrop. An average total of 40 feet of high-volatile bituminous coal was encountered in beds up to 15 feet thick. Gas content was tested at 10 to 350 cubic feet per ton of coal. Wells were cased through the target coalbed, perforated, and hydraulically fractured using water and sand. The wells yielded large volumes of fresh water with TDS levels measuring less than 1,000 mg/L, but little gas (Colorado Oil and Gas Commission, 2001). Water was removed at an average of 21,756 gallons per day per well during testing. Based on records from the Colorado Oil and Gas Commission, Cockrell Oil Corp does not appear to be involved currently with coalbed methane production in this region (Colorado Oil and Gas Commission, 2001).

The Colorado Geological Survey also indicated that faults in Trout Creek Canyon southeast of Craig are on trend with (and thus are likely to be related to) the Cedar Mountain fault system (Colorado Geological Survey, 2002). In addition, KLT Gas Inc. has a pilot program southwest of Craig Dome on the Breeze lease which is on trend with the Cedar Mountain fault system. If a fracture propagates into and along a fault plane, it may contaminate a USDW (Colorado Geological Survey, 2002.)

Limited commercial success has been experienced in the basin. As of 1993, only one commercial operator, Fuelco, was working in the basin. Fuelco was operating 11 wells along Cherokee Arch at 40 to 80 acre spacing. Well depths were to 2,500 feet. A total of 40 feet of coal was found in the Almond Formation (Mesaverde Group) between 810 to 2,360 feet. All wells were cased through the coal, selectively perforated, and stimulated using water and sand. Gas production averaged a total of 50,000 cubic feet per day from four wells. The highest producing well peaked at 100,000 cubic feet per day (Quarterly Review, 1993). Total production of gas through 1993 from the Dixon Field, the only producing field in this region, was about 84 million cubic feet (Kaiser et al., 1993). Total water production for the four wells was high at 126,000 gallons per day due to the high permeability of the coal (Quarterly Review, 1993). Water pumped from the wells contained 1,800 mg/L of TDS and was discharged to the ground with a National Pollution Discharge Elimination System permit (Quarterly Review, 1993).

The Sand Wash Basin has been used by the University of Texas Bureau of Economic Geology in the development of its Coalbed Methane Producibility Model (Kaiser et al., 1994). The development of the model was based on a comparison of basins that included the Sand Wash Basin and the San Juan Basin of southwestern Colorado and northwestern New Mexico. The San Juan Basin has proven to be a very productive coalbed methane resource. The Sand Wash Basin was used as an example of a basin with low potential for productivity (Figure A10-6) (Kaiser et al., 1994).

Hydraulic fracturing has been used in the Sand Wash Basin to improve the flow of gas into the wells. Hydraulic fracturing fluids have typically consisted of water with sand used as a proppant. However, very little information was available regarding specific types and volumes of fluids and proppants used. No indication of the use of other materials was noted in the sources reviewed (Colorado Oil and Gas Commission, 2001).

10.4 Summary

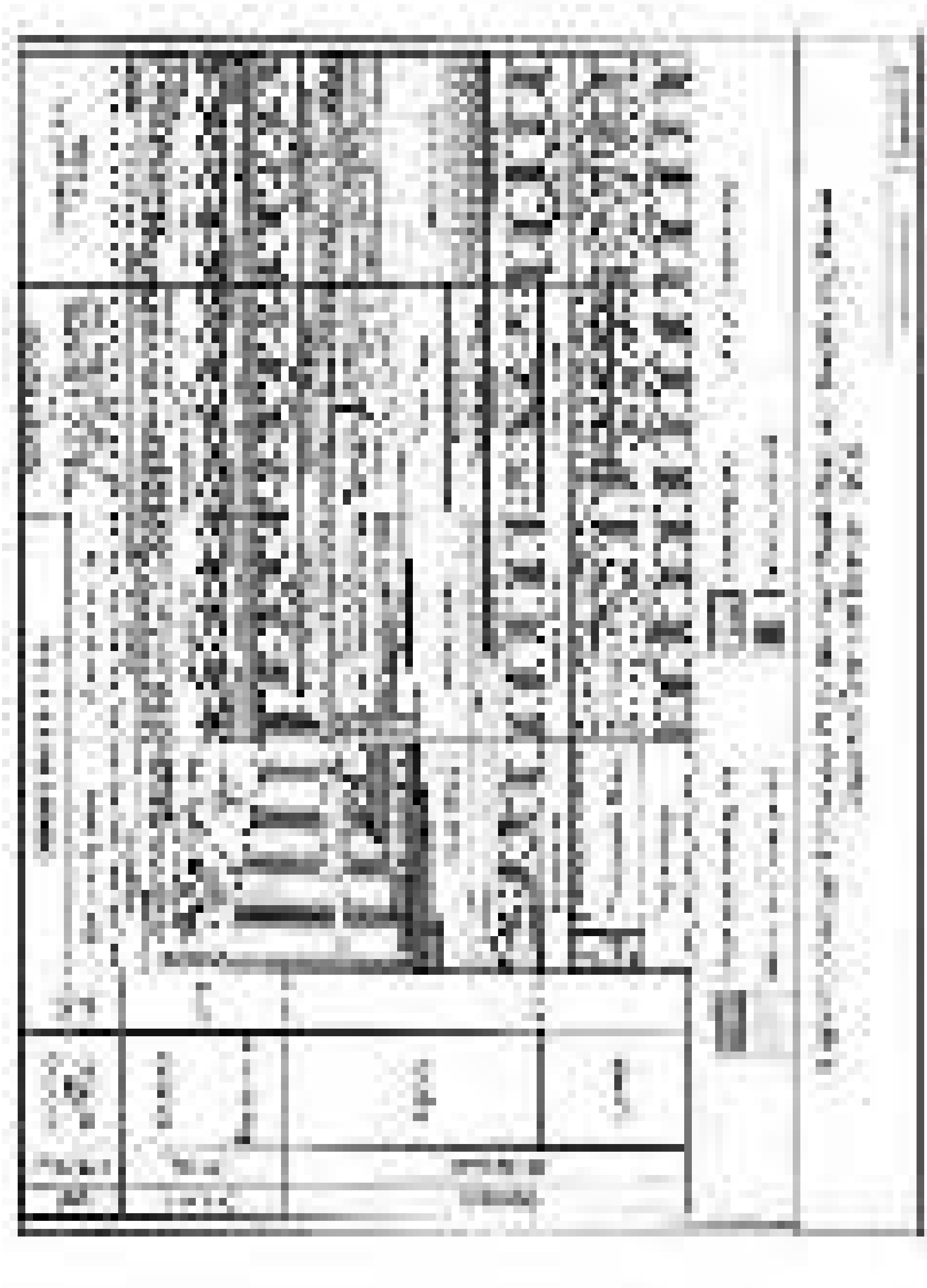
Coalbeds containing methane gas are present within the Sand Wash Basin at accessible depths. Some investigation and very limited commercial development of this resource have occurred, mostly in the late 1980s and early 1990s. There appears to be no commercial production at present. Development of coalbed methane resources in the Sand Wash Basin has been slower than in many other areas due to limited economic viability. The need for extensive dewatering in most wells has been a limiting factor, compounded by relatively low gas recovery.

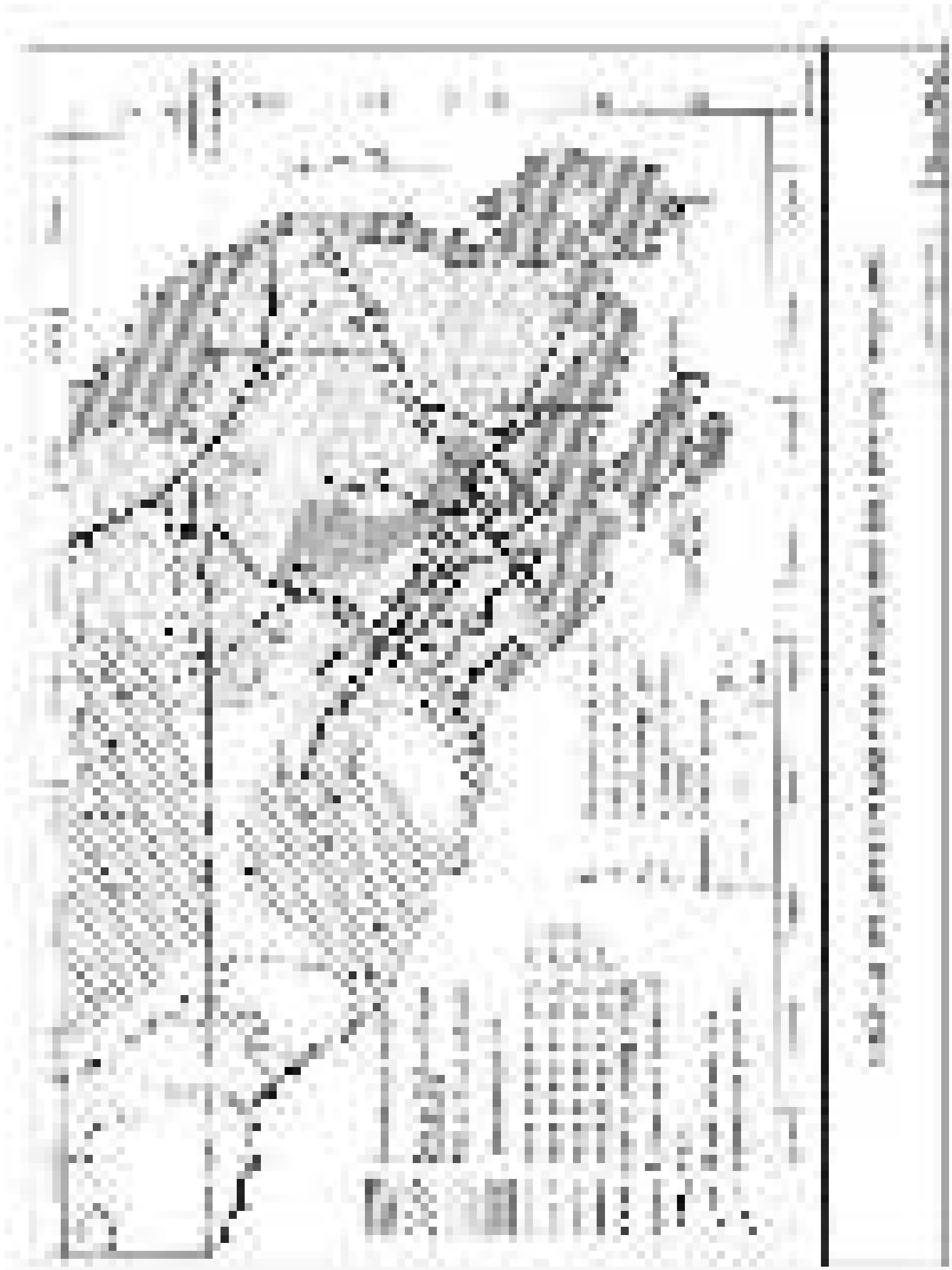
Between 1996 and 1999, no coalbed methane was produced in Moffat County. Permits for new gas wells have been issued indicating that there may be some continued interest in this area (Colorado GIS, 2001).

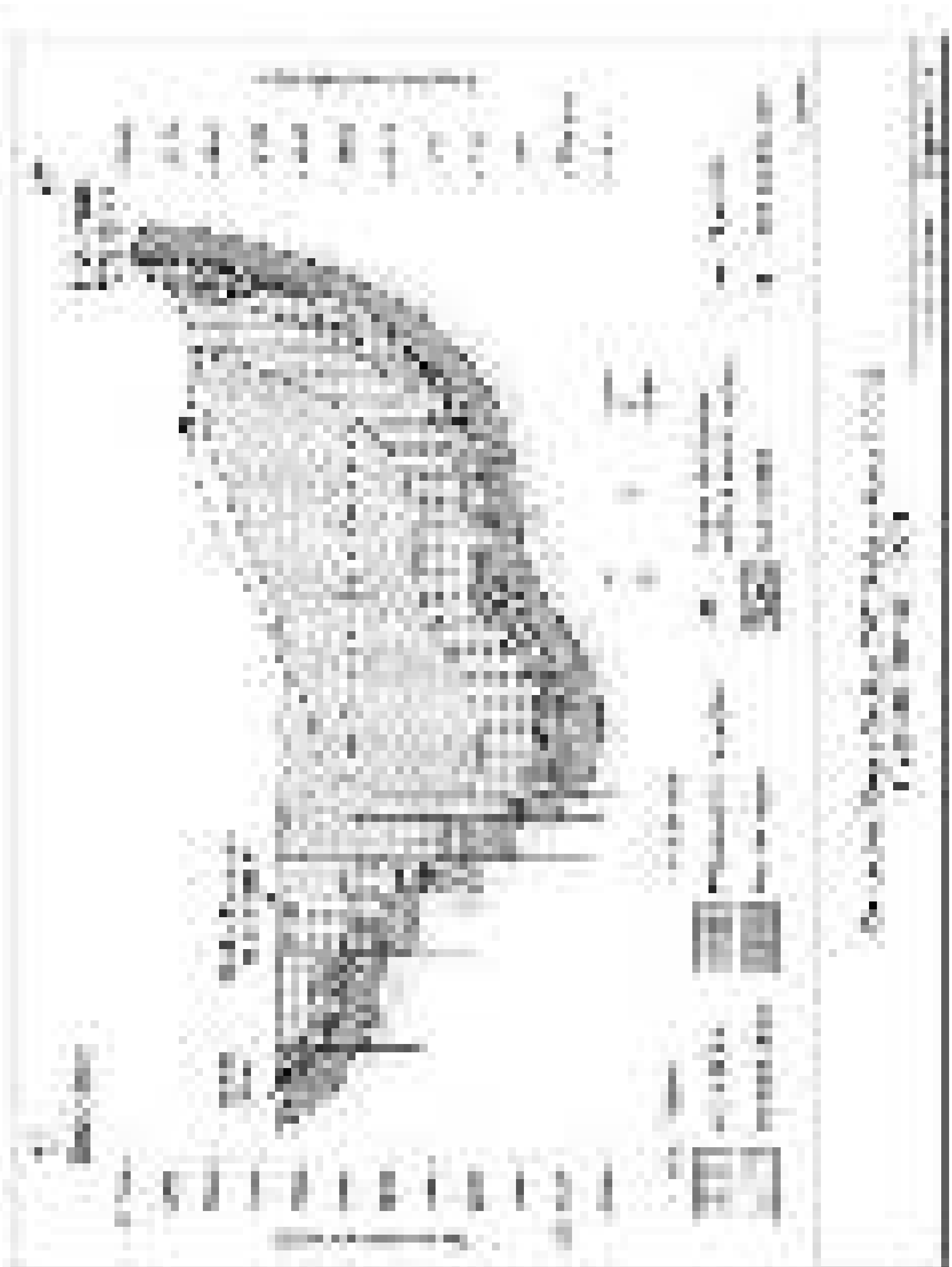
Groundwater quality in the basin varies greatly. Typically, chloride and calcium concentrations within the coal-bearing Mesaverde Group are low and potentially within potable ranges in the eastern and southern parts of the basin, implying the existence of a USDW, and therefore the potential for impacts. Concentrations increase as the water migrates toward the central and western margins of the basin. TDS concentrations significantly higher than the 10,000 mg/L USDW water quality standard have been detected in the western portion of the basin.

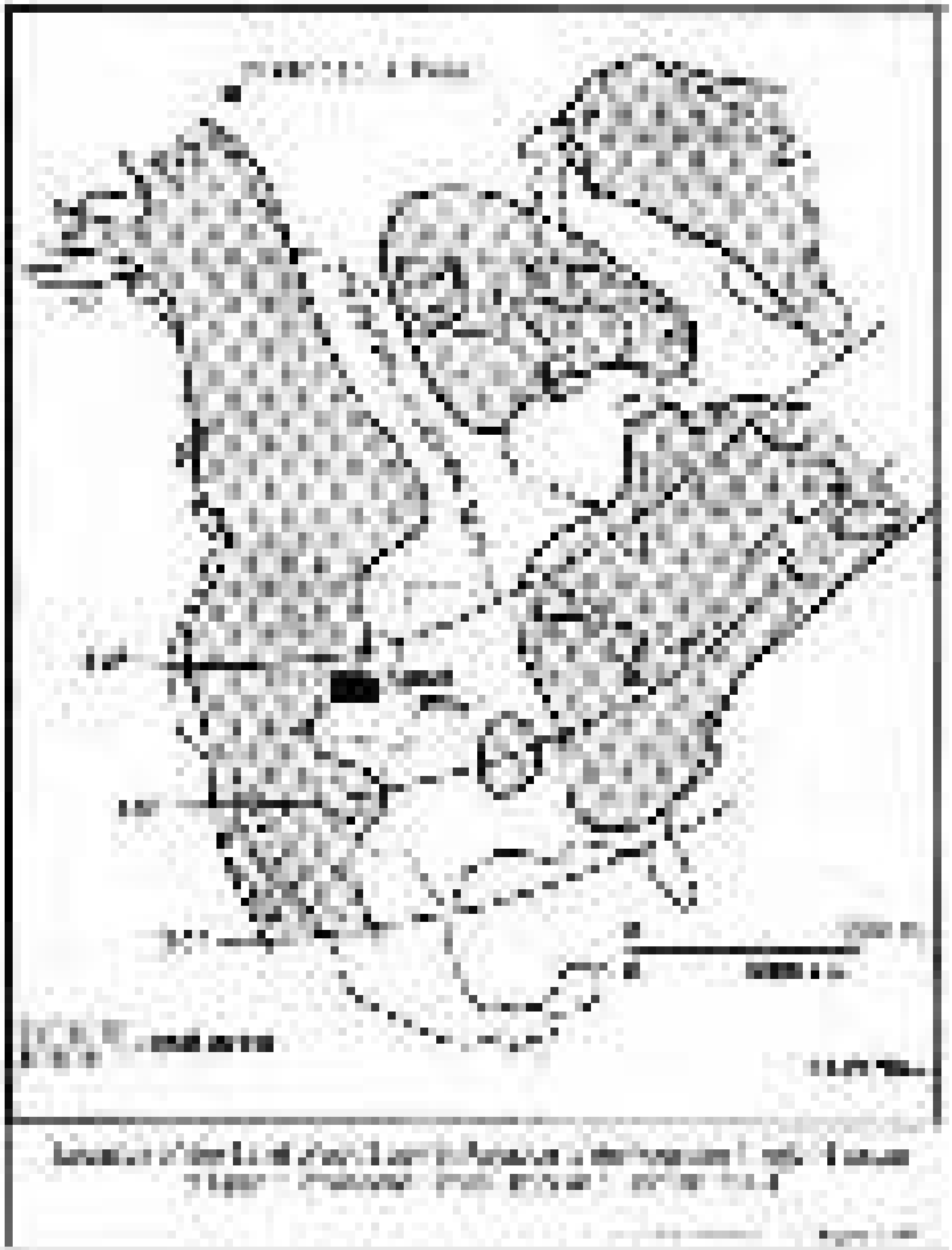
Compared to other potentially productive areas of the country, very little information has been published regarding current developments, groundwater location and conditions, drilling techniques, etc. The level of information available seems to be commensurate with the amount of commercial activity.

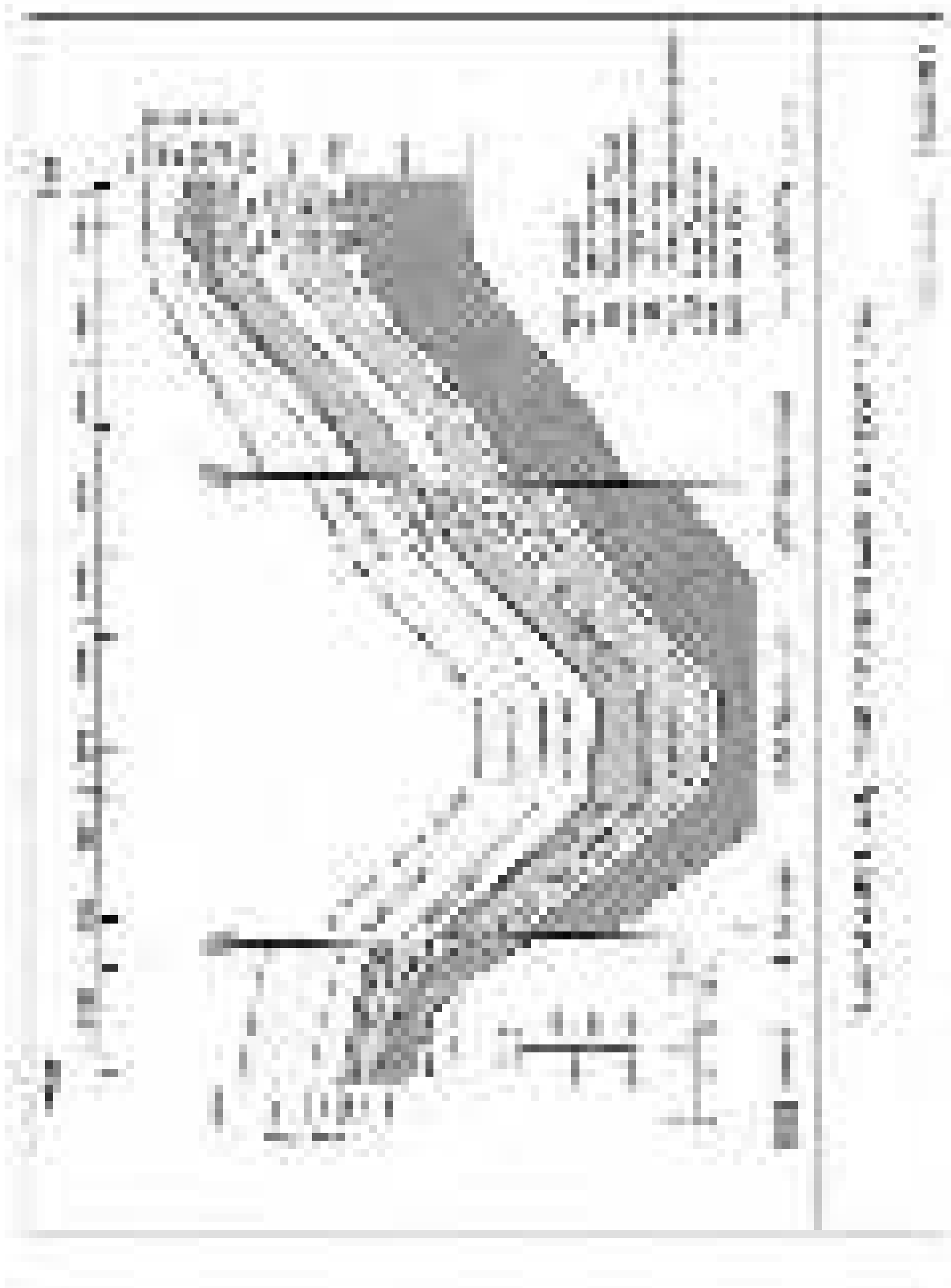
The use of fracturing fluids, specifically water and sand proppant, has been reported for this basin. No record of any other fluid types has been noted. Although variable, the water quality within the fractured coals indicates the presence of USDWs within the coalbeds.

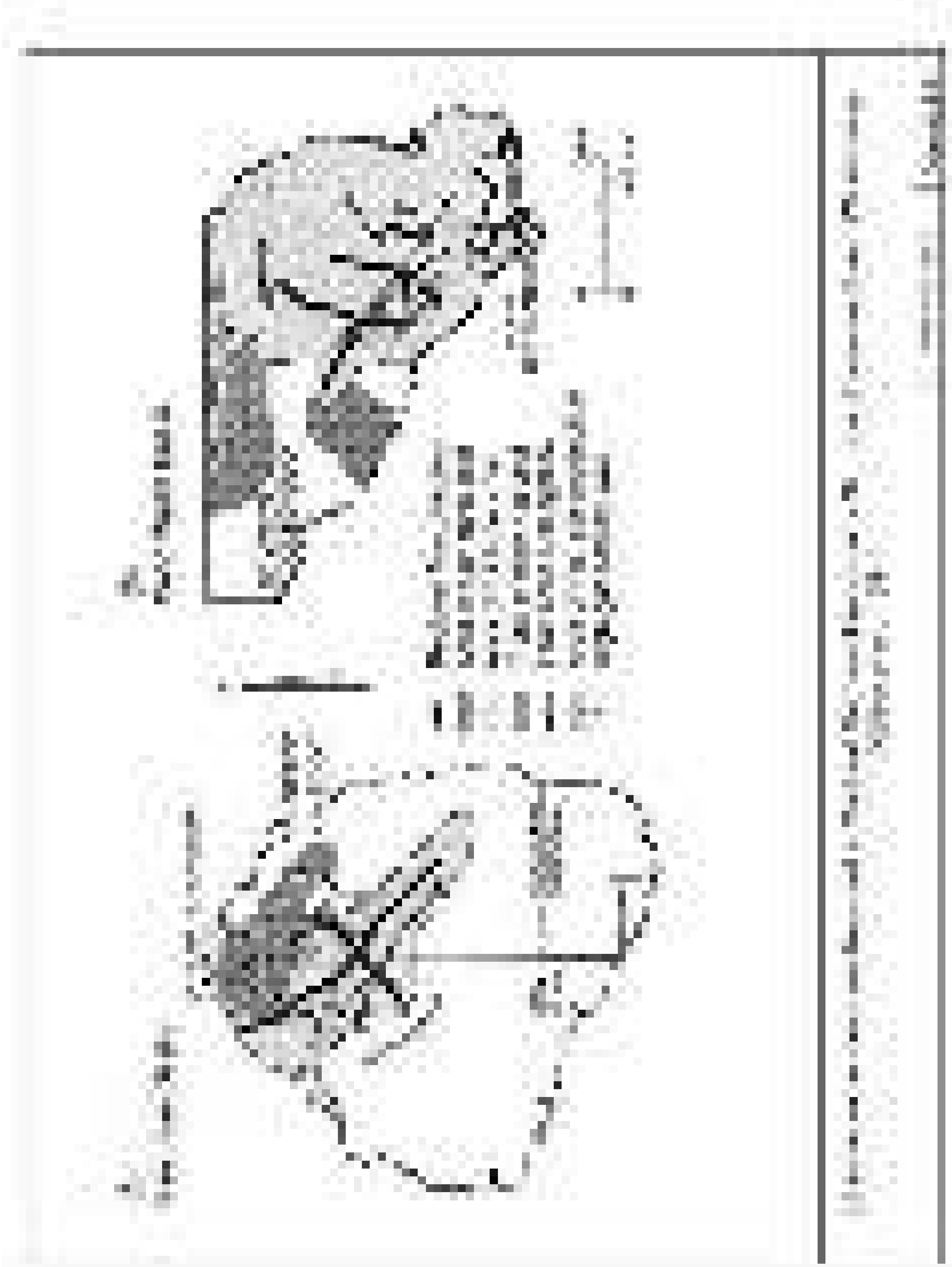












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Attachment 11
The Washington Coal Region (Pacific and Central)

The Pacific Coal Region (Figure A11-1) is approximately 6,500 square miles and lies along the western and eastern flanks of the Cascade Range from Canada into northern Oregon. The coals along the western flank lie within the Puget downwarp. Bellingham, Seattle, Tacoma, and Olympia in the State of Washington, and Portland, Oregon lie in or adjacent to the sub-basins. Choate et al. (1980) estimated coalbed methane resources for four target sub-basins (Figure A11- 1) representing 1,800 square miles of the 6,500 square mile Pacific Coal Region to be 0.3 trillion to 24 trillion cubic feet (Tcf). The Central Coal Region (Figure A11-2) is primarily the Columbia Plateau, between the Cascade Range to the west and the Rocky Mountains in Idaho, to the east. The Region extends from the Okanogan Highlands in the north to the Blue Mountains to the south, and encompasses approximately 63,320 square miles. Pappajohn and Mitchell (1991) estimated the coalbed methane potential of the Central Coal Region to be more than 18 billion cubic feet (Bcf) per square mile. According to the available literature, there were no producing fields in either the Pacific Coal Region or the Central Coal Region in Washington as of 2000 (GTI, 2001).

11.1 Basin Geology

A series of discontinuous coal fields lie along the western flank of the Cascade Range (Figure A11-3). The Roslyn and Taneum-Manastash fields are located on the eastern flank of the Cascade Range (Figure A11-3). The coal-bearing sediments were formed in a swampy fluvial-deltaic coastal plain depositional environment in the Paleocene to late Eocene Eras. In the Columbia Plateau Region, the Cretaceous to Eocene coal-bearing rocks are buried beneath a thick sequence of extrusive basalts.

The coal-bearing deposits of the Pacific and the Central Coal Regions are Cretaceous to Eocene Age and formed within fluvial and deltaic depositional environments prior to the uplift of the Cascade Mountain Range. The coalbeds of the Pacific and Central Basins are thought to result from peat accumulations in poorly drained swamps of the lower deltas while the thinner coalbeds probably formed in the better drained upper deltas (Buckovic, 1979 as cited in Choate et al., 1980). During the Oligocene, Cascade volcanic activity buried the deltaic sediments and compression caused some deformation of the sediments. During the Miocene, extensive volumes of basalt poured out in central Washington and covered the coal-bearing fluvial deposits. During the late Pliocene, the Coast Range and the Cascades continued to be uplifted, separating the Pacific Coal Region from the Central Coal Region, and causing extensive tectonic deformation, folding and faulting, of the coal-bearing sediments.

Deformation of the coal-bearing rocks increases toward the Cascade front. Fracturing may enhance porosity and permeability of the coalbeds, allowing greater methane storage and production (Pappajohn and Mitchell, 1991). On the other hand, however, fracturing may also increase the porosity and permeability of confining beds, allowing methane to escape up the stratigraphic section over time and dissipate in the atmosphere. Continuing deformation, primarily faulting, may be a limiting factor controlling methane production in the Pacific Coal Region as well.

11.1.1 Pacific Coal Region Geology

In the Pacific Coal Region, deformation has increased geologic complexity making it difficult to follow or correlate coalbeds, especially across faults. Geothermal heating along the western flank of the Cascades created a thermally altered zone of increased coal rank ranging into the bituminous and anthracite ranks. The maturation to bituminous rank increases potential methane yields (Walsh and Lingley, 1991; Pappajohn and Mitchell, 1991).

The major coal-bearing areas are in, from north to south, Whatcom, Skagit, King, Pierce, Kittitas, Thurston, Lewis, and Cowlitz Counties in Washington (Figure A11-3). The discussion of regional geology presented here illustrates the geologic conditions in the Green River district in King County, the Wilkerson-Carbonado coalfield in Pierce County, and the Centralia-Chehalis district in northern Lewis and southern Thurston Counties, and does not attempt to provide a detailed description of every coalfield. For more detailed information on the Bellingham area, Whatcom County, the reader is referred to Beikman et al. (1961), for Whatcom and Skagit Counties to Jenkins (1923 and 1924), and for the Roslyn coal area to Walker (1980). Other areas not discussed but important within the Pacific Region are the Toledo-Castle Rock District, and the Roslyn-Cle Elum and Teneum-Manastash fields. The stratigraphy for three sub-basins (Green River, Wilkerson-Carbonado, and Centralia-Chehalis) of the Pacific Coal Region is presented in Figure A11-4. The general setting and geology of each sub-basin is unique and complex.

The coal deposits of King County are located southeast of Seattle (Figure A11-3). The Green River district is the largest and most extensively mined coal-bearing area in King County. The King County coals occur in the Puget Group of Eocene Age (Figure A11-4). Evans (1912) divided the Puget Group into 3 coal zones, which, from oldest to youngest, are the Bayne, Franklin, and Kummer. Deformation has been moderate and most of the coalbeds dip less than 35 degrees. In parts of the Green River district the deformation has been more intense, and dips of 50 degrees or more are common. The King County coals range in rank from subbituminous to high-volatile bituminous. Within the Green River District, the Puget Group is estimated to be at least 6,500 feet thick and contains at least 15 coalbeds up to 40 feet thick (Beikman et al., 1961). The principal coalbeds are located in the Franklin and Kummer zones in the Puget Group (Vine, 1969). Coal has been mined in the Green River District since about 1883, and it has produced more than 25 million tons of coal. Currently there is no coal production in the district.

The Wilkerson-Carbonado coalfield is located in Pierce County, southeast of Seattle (Figure A11- 3). The Pierce County coals occur in the Eocene Carbonado Formation (Beikman et al., 1961). The Carbonado consists of more than 5,000 feet of interbedded, layered lenses of sandstone, siltstone, mudstone, and shale with carbonaceous shale and coal (Figure A11-4). At least 10 coalbeds have been identified in the area. Coalbeds range in thickness from 1 to 5 feet with the maximum thickness of 15 feet. The coals range in rank from high-volatile bituminous to low-volatile bituminous. The Wilkerson-Carbonado coals have the highest rank of any major coal-bearing area in Washington State. Throughout the field, deformation has been intense. Dips of 60 degrees or more are common, and fault displacements range from a few feet to more than 1,500 feet. Although these areas have recently been targets of coalbed methane exploration, there is currently no production.

The coal deposits of Lewis and Thurston Counties occur in the Skookumchuck Formation (Figure A11-3) of late Eocene Age (Snavely et al., 1958). The Centralia-Chehalis district is located in northern Lewis and southern Thurston Counties (Figure A11-3). Deformation of the Skookumchuck is moderate resulting in tightly folded anticlines and broad open synclines. The coal deposits have been cut by a series of high angle reverse faults roughly paralleling the fold axes. The faults dip to the northeast, with the southwest block downthrown, and have displacements ranging from 200 to 500 feet. The coal rank ranges from lignite to anthracite. The central part of the Centralia-Chehalis district contains as many as 14 subbituminous coalbeds ranging from a few inches to over 40 feet in thickness. The district contains more than half of the calculated coal reserves of the State. The TransAlta Centralia Mining Company continues to operate a major strip mine centered about 5 miles northeast of Centralia, where it is anticipated that 9,400 acres will be stripped over 35 years. Within the Centralia mine, the Big Dirty bed is more than 40 feet thick. To the west of Centralia, the Vader coal area contains several lignite beds with thickness up to 20 feet, which may correlate in part with the coals in the Centralia-Chehalis area.

In Whatcom and Skagit counties (Figure A11-3), the Chuckanut Formation contains as many as 15 coalbeds, ranging from 1 to 15 feet thick and ranking from lignite to anthracite, but generally bituminous. The rank of the coal increases eastward towards the crest of the Cascades Range.

The rank of Pacific Region coals varies greatly from place to place, ranging from lignite to anthracite, but generally rank increases toward the crest of the Cascade Range. The coal rank is used to identify bituminous coal-target areas where gas yields may be greatest. While the structural geology is very complex, the thermally-altered metamorphic zone is rather predictable. Both of these factors will play a major role in the design of any exploration and development plans for coalbed methane in the Pacific Coal Region.

The complex stratigraphy and structural deformation of the lenticular coals in the Pacific Coal Region are major obstacles to the exploration and development of coalbed methane fields. Predicting the location of coalbeds is a complex and difficult process because the geology in the area has been modified by intense deformation. Additionally, the faulting that commonly occurs along the axes of anticlines may form conduits for the escape of methane through overlying confining beds. Steeply dipping beds of coal have presented difficulties in controlling drill bit directions and in development and stimulation for coalbed methane production.

Choate et al. (1980) estimated coalbed methane resources for four target sub-basins (Figure A11-1), representing 1,800 square miles of the 6,500 square mile Pacific Coal Region, to be 0.3 trillion to 24 Tcf. Methane had been encountered in 67 oil and gas exploration wells drilled in this region by 1984. Methane gas was found at depths of less than 500 feet in 25 wells, less than 1,000 feet in 38 wells, and less than 2,000 feet in 50 wells. In western Whatcom County, methane has been found in unconsolidated glacial drift capped by impervious clay beds. East of Ferndale, methane gas reportedly has been produced commercially from unconsolidated deposits at depths ranging from 166 to 193 feet at flow rates ranging from 750,000 to 5,000,000 cubic feet per day (Choate et al., 1980).

11.1.2 Central Coal Region Geology

The Central Coal Region refers to the coal-bearing formations east of the Cascade Range. The Columbia River Basalt Group, primarily the Grande Ronde Basalt, Wanapum Basalt, and Saddle Mountains Basalt bury the Cretaceous to Eocene coal-bearing formations of the Central Coal Region. In this region, methane is entrained in groundwater from confined aquifers in the basalts. Interbedded with the flood basalts are epiclastic and volcaniclastic sediments. The less fractured zones of basalt appear to act as aquitards (Johnson et al., 1993). Johnson et al. (1993) have concluded that the greatest volume of methane is derived from upward migration from the underlying Eocene coals. They also suggest that faults through the underlying sediments and basalts provide conduits for the migration of gas-bearing groundwater into the confined zones.

The Yakima fold belt lies between the confluence of the Snake and Columbia Rivers and the Cascade Range, and is a series of broad asymmetric anticlines and synclines whose axes generally trend west northwest to east southeast (Figure A11-5). The anticlinal ridges are typically cut by thrust faults that are inclined and steepen with depth (Reidel et al., 1989). While the anticlines may form structural traps for methane in the source coalbeds, the thrust faults in the anticlines may form conduits for the upward migration of methane through overlying confining beds. The fold structures are very flat and broad and do not result in the steeply dipping strata that are characteristic of the Pacific Coal Region west of the Cascades.

11.2 Basin Hydrology and USDW Identification

Surficial deposits of Pleistocene glacial outwash locally form aquifers capable of sustaining public drinking water supplies in the Pacific and Central Washington Regions. In the Central Coal Region, aquifers in the basalts are extensively developed for irrigation. Public water supplies in Pierce County (Olympia area) and King County (Seattle area) of the Puget Sound Region (Pacific Coal Basin) are obtained from the glacial drift aquifer (Dion, 1984) that overlies Eocene sediments, which may contain coal and methane. Water quality information from four gas test wells indicates the presence of 1,330 to 1,660 milligrams per liter (mg/L) total dissolved solids (TDS) in water within the coalbeds of Pierce County (Dion, 1984). This meets the water quality requirements of an underground source of drinking water (USDW). The Washington Department of Ecology and the EPA deemed this water to be of sufficient quality to permit its discharge to surface waters of the Carbon River (Pappajohn and Mitchell, 1991).

The Columbia River Basalt Group is identified as a major regional multi-aquifer province (Lindholm and Vaccaro, 1988; Dion, 1984). The aquifer is used extensively for irrigation, but may also be used as a source of drinking water. Wells in the Basalts commonly yield 150 to 3,000 gallons per minute. TDSs in the water produced generally range from 250 to 500 mg/L (Dion, 1984).

The occurrence of methane in groundwater is one factor leading to the assessment of the coalbed methane production potential in Washington. Methane in groundwater occurs in the basalts, but only in confined aquifers (porous or fractured zones near the top or bottom of a basalt layer), and is thought to have migrated upward from underlying coalbeds. Water supply wells and irrigation wells in the Columbia River Basalts and water wells in numerous different lithologies in the Pacific Coal Region have been recognized as containing methane. Data demonstrating the co-location of a coal seam and a USDW were found for Pierce County, where methane gas test well results report TDS levels far lower than the 10,000 mg/L USDW water quality threshold (Dion, 1984).

11.3 Coalbed Methane Production Activity

Complex stratigraphy and structural deformation creates major obstacles to the development of gas from the Pacific Coal Region. The coals are known from active and inactive mines to be gassy, folded, faulted, and commonly steeply inclined. The difficulties and dangers involved with underground coal mining led to closure of the mines once the shallow deposits were exhausted. However, their characteristics have been well documented by the mining operations. Many of these same structural characteristics have impeded the development of coalbed methane gas. The available literature indicates that no significant production had been achieved by 1996 (GRI, 1999). According to the available literature, there were no producing fields in either the Pacific Coal Region or the Central Coal Region in Washington as of 2000 (GTI, 2001). However, in northwest Oregon, the Mist gas field was developed in the 1990s.

11.3.1 Pacific Coal Region Production Activity

Between 1986 and 1993, 19 coalbed methane wells were drilled in the northern Pacific Coal Region (Quarterly Review, 1993). Three tests were conducted near the town of Black Diamond in the Green River coal area of King County. One of the wells was hydraulically fractured and the others completed by open-hole cavitation. Steep dips of the strata led to wellbore deviation during drilling and to caving following the fracturing operations. One well produced 32,000 to 62,000 cubic feet per day of coalbed methane gas with no water in an open-hole test. Another was hydraulically fractured with 12/20 mesh sand and nitrogen foam in two zones at depths of 2,228 to 2,442 feet and 2,505 to 2,638 feet, but no test results were released. Caving was so prominent that it interfered with wellbore cleanup following the hydraulic fracturing operations. According to available publications, optimal fracturing and completion methods for use in the structurally difficult Pacific Coal Region are yet to be applied and proven.

11.3.2 Central Coal Region Production Activity

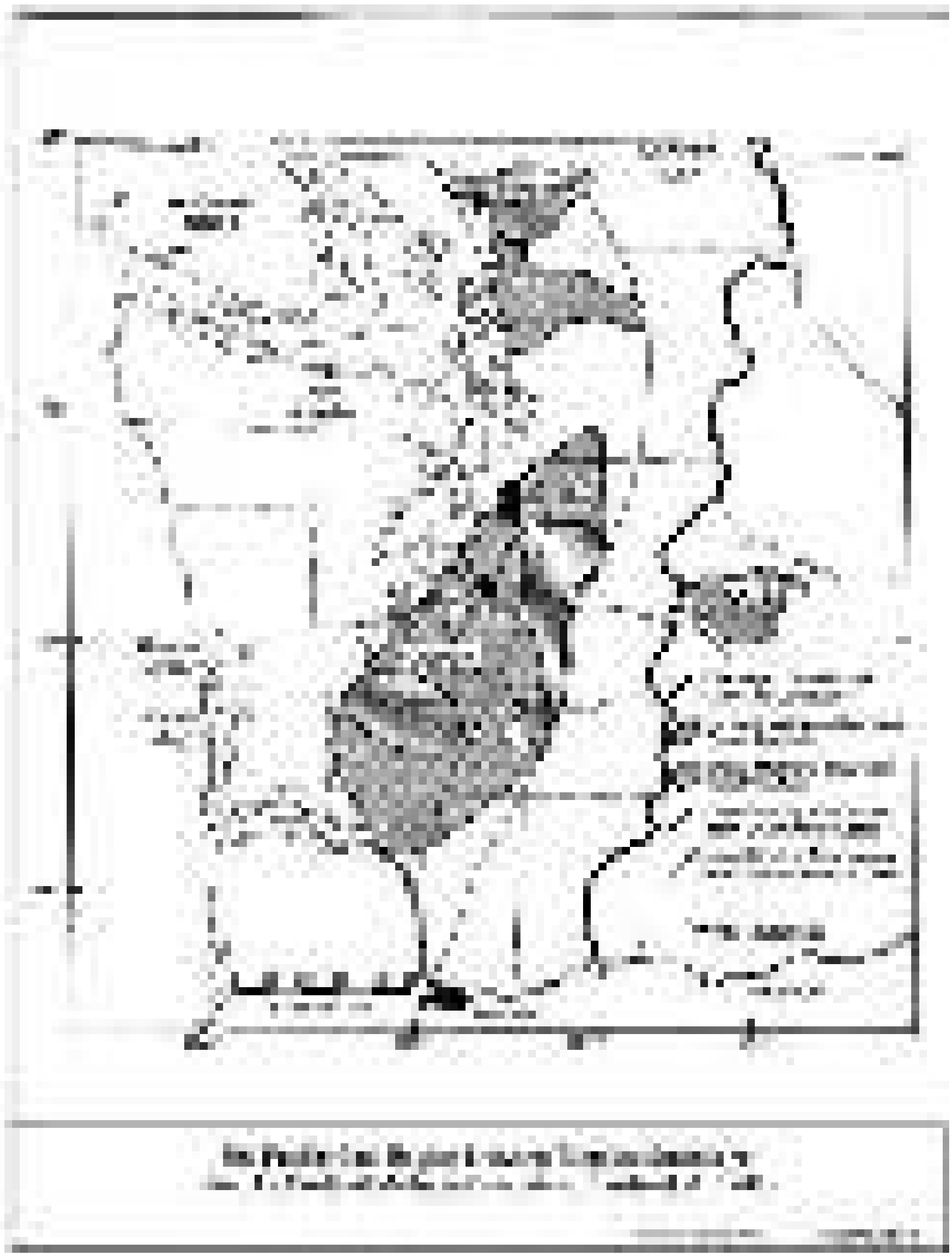
The one commercial gas field (Rattlesnake Hills) in the Central Coal Region was shut down in 1941. Production from the Cretaceous to Eocene coalbeds that lie below the basalts may have large potential. Pappajohn and Mitchell (1991) estimated the coalbed methane potential of the Central Coal Region to be more than 18 Bcf per square mile. It is unlikely that the whole 63,320 square miles of the region could yield that rate because the coals are only known to occur below the basalts in the western part of the basin. Much is not known about the potential coalbed methane production from these obscured deposits, and development depends on successful exploration.

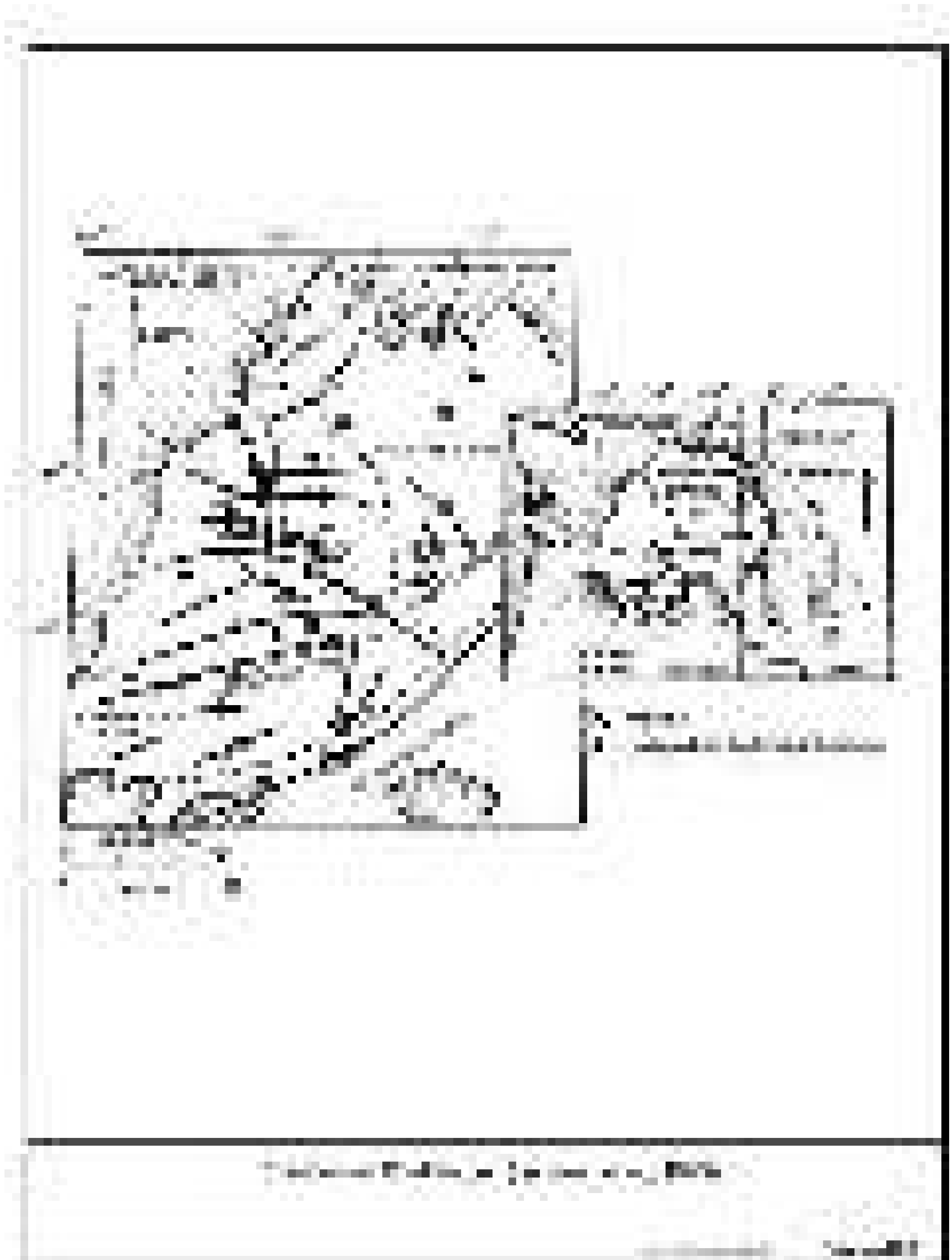
Although the coals of the Central Coal Region may not be as greatly deformed and unpredictable as those in the Pacific Coal Region, they are overlain by the Columbia River Basalt Group, in which individual basalt flows up to 300 feet thick can cover thousands of square miles. The Rattlesnake Hills gas field operated between 1913 and 1941 in the western part of this region and indicates greater potential for development.

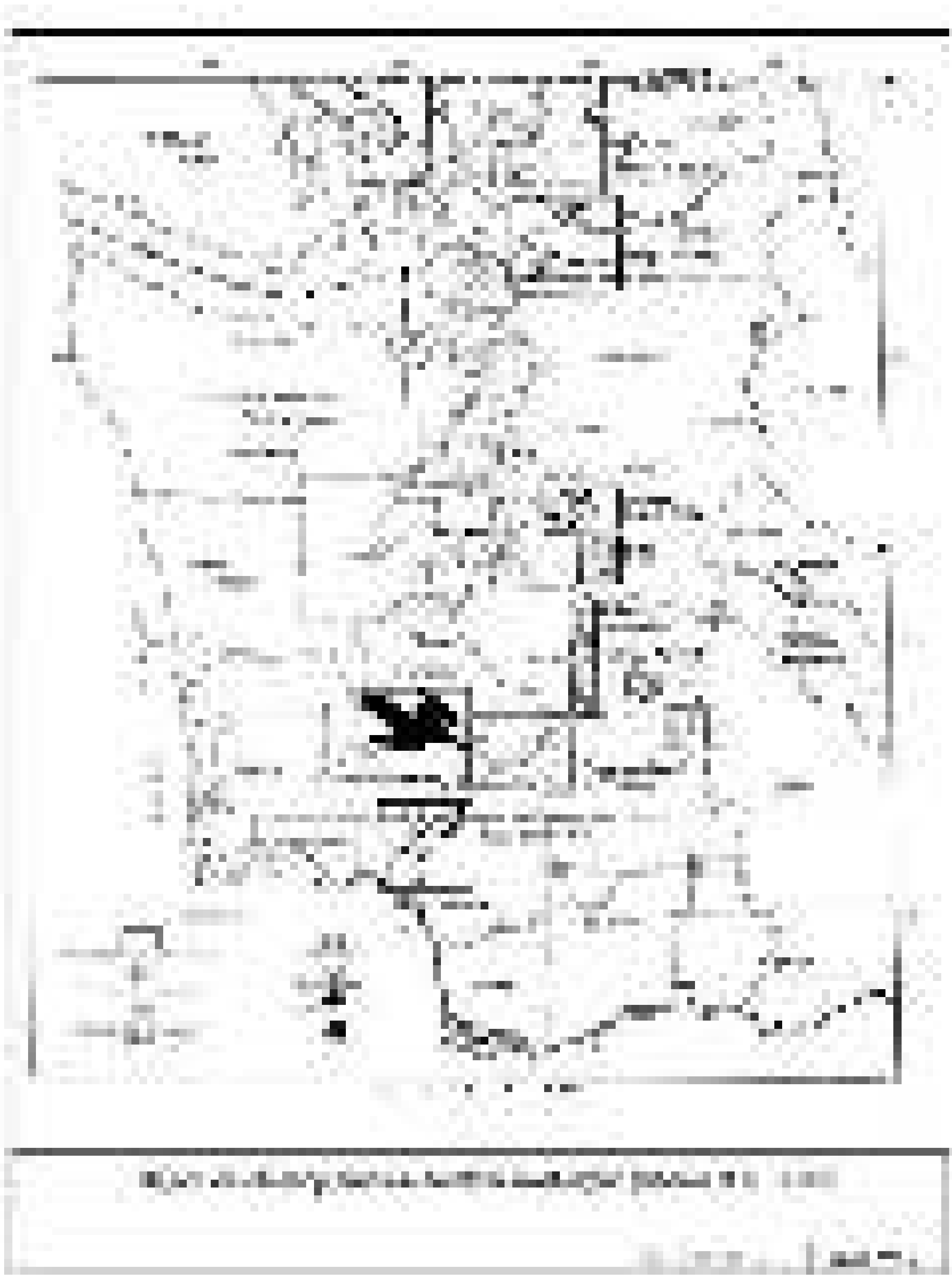
11.4 Summary

The geologic structure of the coal-bearing rocks is difficult to interpret in the Pacific and Central Coal Regions, and methane may be technically difficult to produce in these regions. A connection exists between the Washington coalbeds and a USDW. However, there were no producing coalbed methane wells in the Pacific and Central Coal Regions in Washington as of 2000 (GTI, 2001). In some areas, the Pacific and Central Regions' coals exist within a potential USDW. In other areas of the basin, there is evidence that the coalbeds are below a USDW. Hydraulic fracturing has been documented in this region. Data demonstrating the co-location of a coal seam and a USDW were found for Pierce County, where methane gas test well results report TDS levels of 1,330 to 1,660 mg/L, far less than the USDW classification limit of 10,000 mg/L (Dion, 1984).

In this region, methane occurs in groundwater flowing through fractured zones in basalts, although less fractured zone of the basalts appear to act as hydraulic confining layers. Johnson, et al. (1993) concluded that the greatest volume of this methane has migrated upward from underlying coalbeds. Water supply wells and irrigation wells in the Columbia River Basalts and water wells in numerous different lithologies in the Pacific Coal Region have been recognized as containing methane. Development of coalbed methane in the Washington Coal Region may have some impact on highly productive basalt aquifers that meet the requirements of a USDW and are already in use as large sources of irrigation water for agriculture.







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Attachment 2
The Black Warrior Basin

The Black Warrior Basin covers an area of about 23,000 square miles in Alabama and Mississippi. The basin is approximately 230 miles long from west to east and approximately 188 miles long from north to south. Coalbed methane production in Alabama is limited to the bituminous coalfields of west-central Alabama, primarily in Jefferson and Tuscaloosa Counties.

Coalbed methane production in the Black Warrior Basin is among the highest in the United States. In 1996, approximately 5,000 coalbed methane wells were permitted in Alabama. In 2000, this number increased to over 5,800 wells (Alabama Oil and Gas Board, 2002). Coalbed methane well production rates range from less than 20 to more than one million cubic feet per day per well (Alabama Oil and Gas Board, 2002). Between 1980 and 2000, coalbed methane wells in Alabama produced roughly 1.2 trillion cubic feet of gas. According to the Gas Technology Institute (GTI), annual gas production was 112 billion cubic feet in 2000 (GTI, 2002).

2.1 Basin Geology

Coalbed methane production in the Black Warrior Basin (Figure A2-1) is contained within the Upper Pottsville Formation of Pennsylvanian age (300 million years). The depositional history along the ancient coastline of prehistoric Alabama was characterized by 8 to 10 "coal deposition cycles" of sea level rising and lowering. Each of these 10 geologic "coal deposition cycles" features mudstone at the base of the cycle (deeper water) and coalbeds at the top of the cycle (emergence) (Pashin and Hinkle, 1997).

The geologic structure of the Black Warrior Basin is complex. Due to erosion and structural uplift, not all of the coal zones are present at all locations (Pashin et al., 1991; Young et al., 1993). In general, however, most coalbed methane wells tap the Black Creek/Mary Lee/Pratt cycles, at depths that range from 350 to 2,500 feet deep (Holditch, 1990).

Alabama coalbeds are typically very thin, ranging from less than 1 inch in thickness to 4 feet (in rare cases they may be up to 8 feet thick in surface mines) (Horsey, 1981; Heckel, 1986; Eble et al., 1991; Carrol et al., 1993; Pashin, 1994) (Figure A2-2). In the area of coalbed methane development, the Pottsville Formation exists at or near the surface, and the depth to commercial coalbeds ranges from the surface outcrop to 3,500 feet, depending on location (Figure A2-3).

2.2 Basin Hydrology and USDW Identification

In the location where coalbed methane development is taking place in west-central Alabama, the Pottsville Formation is an unconfined aquifer. The matrix permeability of Pottsville rocks (e.g., mudstone, cemented sandstone) is low, but water is present and flows within an extensive system of faults, fractures, and joints. Flow patterns within the Pottsville Formation are strongly controlled by fault- and fold-related isotropic joints and fractures (Koenig, 1989). The close spacing and systematic pattern of cleats, however, make coalbeds the most productive aquifers within the Pottsville Formation (Koenig, 1989; Pashin et al., 1991; Pashin and Hinkle, 1997). In the early 1990s, several authors reported fresh water production from coalbed wells at rates up to 30 gallons per minute (Ellard et al., 1992; Pashin et al., 1991).

Most of the recharge to the Pottsville aquifer is precipitation that infiltrates from the surface, but some recharge occurs where streamflow enters the outcrop and moves laterally into the aquifer along folded anticlinal beds (Pashin and Hinkle, 1997) (Figure A2-4). Several researchers also propose upwelling of more saline waters from deeper zones, which takes place along vertical, fault-related, rubble zones (Pashin et al., 1991). Discharge from the Pottsville aquifer is primarily from the dewatering of coalbeds due to mining and coalbed methane production (Pashin et al., 1991).

Formation water produced from Alabama coalbed methane wells contains between less than 50 to over 10,000 milligrams per liter (mg/L) total dissolved solids (TDS) (Koenig, 1989; Pashin et al., 1991; Pashin and Hinkle, 1997). Some portions of the Pottsville Formation contain waters which meet the quality criterion of less than 10,000 mg/L TDS for an underground source of drinking water (USDW) (Figure A2-7). According to the Alabama Oil and Gas Board, some waters in the Pottsville Formation do not meet the definition of a USDW and have TDS levels which are considerably higher than 10,000 mg/L (Alabama Oil and Gas Board, 2002). Water quality generally decreases with increasing depth (Figures A2-7 and A2-8), and areally is related to the faulting pattern (Figure A2-9) (Pashin et al., 1991; Pashin and Hinkle, 1997). Waters exceeding 10,000 mg/L TDS can be found below 3,000 feet in areas near deep vertical faults, suggesting upwelling from deeper, more saline zones (Pashin and Hinkle, 1997).

2.3 Coalbed Methane Production Activities

Alabama coalbed methane wells are categorized into three distinct types. The first two types, "gob" wells and horizontal wells, are less common. Gob wells are associated with mines. The well is drilled to a depth above the mine roof, and when the mine is abandoned, the roof collapses. Gob wells produce coalbed methane from the fractured mine debris. A few horizontal wells are drilled from within mines to reduce coalbed methane concentration in advance of a working face. The third type, which includes 98 percent of all Alabama methane wells, includes vertically drilled wells that utilize mainstream oilfield technologies (Pashin and Hinkle, 1997). Because neither gob nor horizontal wells typically are hydraulically fractured, this discussion is limited to vertical wells.

According to literature, most coalbed methane wells are drilled using water or air rotary methods or water-based mud, due to lower cost and concerns that mud fluids will invade the coal. Wells in Alabama are completed with tubing. Water is pumped up the tubing for disposal, whereas gas is produced up the annulus. Wells are drilled to a depth 10 to 30 feet below the lowest coalbed to create a sump that collects coal fines and allows water to separate from the coalbed methane (Holditch, 1990).

About 95 percent of produced water is disposed by discharge into surface water, via Type II National Pollution Discharge Elimination System permits (O'Neil et al., 1989; O'Neil et al., 1993; Pashin and Hinkle, 1997). These permits require some water quality monitoring and limit instream water quality to 230 mg/L TDS (Pashin and Hinkle, 1997). Since 1991, about 5 percent of produced water has been injected for disposal into Class II injection wells (Pashin and Hinkle, 1997). Eight Class II wells are currently active (Alabama Oil and Gas Board, 2001), disposing coalbed waters into zones between 4,300 and 10,000 feet deep (Ortiz et al., 1993).

Most wells are completed in multiple coal zones using perforations. Some wells are completed in mudstones immediately below a coal zone, rather than within the coal ("limited entry" completions), and a few wells feature un-cased, open-hole completions. Each well is hydraulically fractured to allow communication with the thin coal seams outside of the casing, and most wells are fractured more than once as described below:

- In wells with multiple coal seams present, the hydraulic fracturing process may involve several or multiple stimulations, using 2 to 5 hydraulic fracture treatments per well (depending on the number of seams and spacing between seams); and,

- Many coalbed methane wells are re-fractured at some time after the initial treatment, in an effort to re-connect the wellbore to the production zones to overcome plugging or other well problems (remedial fracture-stimulation) (Holditch, 1990; Saulsberry et al., 1990; Palmer et al., 1991a and 1991b; Schraufnagel et al., 1991; Holditch, 1993; Palmer et al., 1993b; Spafford et al., 1993; Schraufnagel et al., 1993) (Figure A2-10).

The geometry of hydraulic fractures in coalbed methane zones usually differs from that observed in conventional oil and gas scenarios. In conventional hydrocarbon zones, the gas and/or oil are physically "trapped" by the presence of an impermeable confining layer, usually shale. Shale formations may present a barrier to upward fracture growth because of the stress contrast between the coalbed and the higher-stress shale (see Appendix A). Therefore, for conventional fracturing, the vertical growth of fractures out of the target zone may be limited by the presence (i.e., stress contrast) of overlying shales. In conventional gas-well fracture environments, fracture half-length (200-1,600 feet from the well bore) almost always exceeds fracture height (10-200 feet above the perforations). In the Black Warrior Basin, however, the lithologic properties and stress fields of the coal cycles typically produce fractures that are higher than they are long ("length" refers to horizontal distance from the well bore) (Morales et al., 1990; Zuber et al., 1990; Holditch et al., 1989; Palmer and Sparks, 1990; Jones and Schraufnagel, 1991; Steidl, 1991; Wright, 1992; Palmer et al., 1991b and 1993a).

In the Black Warrior Basin of Alabama, hydraulic fractures created in coalbed methane deposits are able to grow much higher than some fractures in "conventional" gas reservoirs. There are three primary reasons for this phenomenon:

- Due to coal's low modulus of elasticity (i.e., brittleness, stiffness) and complex fracture geometries, high pressures are required to fracture coal hydraulically (500 to 2,000 pounds per square inch (psi), or 0.7 to 2.0 psi/ft), and high treatment pressure often causes preferential extension of the fracture in the vertical dimension (Jones et al., 1987; Reeves et al., 1987; Morales et al., 1990; Palmer et al., 1991a);

- The economics of coalbed methane production in this basin requires tall fractures that penetrate several coal seams. The coal seams are typically thin (1 to 12 inches) and economically viable production requires the drainage of as many seams as possible. Because coal seams may be vertically separated by up to hundreds of feet of intervening rocks, operators usually design fracture treatments to enhance the vertical dimension and might perform several fracture treatments within a single well (Ely, et al., 1990; Holditch, 1990; Saulsberry et al., 1990; Spafford, 1991; Holditch, 1993); and,

- The other rocks within the Pottsville coal cycles (jointed mudstone and sandstone) fracture much more easily than coal (Teufel and Clark, 1981; Saulsberry et al., 1990; Jones and Scraufnagel, 1991; Spafford, 1991). Because there are no significant barriers to fracture height (Simonson et al., 1978; Ely et al., 1990; Palmer et al., 1991a), vertical fractures in the Black Warrior basin typically penetrate several thin coalbeds and hundreds of feet of intervening rocks (Teufel and Clark, 1981; Hanson et al., 1987; Holditch et al., 1989; Ely et al., 1990; Palmer et al., 1991c; Schraufnagel et al., 1991; Spafford, 1991; Palmer et al., 1993b) (Figure A2-11).

Mined-through studies in the Black Warrior Basin identified many instances where thin (less than 1-foot thick) shales overlying targeted coalbeds were fractured. Penetration into layers above the coal was observed in more than 80 percent of the fractures intercepted by mines underground in the Black Warrior Basin (Diamond, 1987b). Some fractures continued completely through very thin shales (Diamond, 1987a and b). These studies did not conduct a systematic assessment of the extent of the vertical fractures through and above the roof rock shales.

Several researchers conclude (based on pressure behavior during fracturing and several examples where mines penetrated hydraulic fractures) that shallow fractures have a horizontal component as described below:

- Fractures that are created at shallow depth typically have more of a horizontal component and less of a vertical component. The vertical component is most likely due to the presence of vertical natural fractures (cleats and joints) as pre-existing planes of weakness from which vertical fractures can initiate.

- Fractures created at a greater depth can propagate vertically to shallower depth, and develop a horizontal component. In these “T-fractures”, the fracture tip may fill with coal fines and/or intercept a zone of stress contrast, which causes the fracture to “turn” and to develop horizontally.

As noted above, penetration of the layers above the coal was observed in more than 80 percent of the fractures intercepted by mines underground in the Black Warrior Basin (Diamond, 1987b), but, as coals become shallower, the potential for fracture height growth decreases. In general, horizontal fractures are most likely to exist at shallow depths (less than 1,000 feet). As depths increase, it is more likely that a simple vertical fracture will occur (Gas Research Institute, 1995).

Sand is the most common proppant used in coalbed methane treatments in Alabama. The amount of sand injected per fracture treatment ranges from 10,000 to 120,000 pounds (Holditch et al., 1989; Palmer et al., 1991b and 1993a). Fracture widths in the formation vary from 0.5 inches to closed (i.e., no proppant emplaced), depending on distance from wellbore and efficiency of the proppant displacement into the length of the fracture (Palmer and Sparks, 1990; Palmer et al., 1993a; Steidl, 1993).

Fracturing fluid (30,000 to 200,000 gallons per treatment) is injected at a rate of 5 to 50 barrels per minute (which equals 210 to 2,100 gallons per minute) at injection pressures ranging from 500 to 2,300 psi (Palmer et al., 1989 and 1993b; Holditch et al., 1989; Pashin and Hinkle, 1997). The most common constituent of fracturing fluid is plain water. Several researchers conclude that approximately 75 percent of all coalbed methane wells in Alabama were fractured using cross-linked gel fluids (Palmer et al., 1993a; Pashin and Hinkle, 1997).

According to service companies, diesel fuel is no longer used as a component of fracturing fluids in Alabama. In addition, additives that could introduce chemicals exceeding maximum contaminant levels (MCLs) are no longer used in fracturing fluids in Alabama.

Table A2-1 presents some data concerning the general chemical makeup of common fracturing fluids used in Alabama from literature published prior to the Alabama hydraulic fracturing regulation (Economides and Nolte, 1989; Penny et al., 1991). In addition, most gel fluids utilize a breaker compound (usually a borate or persulfate compound or an enzyme, at 2 lb/1,000 gal) to allow post-treatment thinning and easier recovery of gels from the fracture. Several researchers conclude that approximately 75 percent of all coalbed methane wells in Alabama were fractured using cross-linked gel fluids (Palmer et al., 1993a; Pashin and Hinkle, 1997).

According to Hunt and Steele (1992), environmental regulations restrict local disposal of used fracturing fluids, and fracturing fluids are transported to regulated disposal sites. Robb and Spafford (1991) reported that acids were used to fracture production zones as shallow as 400 feet deep.

In fracture treatments of wells in homogeneous formations in conventional gas fields, injection is temporary and the majority of fracturing fluid is subsequently pumped back up through the well when production resumes.

There are limited data in the literature concerning the volume of fracturing fluids subsequently pumped back to the well after stimulation has ceased. Palmer et al. (1991b) found that only 61 percent of fracturing fluids were recovered during production sampling of a coalbed well in the Black Warrior Basin of Alabama, and projected that 20 to 30 percent would remain in the formation.

2.4 Summary

Coalbed methane development and hydraulic fracturing in the Black Warrior Basin of Alabama takes place within a USDW, the Pottsville formation. Some portions of the Pottsville Formation contain waters which meet the quality criteria of less than 10,000 mg/L TDS for a USDW. Some waters in the Pottsville Formation do not meet the definition of a USDW and have TDS levels that are considerably higher than 10,000 mg/L (Alabama Oil and Gas Board, 2002).

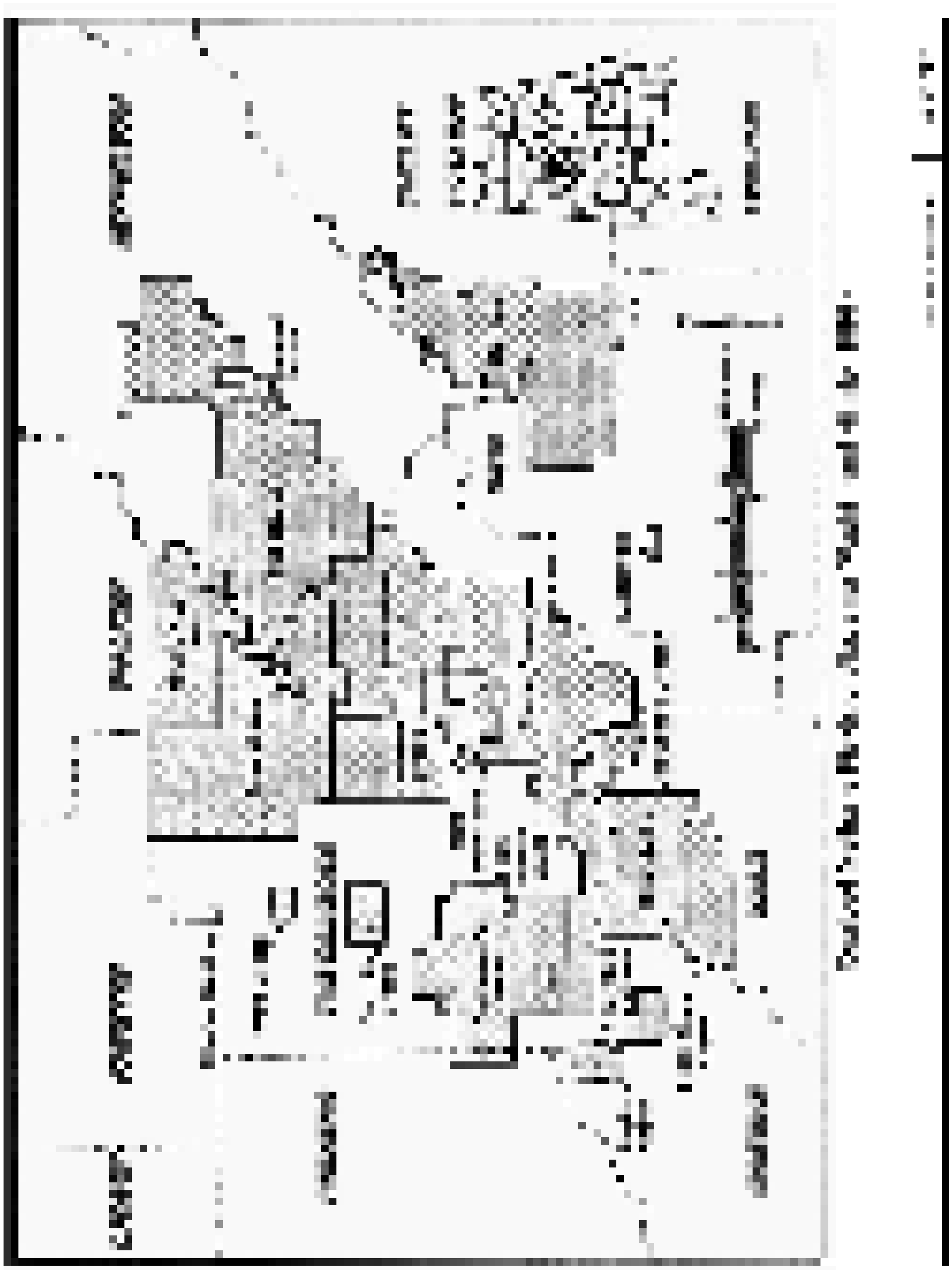
According to service companies, diesel fuel is no longer used as a component of fracturing fluids in Alabama. In addition, additives that could introduce chemicals exceeding MCLs are no longer used in fracturing fluids in Alabama.

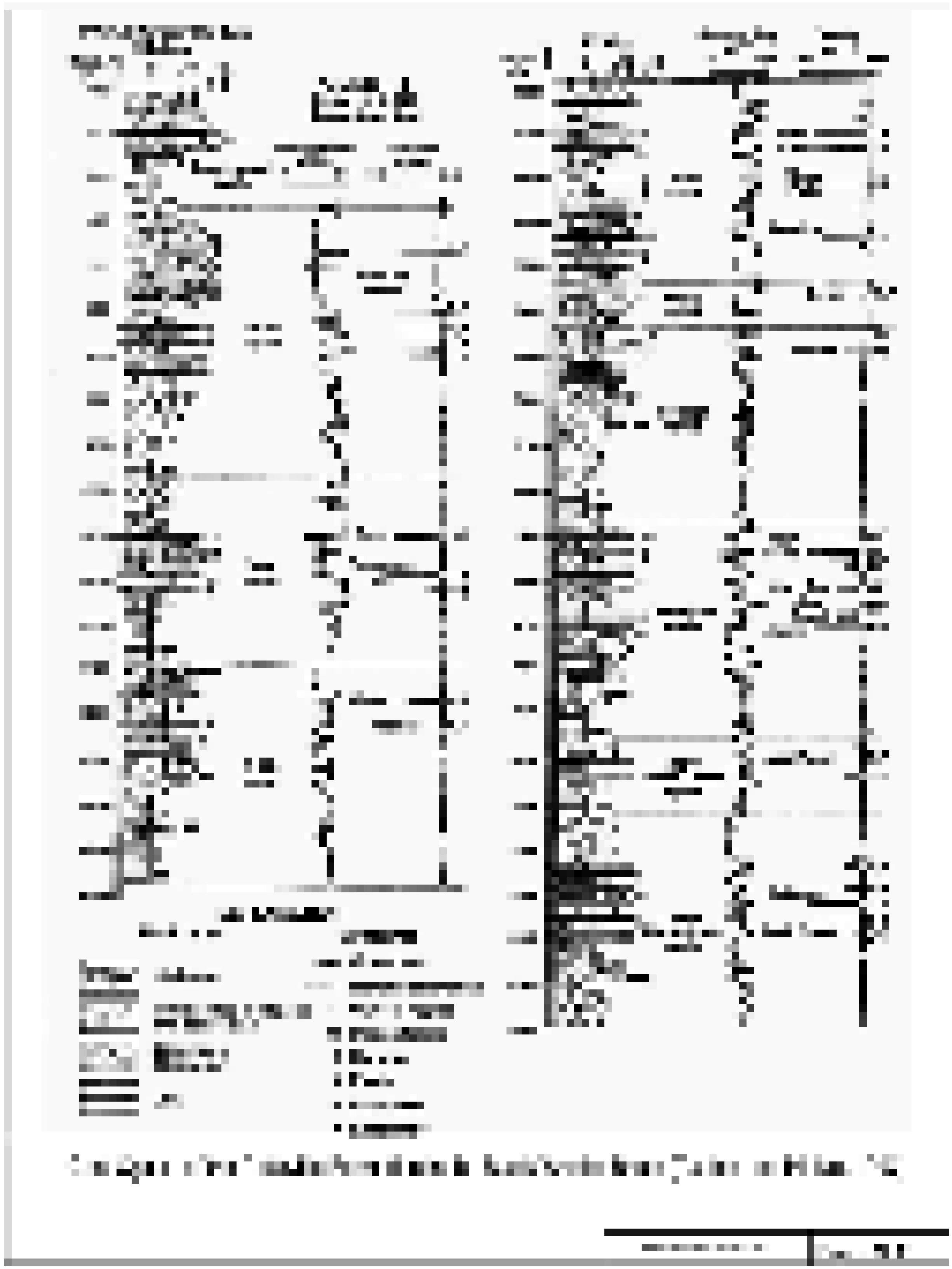
In the Pottsville Formation, the lack of a significant vertical barrier can provide for extensive fracture height growth (Holditch et al., 1989; Lambert et al., 1989; Ely et al., 1990; Saulsberry et al., 1990; Palmer and Sparks, 1990; Spafford, 1991; Palmer et al., 1991a and 1993a; Spafford et al., 1993; Gas Research Institute, 1995). Mined-through studies in the Black Warrior Basin identified many instances where thin (less than 1-foot thick) shales overlying targeted coalbeds were fractured. Penetration into layers above the coal, which are typically shale, was observed in more than 80 percent of the fractures intercepted by mines underground in the Black Warrior Basin (Diamond, 1987b).

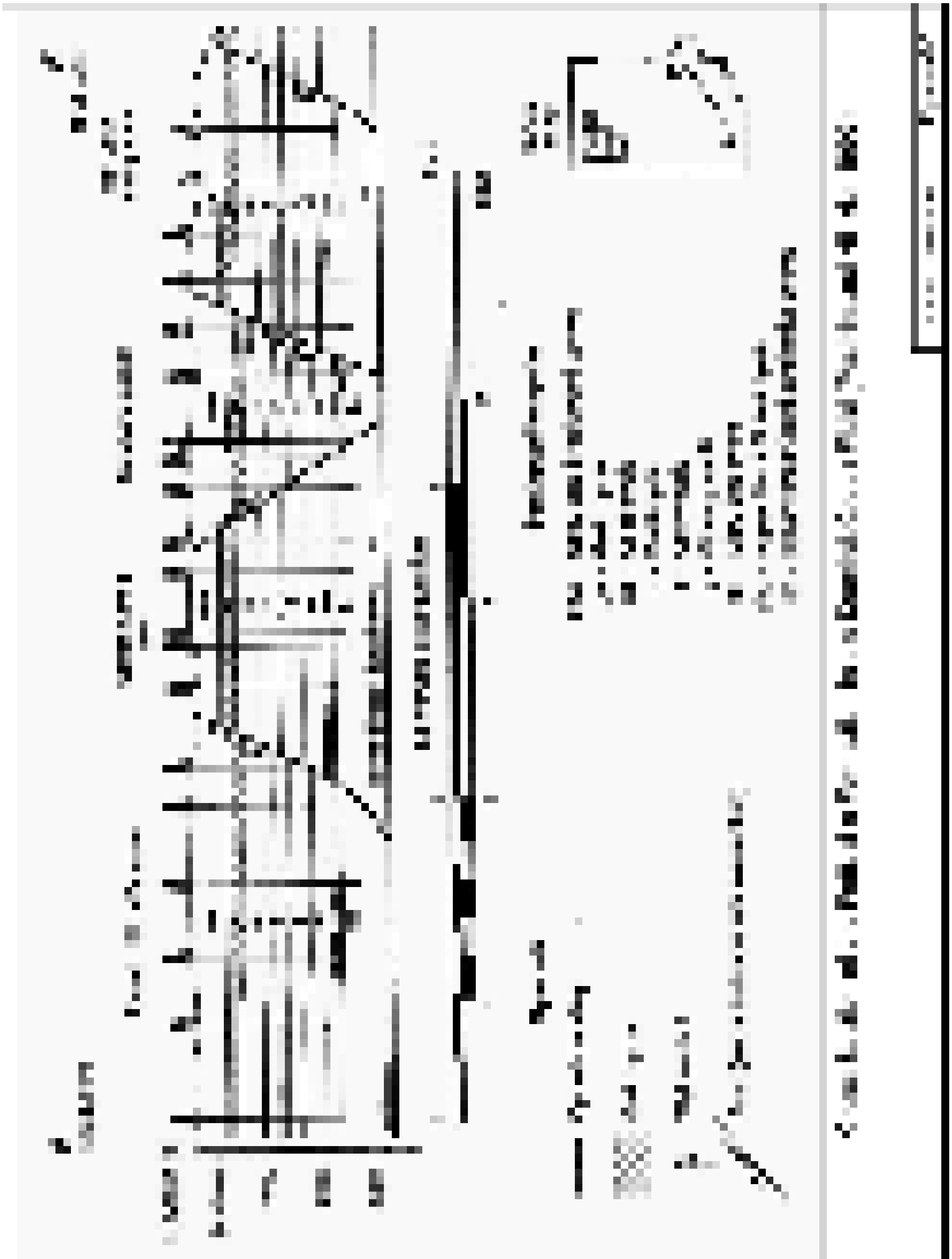
Table A2-1. Chemical Components Previously Used in Typical Fracturing/Stimulation Fluids for Alabama Coalbed Methane Wells

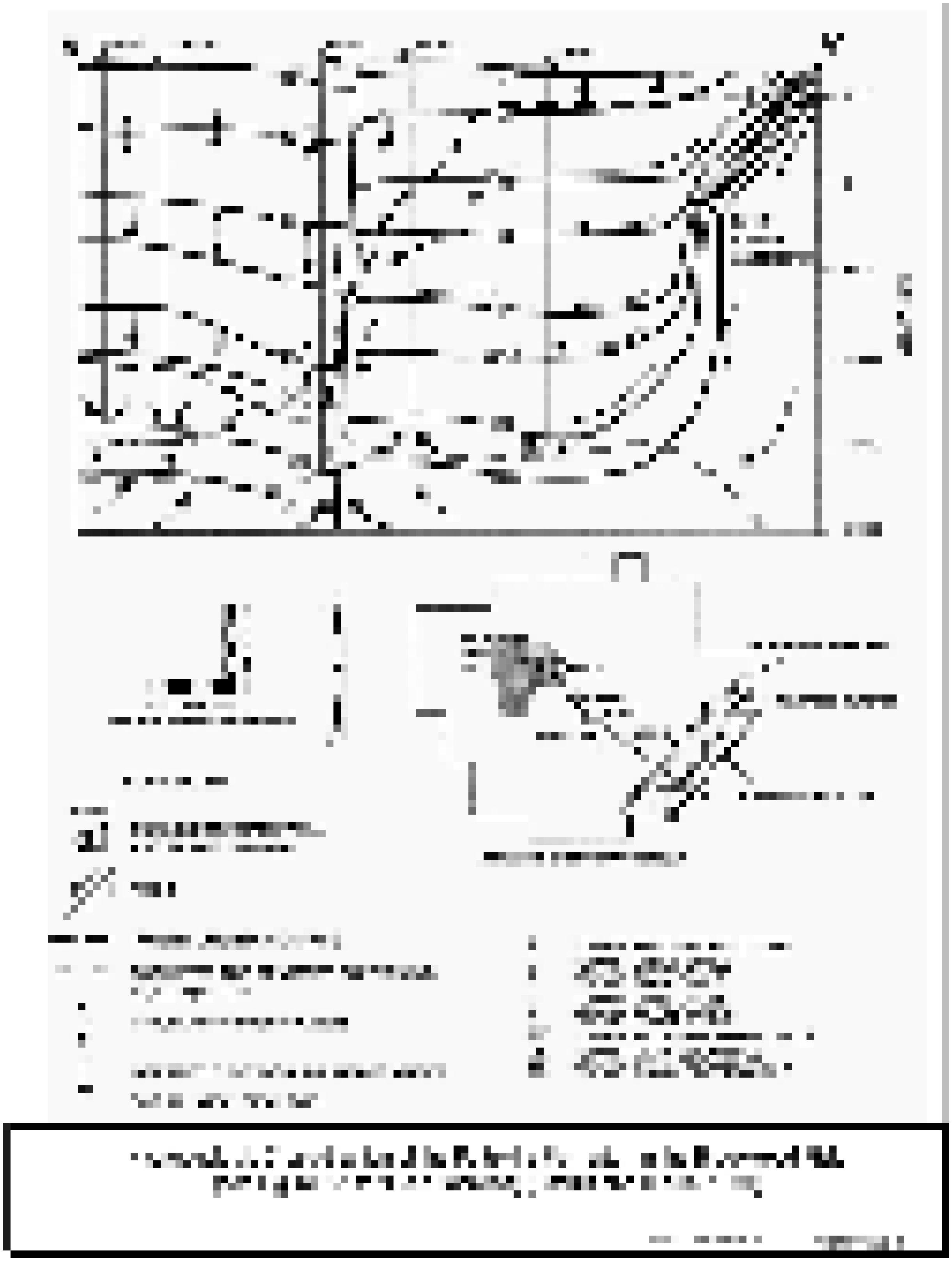
Type of Stimulation Fluid	Composition	pH
Fluids		
Hydrochloric acid	15% HCl water solution	<1-3
"Slick" water	water-soluble solvent as viscosity reducer (% unknown)	NA
Diesel oil	NA	NA
Gels[1]		
R-F	3% resorcinol, 3% formaldehyde, 0.5% KCl, 0.4% $NaHCO_3$	6.5
Pfizer Flocon 4800	0.4% xanthan, 154 ppm Cr^{3+} (as $CrCl_3$), 0.5% KCl	4.0
Marathon MARCIT	1.4% polyacrylamide (HPAM), 636 ppm Cr^{3+} (as acetate), 1% NaCl	6.0
DuPont LuDox SM	10% colloidal silica, 0.7% NaCl	8.2
CPAM crosslinked with Pfizer Floperm 500	0.4% cationic polyacrylamide (CPAM), 1520 ppm glyoxal, 2% KCl	7.3
Drilling Specialties HE-100 Crosslinked	0.3% HPAM-AMPS, 100 ppm Cr^{3+} (as acetate), 2% KCl	5.0

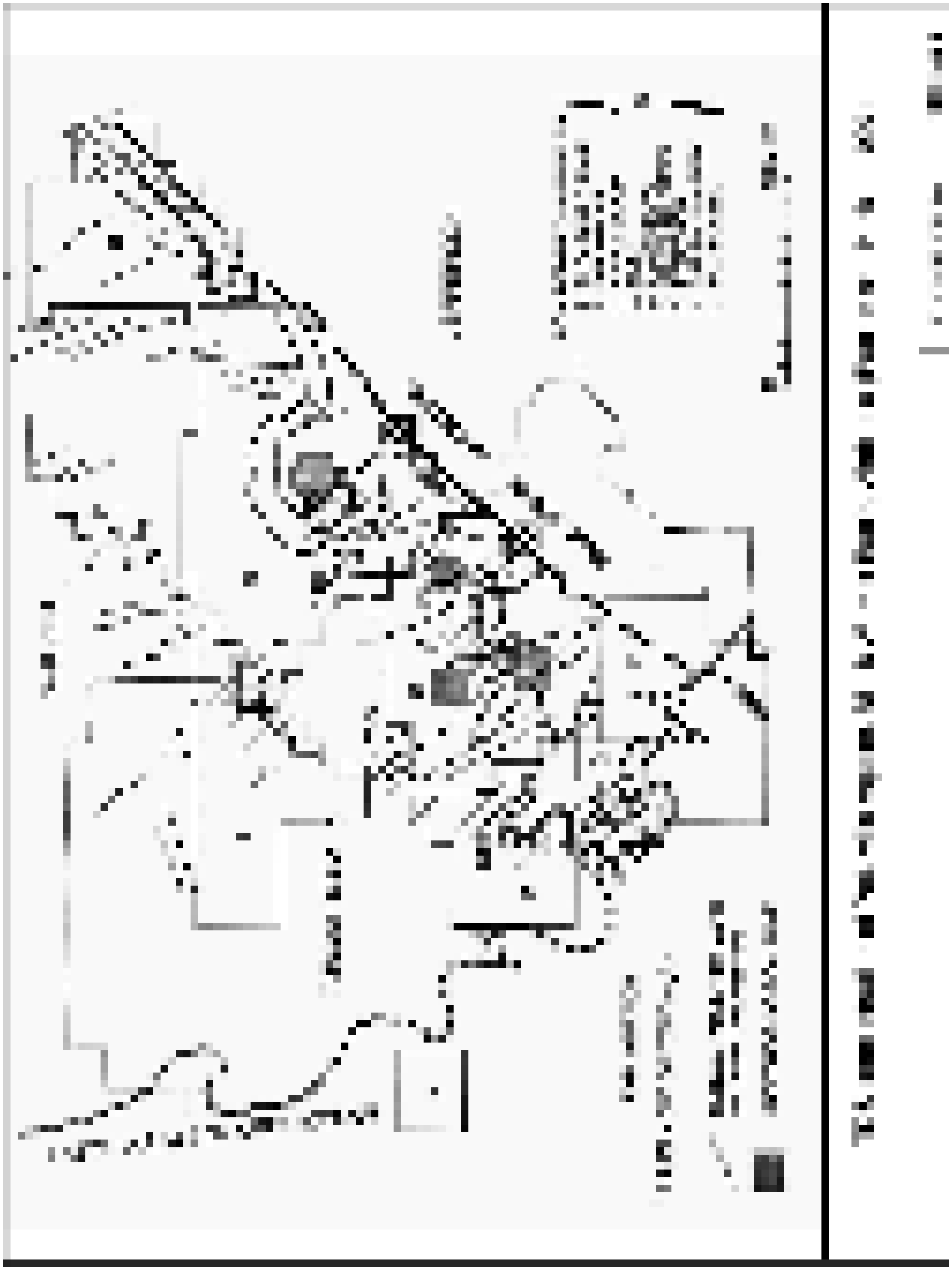
[1] Gels are typically mixed at a ratio of 40 lbs. per 1000 gal. water; compositions shown are "as mixed".

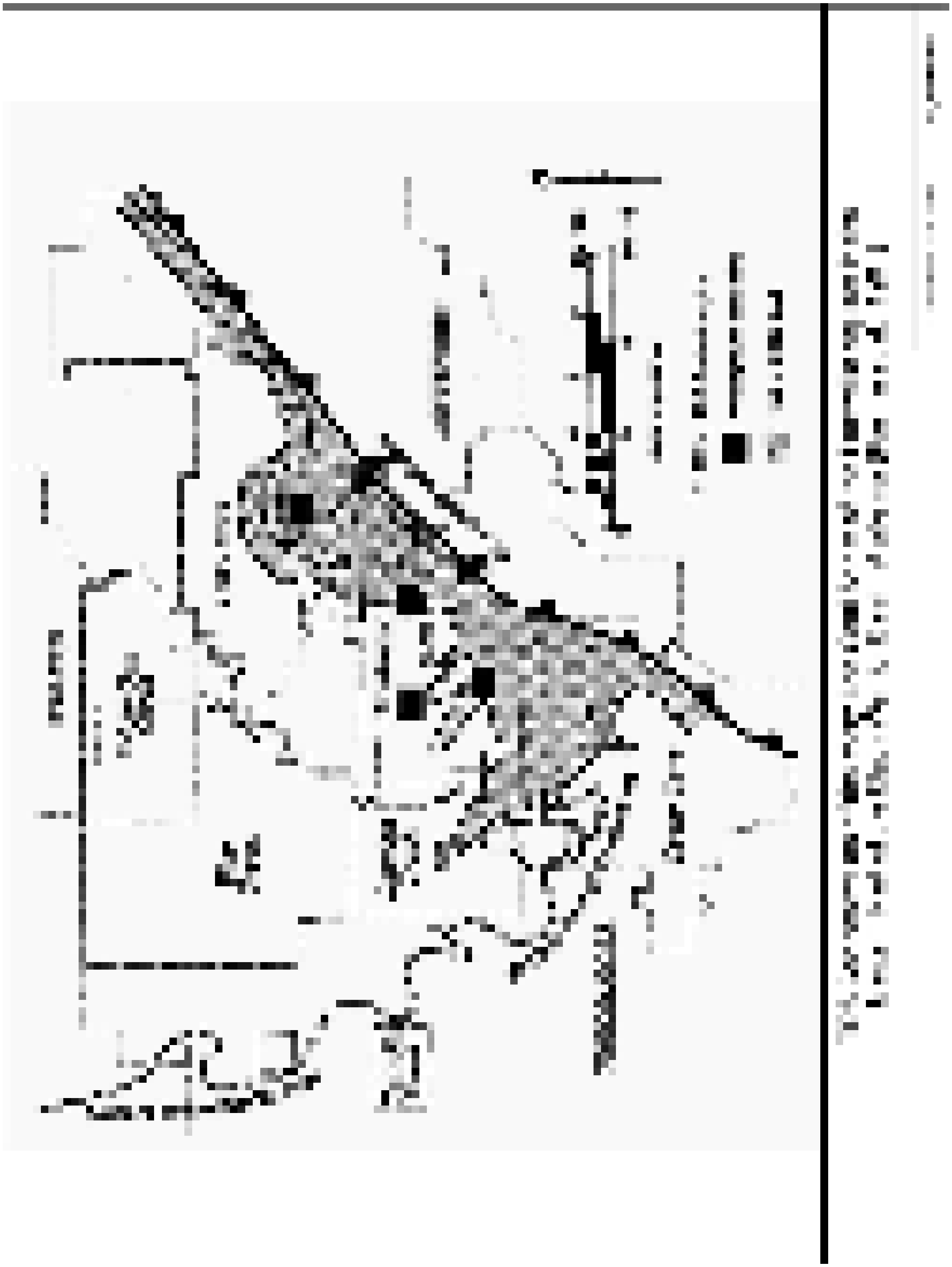


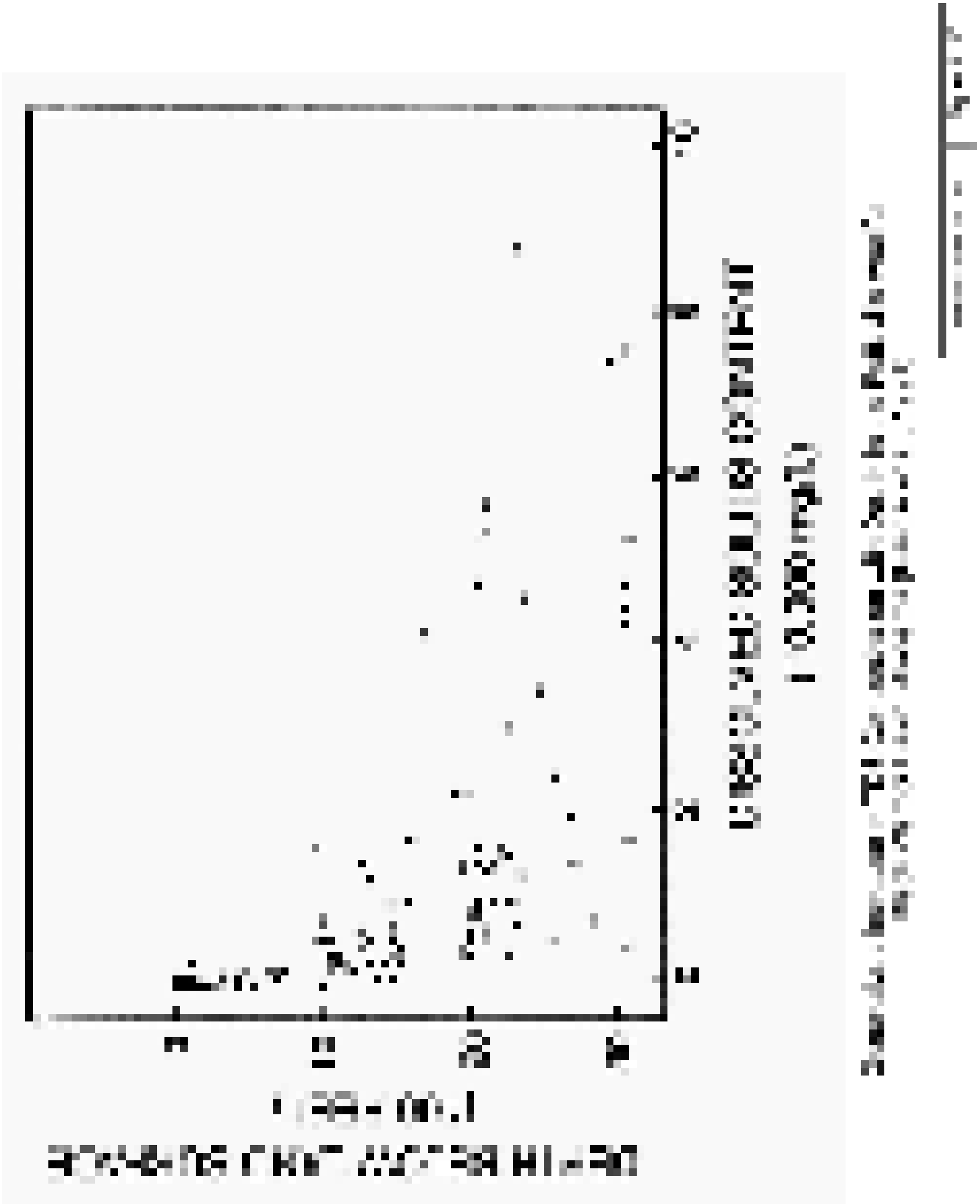


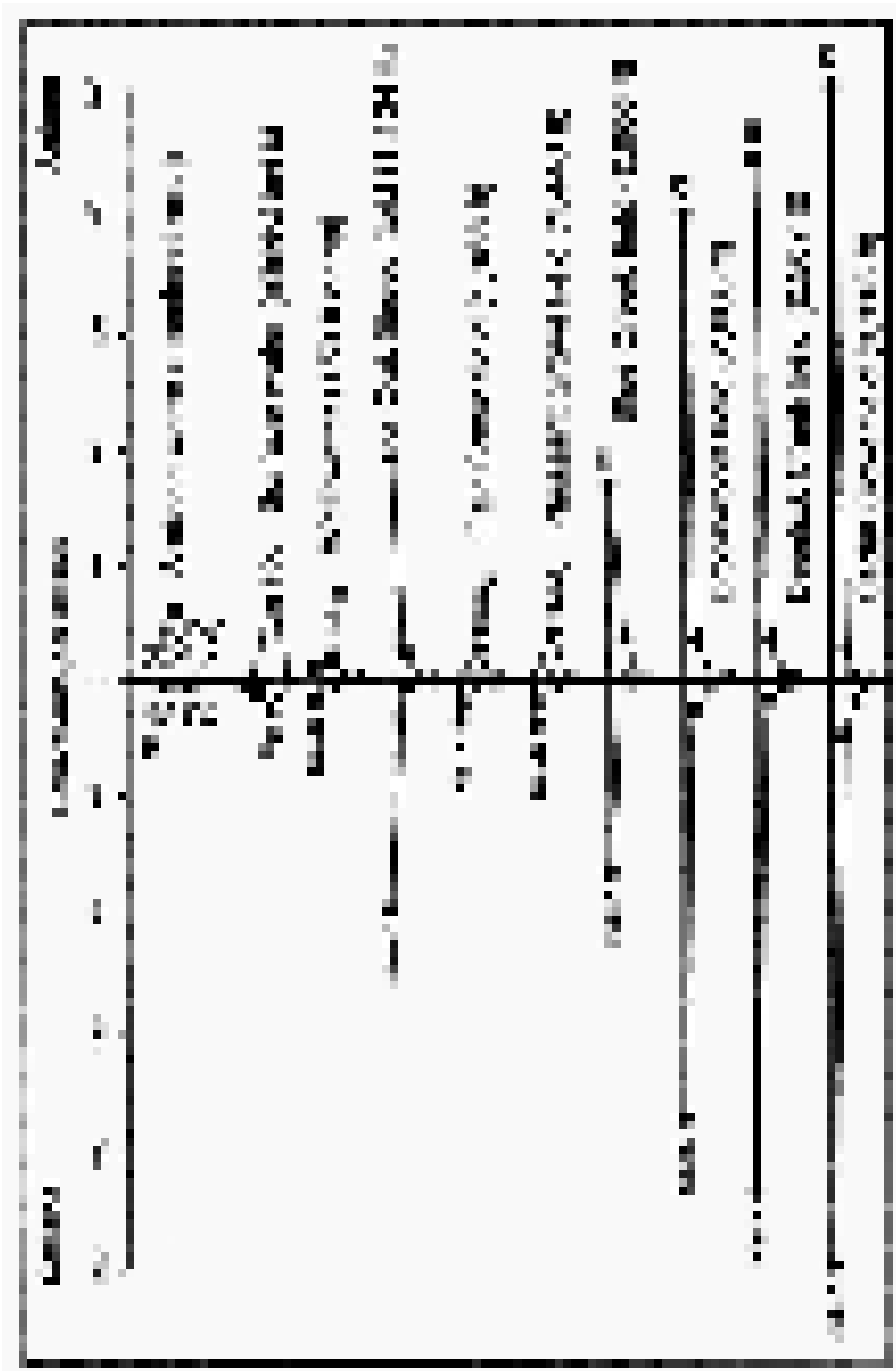


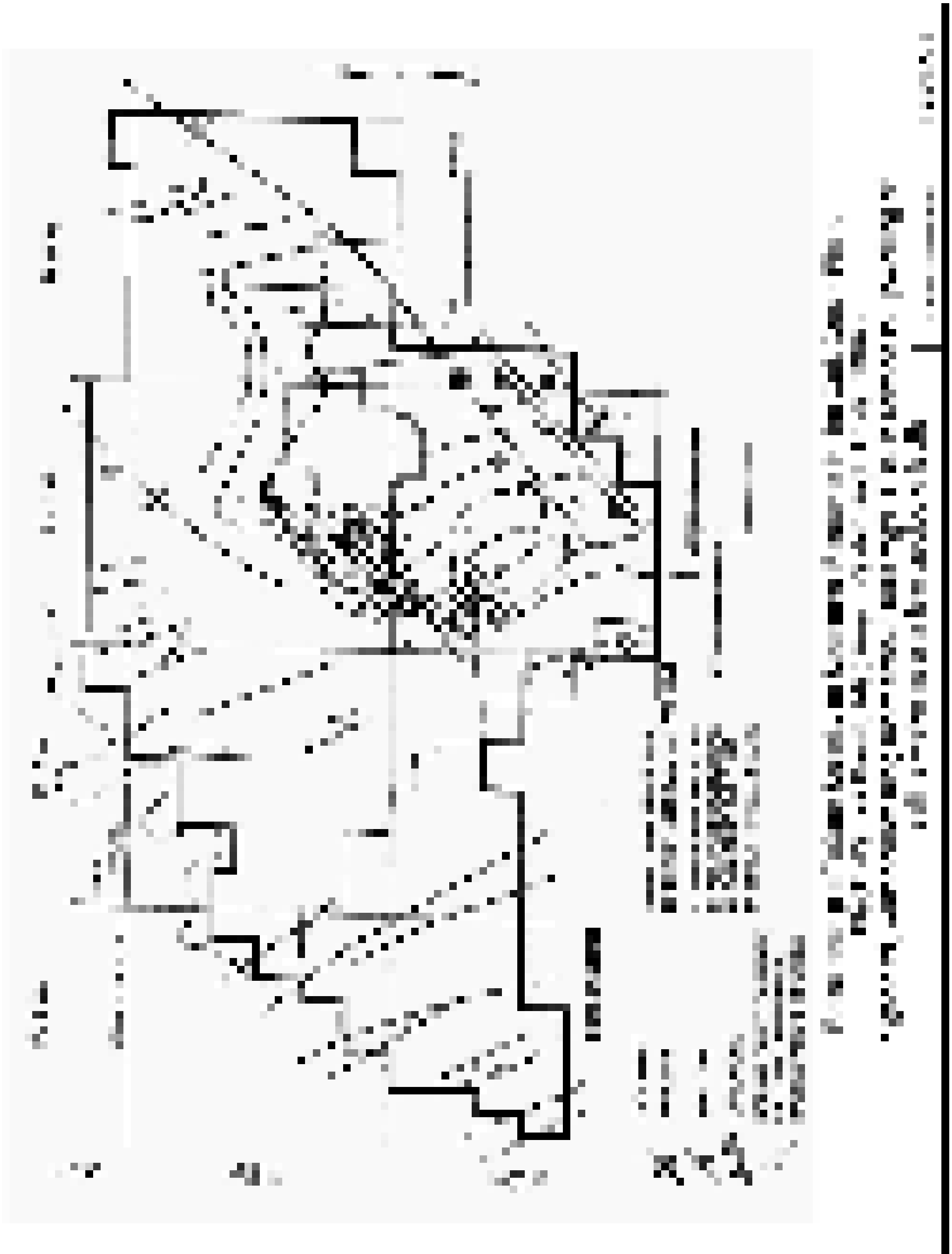


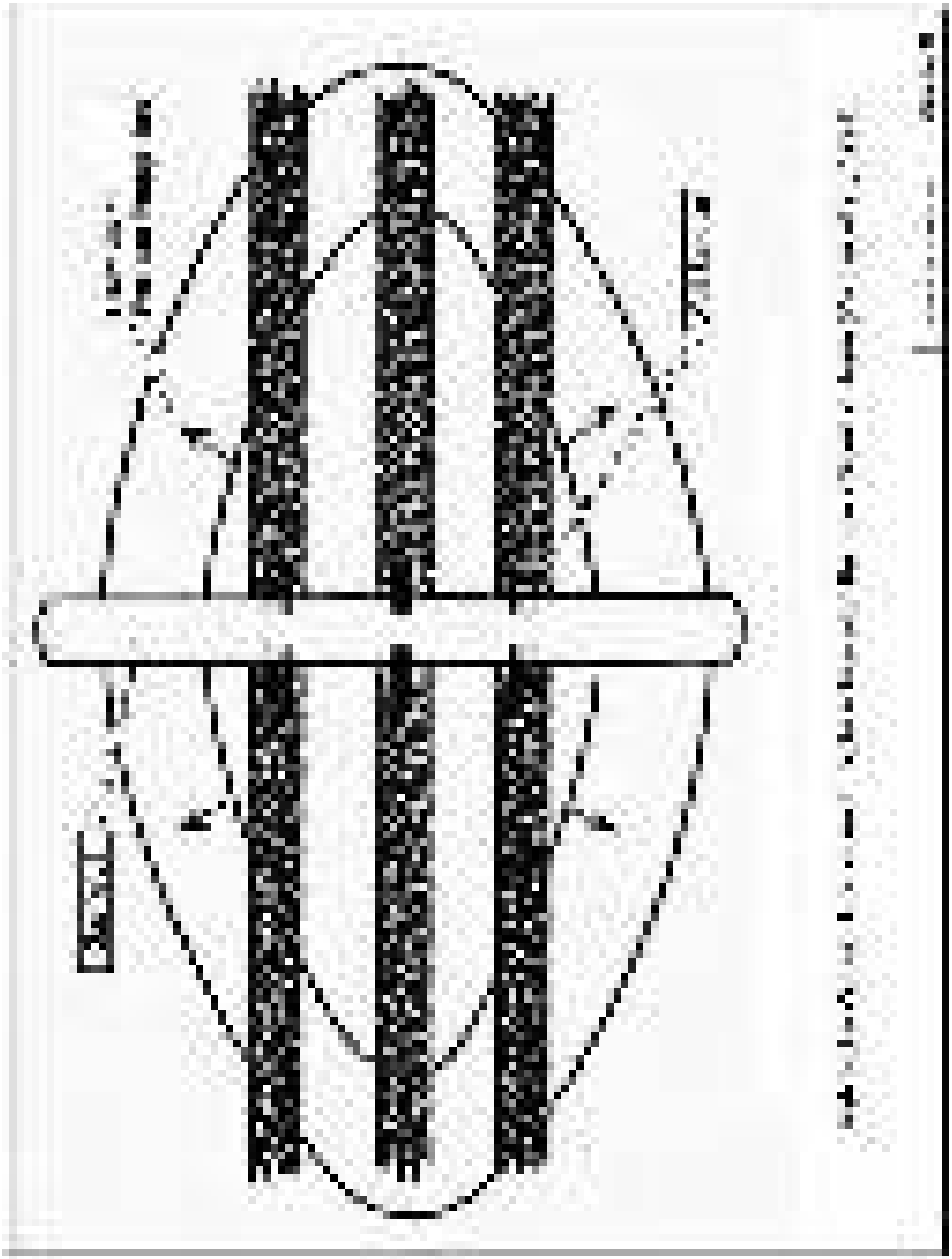












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Attachment 3
The Piceance Basin

The Piceance Coal Basin is entirely within the northwest corner of Colorado. (Figure A3-1). The coalbed methane reservoirs are found in the Upper Cretaceous Mesaverde Group, which covers about 7,225 square miles and ranges in thickness from about 2,000 feet on the west to about 6,500 feet on the east side of the basin (Johnson, 1989). It is estimated that 80 trillion to 136 trillion cubic feet (Tcf) of gas are contained in coalbeds within the basin (Tyler et al., 1998). Total coalbed methane production was 1.2 billion cubic feet in 2000 (GTI, 2002).

3.1 Basin Geology

The Piceance is a northwest trending asymmetrical, Laramide-age basin in the Rocky Mountain foreland with gently dipping western and southwestern flanks and a sharply upturned eastern flank (Figure A3-1) (Tremain and Tyler, 1997). The Douglas Creek Arch bounds the basin on the northwest, and separates it from the Uinta Coal Basin, which lies almost entirely in Utah. The Mesaverde Group is sharply upturned to near vertical along the Grand Hogback, which forms the eastern boundary of the basin and separates the basin from the White River uplift to the east. Most of the Piceance Basin's coal deposits are contained in the Iles and Williams Fork Formations of the Late Cretaceous Age Mesaverde Group, which are approximately 100 to 65 million years in age (McFall et al., 1986). These formations composed of sandstone and shale, were deposited in a series of regressive marine environments (McFall et al., 1986; Johnson, 1989). It is believed that the coals were deposited in marine transitional, brackish, interdistributary marshes and freshwater deltaic swamps (Collins, 1976 in McFall et al., 1986). Figure A3-2 presents a stratigraphic section shown with a gamma ray-induction log from the Barrett 1-27 Arco Deep well (Reinecke et al., 1991). The Mesaverde Group is underlain by the marine Mancos Shale and overlain by the lower Tertiary Age Fort Union and Wasatch Formations, which consist of fluvial sandstones and shales. The Mancos Shale, Fort Union, and Wasatch Formations are essentially barren of coals (McFall et al., 1986). Depths to the coal-bearing sediments vary from outcrops around the margins of the basin (Figure A3-1) to more than 12,000 feet in the deepest part of the basin (Tyler et al., 1996).

The major fold structure of the Piceance Basin is the Grand Hogback Monocline, formed as the White River Uplift was uplifted and thrust westward during the Laramide Orogeny in Late Cretaceous through Eocene time (McFall et al., 1986). Broad folds, such as the Crystal Creek and Rangley Syncline, trend northwest to southeast, and generally parallel to the axis of the basin (Figure A3-1). Intrusions occur throughout the southeast part of the basin, locally elevating coal ranks to as high as anthracite grade. A buried laccolith intrusion is thought to be present under a coal basin anticline along the southeast margin of the basin (Figure A3-1) where high quality coking coal was mined since the 1800s (Collins, 1976).

Coalbed methane reservoirs occur exclusively in the Upper Cretaceous Mesaverde Group (Figure A3-2), which covers an area of approximately 7,255 square miles (Tremain and Tyler, 1997). Depths to the Mesaverde Group range from outcrop to greater than 12,000 feet along the axis of the basin (Tyler et al., 1996; Tremain and Tyler, 1997). Two-thirds of the coalbed methane occurs in coals deeper than 5,000 feet, making the Piceance Basin one of the deepest coalbed methane areas in the United States (Quarterly Review, August 1993).

The major coalbed methane target, the Cameo-Wheeler-Fairfield coal zone (Figure A3-3), is contained within the Williams Fork Formation of the Mesaverde Group and holds approximately 80 to 136 Tcf of coalbed methane (Tyler et al., 1998). This coal zone ranges in thickness from 300 to 600 feet, and lies more than 6,000 feet below the ground surface over a large portion of the basin (Tyler et al., 1998). Individual coal seams of up to 20 to 35 feet thick can be found within the group, with net coal thickness of the Williams Fork Formation averaging 80 to 150 feet thick. In 1991, at the Grand Valley field (Figure A3-4), there were 23 coalbed methane wells and 18 conventional gas wells (Reinecke et al., 1991). However, in 1984, most wells at the Rulison field (Figure A3-4) were conventional gas wells.

Initially, it was anticipated that coalbed methane wells in the sandstones and coals of the Cameo Zone would have high production rates of water. However, testing later showed that they produced very little water (Reinecke et al., 1991). Both the sandstones and coalbeds are tight, poorly permeable, and are generally saturated with gas rather than water or a mixture of water and gas. The dynamic flow of a hydrologic system enhances the collection of gas in traps, but in much of the Piceance Basin that flow is not present because of the over-pressuring and saturation with gas.

Consequently, the conventional models for coalbed methane accumulation developed for other basins do not apply well for exploration and development in the Piceance Basin. Tyler et al. (1996) concluded, “very low permeability and extensive hydrocarbon overpressure indicate that meteoric recharge, and, hence, hydropressure, is limited to the basin margins and that long-distance migration of groundwater is controlled by fault systems.” Recharge is limited along the eastern and northeastern margins of the basin because of offsetting faults, but zones of transition between hydropressure and hydrocarbon overpressure in the western part of the basin and on the flanks of the Divide Creek Anticline in the southeastern part of the basin may possess better coalbed methane potential, as indicated by the exploration targets delineated in Tyler et al. (1998) (Figure A3-5).

3.2 Basin Hydrology and USDW Identification

The Piceance Basin contains both alluvial and bedrock aquifers. Unconsolidated alluvial aquifers are the most productive aquifers in the Piceance Basin. These alluvial deposits are narrow, and thin deposits of sand and gravel formed primarily along stream courses. The City of Meeker, Colorado is supplied by wells tapping these deposits where they are over 100 feet thick in the White River Valley (Taylor, 1987).

The most important bedrock aquifers are known as the upper and lower Piceance Basin aquifer systems. These consolidated rock aquifers are lower Tertiary Eocene in age and occur within and above the large oil shale reserves. The upper and lower aquifers are separated by the Mahogany Zone of the Parachute Creek Member (Figure A3-6). The Mahogany Zone is a poorly permeable oil shale, which retards water movement but does not stop it. Both bedrock aquifers overlie the older Cretaceous Mesaverde Group where the coal and coalbed methane are located.

The upper aquifer system is about 700 feet thick and consists of several permeable zones in the Eocene Uinta Formation and the upper part of the Parachute Creek Member of the Eocene Green River Formation. Sub-aquifers of the Uinta Formations are silty sandstone and siltstone, while those of the Parachute Creek Member of the Green River Formation are fractured dolomite marlstone. There is some primary porosity (i.e., the porosity preserved from during or shortly after sediment deposition, such as the spaces between grains) in the sandstone and the permeability of the sub-aquifers has been enhanced by natural fracturing that occurred during post-deposition deformation. Layers between the individual sub-aquifers are less permeable than the sub-aquifers themselves, but they do not prevent water movement between the sub-aquifers.

The lower aquifer system is about 900 feet thick and consists of a fractured dolomitic marlstone of part of the lower Parachute Creek Member of the Green River Formation. It is semi-confined below the Mahogany Zone and above the Garden Gulch Member of the Green River Formation and a high resistivity zone just above it (USGS, 1984 and Taylor, 1987) (Figure A3-6). Fracturing during deformation of the rocks and subsequent solution enlargement owing to dissolution of soluble evaporite minerals has increased permeability of this lower aquifer system.

Groundwater is recharged from snowmelt on high ground from where it travels down through the upper aquifer system, the Mahogany Zone, and into the lower aquifer system. The groundwater then moves laterally and/or upward discharging from both the upper and lower aquifer systems into streams (Figure A3-7). The minerals nahcolite ($NaHCO_3$), dawsonite ($NaAl(OH)_2CO_3$) and halite ($NaCl$) are present in the groundwater, and the circulation of the groundwater (with these minerals in solution) has caused enlargement of the natural fractures (Taylor, 1987). Water in the lower aquifer is reported to contain several hundred milligrams per liter (mg/L) of chloride (Taylor, 1987).

Wells in these two bedrock aquifer systems, the upper and lower Piceance Basin aquifers, typically range in depth from 500 to 2,000 feet and commonly produce between 2 to 500 gallons per minute of water (USGS, 1984). These Tertiary bedrock aquifers are stratigraphically separated from the base of the Cameo Coal Zone in the Cretaceous Mesaverde Group by from less than 1,500 feet of strata along the Douglas Creek Arch to more than 11,000 feet along the basin trough just west of the Grand Hogback (Johnson and Nuccio, 1986) (Figure A3-2).

Aquifer maps do not exist for the Piceance Basin, but water quality in the Piceance Basin is poor owing to nahcolite (sodium bicarbonate) deposits and salt beds within the basin (Graham, 2001). Only very shallow waters such as those from the surficial Green River Formation are used for drinking water (Graham, CDWR, 2001). In general, the potable water wells in the Piceance Basin extend no further than 200 feet in depth, based on well records maintained by the Colorado Division of Water Resources (CDWR). At least two wells in the area are approximately 1,000 feet in depth, but they are used for stock watering. A composite water quality sample taken from 4,637 to 5,430 feet deep within the Cameo Coal Group in the Williams Fork Formation exhibited a total dissolved solid (TDS) level of 15,500 mg/L, which is above the 10,000 TDS water quality criterion for a underground source of water (USDW) (Graham, CDWR, 2001). The produced water from coalbed methane extraction in the Piceance Basin is of such low quality that it must be disposed of in evaporation ponds or re-injected into the formation from which it came or at even greater depths (Tessin, 2001).

It is unlikely that any USDWs and coals targeted for methane production would coincide in this basin. These targeted coals are generally located at great depth, of at least 4,000 feet. The thousands of feet of stratigraphic separation between the coal gas bearing Cameo Zone and the lower aquifer system in the Green River Formation should prevent any of the effects from the hydrofracturing of gas-bearing strata from reaching either the upper or the lower bedrock aquifers.

Permeability of the coal and the surrounding sandstone and shale is generally quite low except near outcrop, creating little potential for these rocks to contain a USDW. Researchers (Reinecke et al., 1991) report that the permeability of gas-bearing coal and sandstone of the Cameo Zone is so low that the gas is over-pressured and has forced groundwater out of the zone, a condition that tends to disfavor the entrapment of methane. Tyler et al. (1998) state that high coalbed methane gas productivity requires geologic and hydrologic conditions, and that these conditions are not optimal throughout much of the Piceance Basin because of the absence of dynamic groundwater flow and the low permeability of the host rocks.

The above conditions prevail in the central part of the basin, previously favored as a coalbed methane development fairway, and heavily targeted for exploration (Nowak, 1991). However, analyses by Tyler et al. (1998) suggest that a transitional zone, between the deeply buried coal and the outcrops at the boundaries of the basin, where groundwater circulation may be sufficient to create more favorable trapping conditions (Figure A3-5), may be a better target area for coalbed methane production exploration. These exploration target zones could possibly have sufficient meteoric groundwater circulation to meet the water quality criterion of USDWs. However, Figure A3-3 shows that the depths to coals in the targeted methane producing zones (Figure A3-5) are greater than 4,000 feet below ground surface and therefore, are not likely to contain water that would meet the USDW quality criterion of less than 10,000 mg/L TDS. Currently, test-drilling information is insufficient to determine if this is the case. Nevertheless, due to the very low permeability, great depth, and expected poor water quality of the targeted coalbed methane producing zones, conflicts with USDWs are considered to be of very low probability.

3.3 Coalbed Methane Production Activity

Measurements of coal permeabilities in the Piceance Basin have shown that the deep coals typical of the basin are much less permeable than coals in top-producing coalbed methane basins such as the San Juan Basin in Colorado (Quarterly Review, 1993). Consequently, operators rely on large hydraulic fractures to produce coalbed methane from the deep, low permeability coals (Quarterly Review, 1993).

Exploration for coalbed methane began in the basin during the early 1980s, but viable commercial production did not begin until 1989 (Quarterly Review, 1993). The first well to commercially produce coalbed methane from the Piceance Basin, Exxon’s Vega No. 2 well in Mesa County, went off-line in 1983 (Quarterly Review, 1993). Amoco Production Company attempted multi-well coalbed methane development in the late 1980s, and finally ceased activity in 1989. Commercial production was finally achieved in 1989 in the Parachute fields operated by Barrett Resources. Barrett Resources drilled 68 wells in 1990 and had planned for 22 more in 1991 (Western Oil World, 1991). The wells targeted both coals and sandstone within the Cameo Coal Zone and the Mesaverde sandstones, just above the Cameo coals. Other operators soon followed suit, including Fuelco at White River Dome field in the northern part of the basin (Figure A3-1), Conquest Oil Company near Barretts Resource’s production in the central part of the basin, Chevron USA Inc., and many others. However, not all operators were successful in locating or producing coalbed gas. Ultimately, Barrett found the sandstones to be far more productive than the coalbeds, and attempts to complete wells in the coalbeds were largely abandoned.

According to the Colorado Geological Survey (2002), some operators are having success in their pilot coalbed methane production program in White River Dome Field northwest of Meeker. Their success is attributed to the extensive natural fracturing found in the coal seams at White River Dome. Fracturing may be particularly extensive as a result of the formation of the White River anticline and the proximity to the large Danforth Hills Mesaverde outcrop. As a result, operators are taking another look at coalbed methane development in the Piceance Basin. In addition, one of the operators is drilling (but not fracturing) horizontal wells in the coal seams to take advantage of the anomalous natural fracturing found at White River Dome field. In some areas of coalbed methane potential, horizontal well technology may replace hydraulic fracturing as a method to enhance coalbed methane well performance.

Within the Cameo Coal Zone, Barrett Resources typically used 3,000 to 3,500 barrels of gelled 2% potassium chloride water with 273,000 to 437,000 pounds of sand over a maximum 450 feet of the Cameo Coal Zone to stimulate coalbed methane wells (Quarterly Review, 1993). It was shown that these hydraulic stimulations created short (100-foot), multiple fractures around the wells (Quarterly Review, August 1993). Fuel Resources Development Company used 3,000 to 10,000 barrels of gelled water and 200,000 to 1,300,000 pounds of sand to fracture their wells in the White River Dome Field (Quarterly Review, 1993). All but one of Conquest Oil Company's wells was hydraulically fractured with 1,500 barrels of water or cross-linked gel and 31,000 to 230,000 pounds of regular or resin-coated sand (Quarterly Review, 1993).

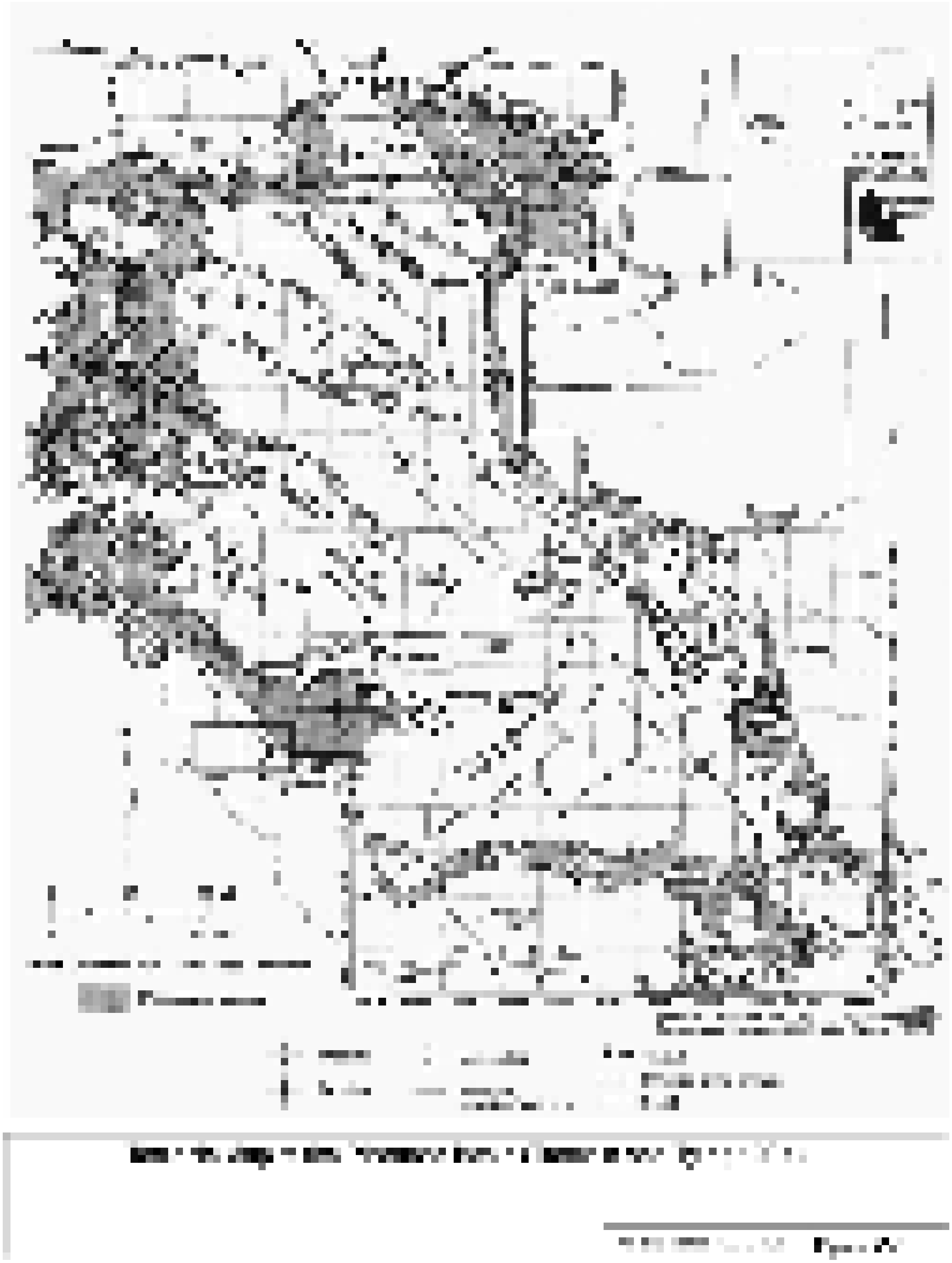
3.4 Summary

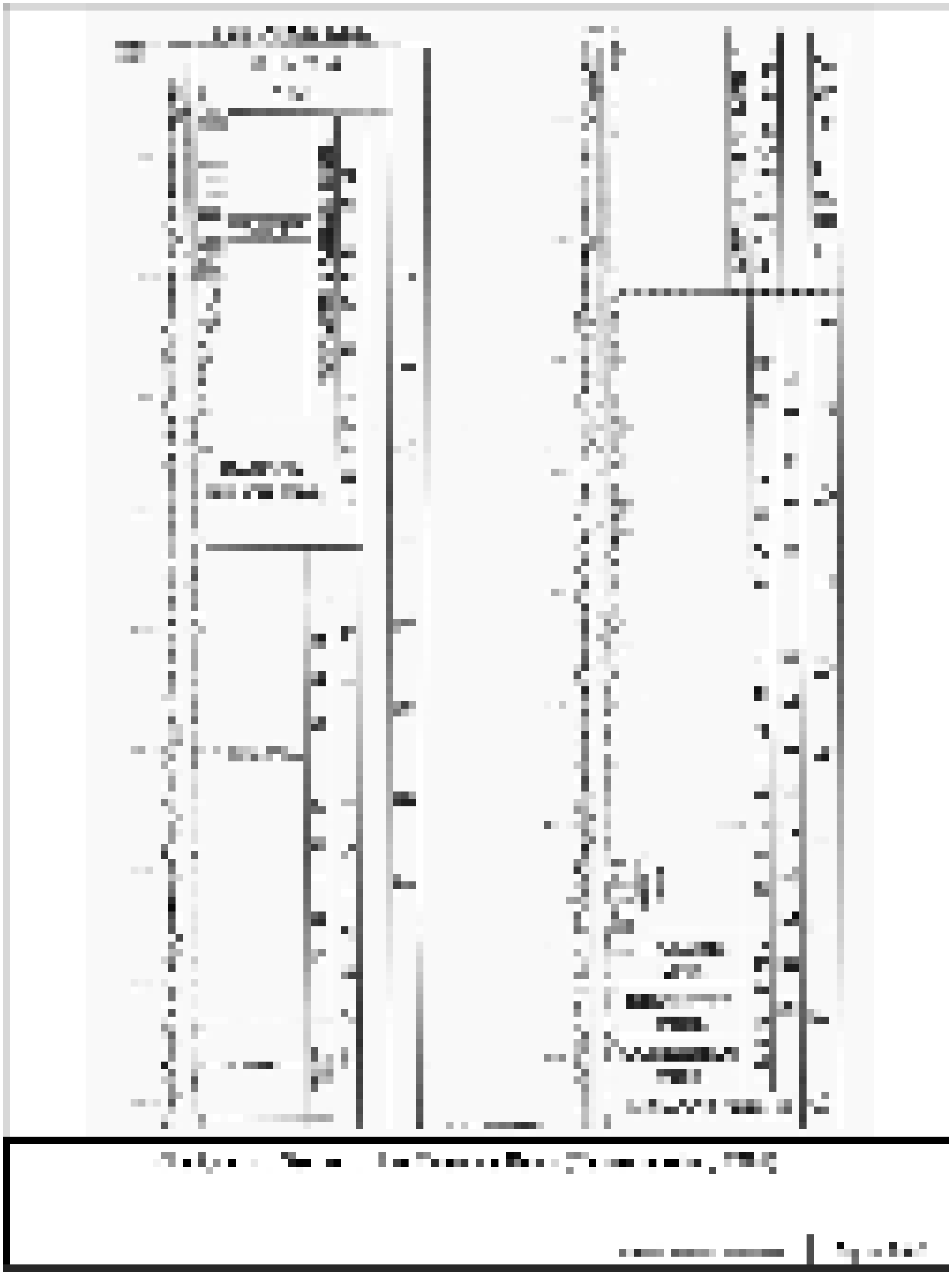
The Piceance Basin shows promise as a source for coalbed methane production based on the estimated 80 to 136 Tcf of gas contained within the Cameo-Wheeler-Fairfield coal zone (Tyler et al., 1998). However, overall low permeabilities as well as great depths to coalbeds appear to have slowed coalbed methane development in the basin. Nevertheless, a pilot program in White River Dome Field has had success in coalbed methane production, attributable primarily to the extensive natural fracturing in the area. As a result, operators are taking another look at coalbed methane development in this basin.

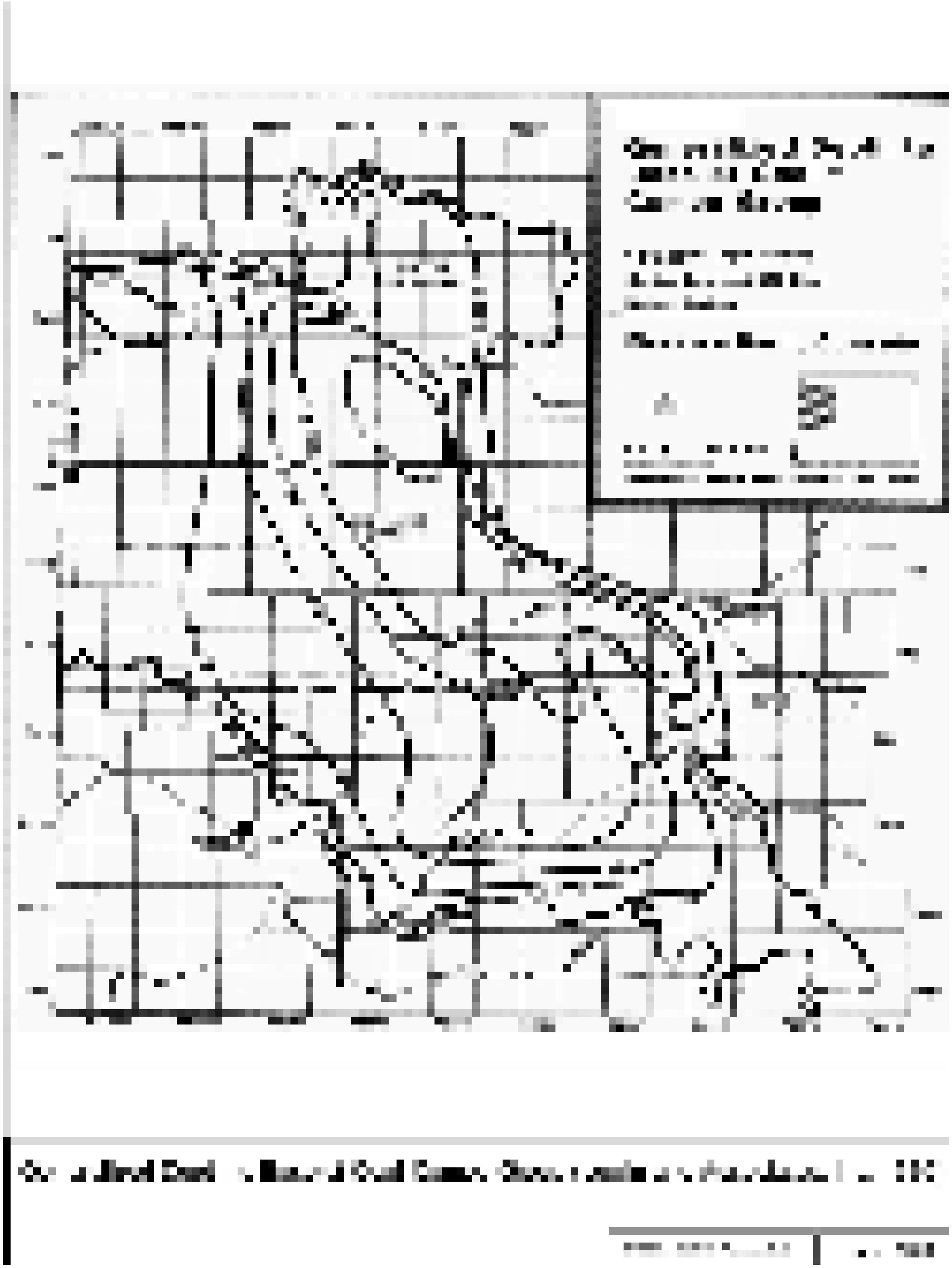
Hydraulic fracturing is the common method used to extract coalbed methane. Drilling of horizontal wells in the coal seams is a method that is being evaluated in the White River Dome Field pilot project as an alternative to hydraulic fracturing. In some areas of coalbed methane potential, horizontal well technology may replace hydraulic fracturing as a method to enhance coalbed methane performance.

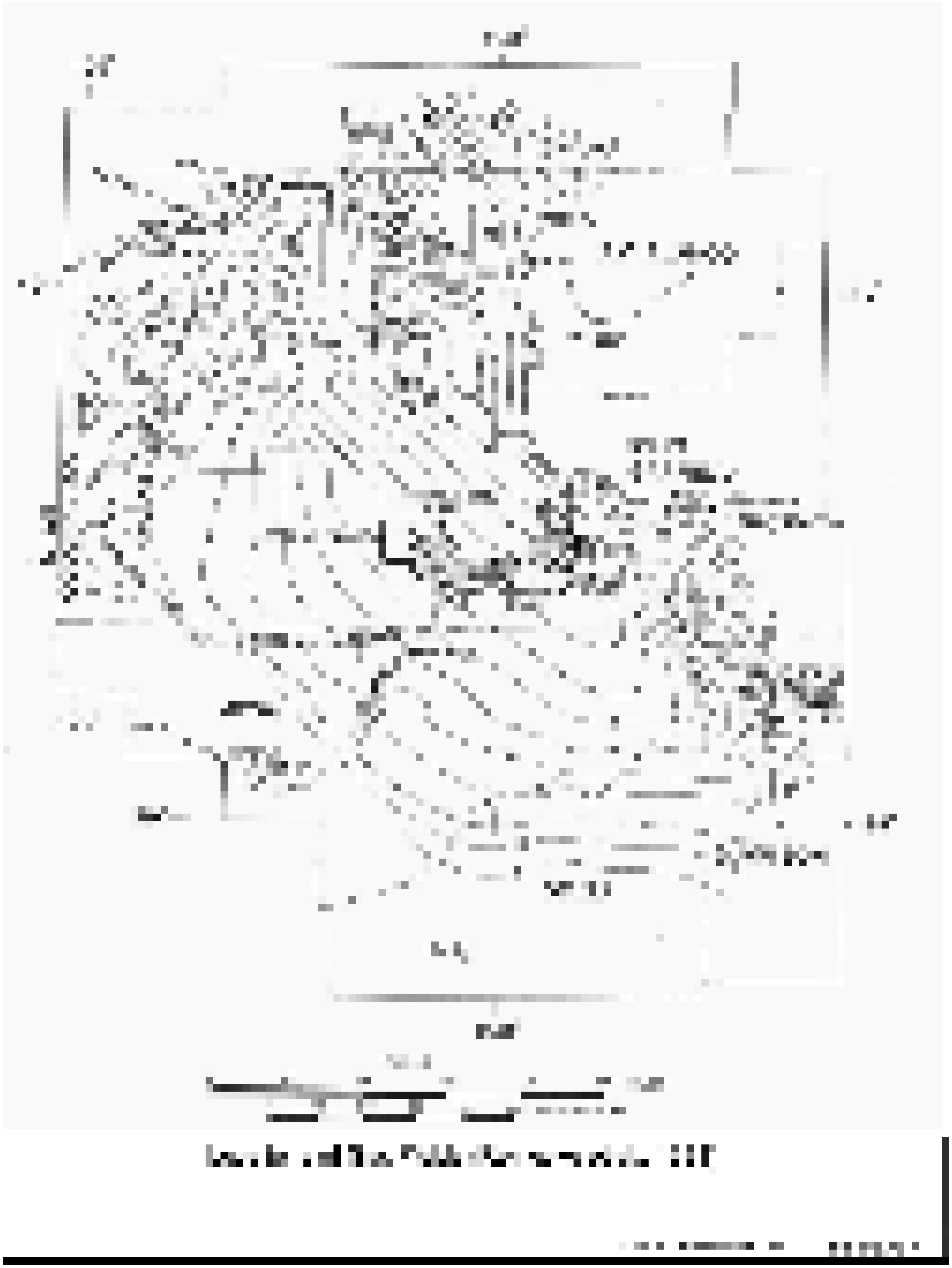
The fluids used for fracturing vary from water with sand proppant to gelled water and sand. Between 1,500 to more than 11,000 feet of strata separate the coals from the shallow USDWs, indicating that the potential for water quality contamination from hydraulic fracturing techniques is minimal. The only hydraulic fracturing fluid contamination pathway to the USDWs might be through faults or fractures extending between the deep coal layers and the shallow aquifers. The occurrence of these fractures and faults has not been substantiated in any of the literature examined for this investigation.

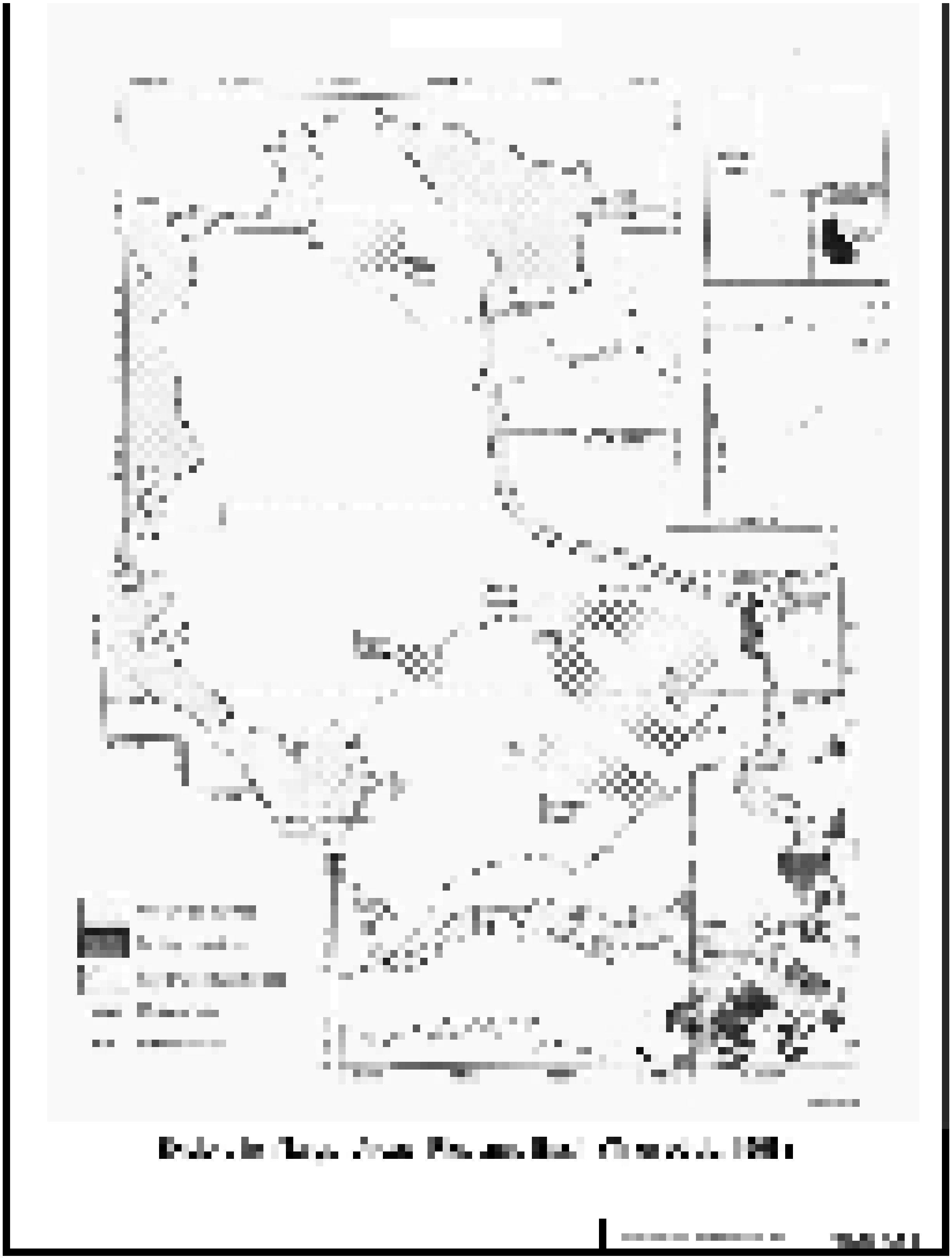
Research suggests that exploration may target areas where groundwater circulation may enhance gas accumulation in the coal and associated sandstones (Tyler et al., 1998). Under these exploration and development conditions, a USDW located in shallower Cretaceous rocks near the margins of the basin, could be affected by hydraulic fracturing. The depth to methane-bearing coals (about 6,000 feet) seems to indicate that, in the Piceance Basin, the chances of contaminating any overlying, shallower USDWs (no deeper than 1,000 feet) from injection of hydraulic fracturing fluids and subsequent subsurface fluid transport are minimal. Potable wells in the Piceance Basin generally extend no further than 200 feet in depth. The coalbed methane producing Cameo Zone and the deepest known aquifer, the lower bedrock aquifer, have a stratigraphic separation of over 6,000 feet.

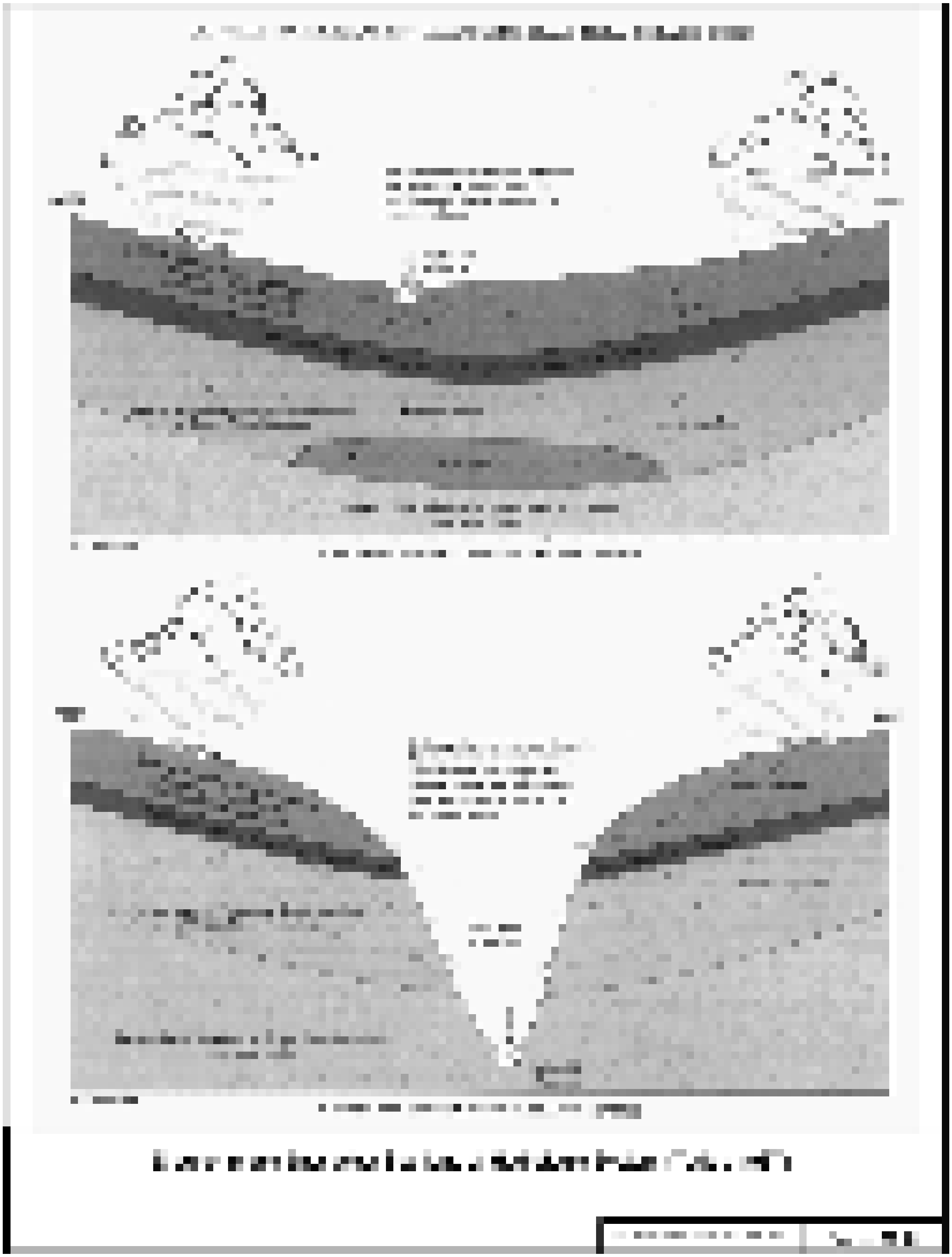


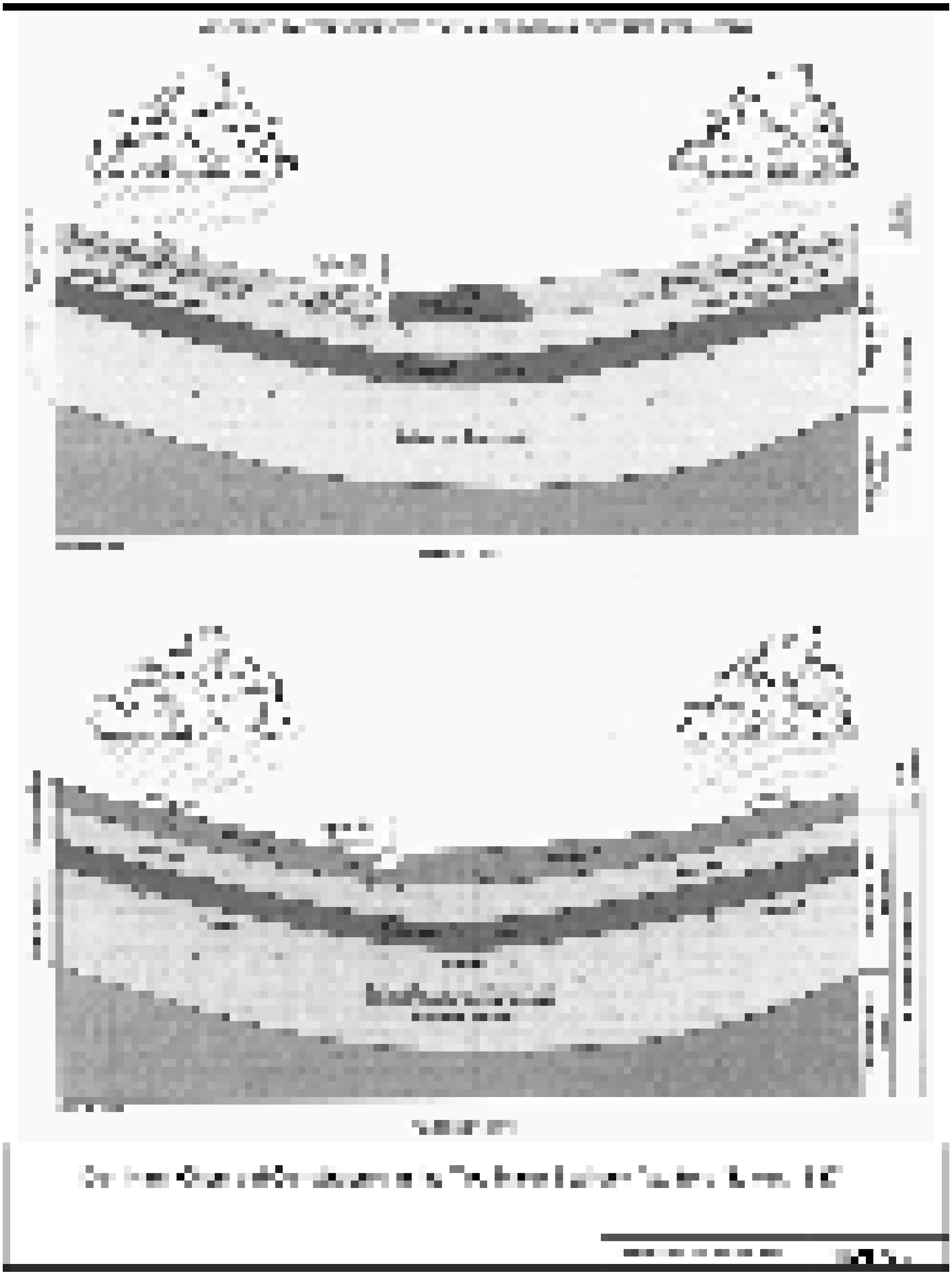












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Attachment 4
The Uinta Basin

The Uinta Coal Basin is located mostly within eastern Utah; a very small portion of the basin is in northwestern Colorado (Figure A4-1). The basin covers approximately 14,450 square miles (Quarterly Review, 1993) and is structurally separated from the Piceance Basin by the Douglas Creek Arch (Figure A4-1), an up-warp near the Utah – Colorado state line. Coalbeds are present within Cretaceous strata throughout much of the Uinta Basin. However, coalbed methane exploration, to date, has targeted coalbeds in the Ferron Sandstone Member of the Mancos Shale and coalbeds in the Blackhawk Formation of the Mesaverde Group. The total, in-place, coalbed gas resources in the Wasatch Plateau, Emory, Book Cliffs and Sago coal fields have been estimated at 8 trillion cubic feet (Tcf) to more than 10 Tcf by the Utah Geological Survey (Gloyn and Sommer, 1993). This estimate is based on extrapolation of known coal resources to a depth of 9,000 feet and an average projected gas content of 330 cubic feet per ton and does not include the Tabby Mountain or Vernal coalfields, or the Sevier-Sanpete coal region. Total production stood at 75.7 billion cubic feet (Bcf) of coalbed methane in 2000 (GTI, 2002).

4.1 Basin Geology

Much of the Rocky Mountain region, including the Uinta Basin was covered by an epicontinental sea. Deposition in the sea lasted from the Albian (about 100 million years ago) through the Cenonmanian (about 83 million years ago), with the deposition of the upper part of the Mesaverde Group generally marking the end of marine deposition in the basin (Howells et al., 1987).

The Uinta Basin formed as a result of uplift and deformation that began in the Late Cretaceous. The Cretaceous sediments outcrop along the perimeter of the basin. The basin is asymmetrical in shape with strata on the northern flank of the basin dipping steeply toward the basin axis, while strata on the southern flank dip gently toward the basin axis. The stratigraphic units of the coal bearing Cretaceous rocks of the Uinta Basin are shown in Figure A4-2.

Two Cretaceous stratigraphic units have been targeted for coalbed methane exploration: the Ferron Sandstone Member of the Mancos Shale and the Blackhawk Formation of the Mesaverde Group (Figure A4-2). The Ferron Sandstone Member was deposited in the Last Chance delta, a fluvial-deltaic environment (Garrison et al., 1997). The coalbeds and interbedded sandstone units form a wedge of clastic sediment 150 to 750 feet thick stratigraphically above the Tunuck Shale Member of the Mancos Shale and below the Lower Blue Gate Shale Member of the Mancos Shale (Figure A4-2). Both of these shale units have a very low permeability and constitute confining units for water and gas in the Ferron Sandstone Member. The coal-bearing rocks are thickest to the west and south margins of the basin, nearer to the upland sources of sediment. Coalmines producing from the Ferron Sandstone Member are located along the eastern boundary of the Wasatch Plateau south of Castle Dale, Utah (Figure A4-1). Depths to coal in the Ferron Sandstone Member range from 1,000 to over 7,000 feet (Garrison et al., 1997). Primary coalbed methane activity from the Ferron Sandstone takes place in the Drunkard's Wash Unit. Total coal thickness in this area ranges from 4 to 48 feet (averaging 24 feet) from depths of 1,200 to 3,400 feet (Lamarre and Burns, 1996).

The Blackhawk Formation consists of coal interbedded with sandstone and a combination of shale and siltstone. The Blackhawk Formation is underlain by the Star Point Sandstone and overlain by the Castlegate Sandstone (Figure A4-2). The Castlegate Project in the Book Cliffs coalfield initially targeted coals in the Blackhawk Formation at depths ranging from 4,200 to 4,400 feet (Gloyn and Sommer, 1993).

4.2 Basin Hydrology and USDW Identification

Groundwater hydrology of the Uinta Basin is controlled primarily by the geologic structure of the region (Howells et al., 1987). Variations of aquifer and aquitard permeability owing to differences of lithology and facies changes also play an important role in the hydrology, as does widespread faulting and fracturing of the rocks (Howells et al., 1987). Because of the basin's structure, the area may be a groundwater basin with internal drainage. If there were a deep groundwater outlet for the basin, it would be along or near the axis of the Uinta Basin at its western edge. The general pattern of groundwater flow is centripetal, with water flowing inward from recharge areas at exposures of permeable strata at the margins of the basin. Recharge is greatest near the northern edge of the basin. Other recharge areas include Eocene and Oligocene Formations in the basin interior.

Most of the sandstone formations in the Mesozoic rocks in the Upper Colorado River Basin are identified as aquifers by the United States Geological Survey (Freethey and Cordy, 1991). Freethey and Cordy stated that in the Uinta Basin, the older and deeper aquifers in strata below the Ferron Sandstone Member, (for example, the Navajo-Nugget Aquifer, Entrada-Preuss Aquifer, Morrison Aquifer, and the Dakota Aquifer) generally contain very saline to briny water, with total dissolved solids (TDS) values greater than 10,000 milligrams per liter (mg/L). The water quality component of the underground source of drinking water (USDW) definition specifies that a USDW contain less than 10,000 mg/L of TDS. The Ferron Sandstone Member (Figure A4-3) is designated as a producing aquifer in east-central Utah (Freethey and Cordy, 1991). In regard to the Mesaverde Group Aquifer, which includes the Star Point Sandstone, the Blackhawk Formation, the Castlegate Sandstone and the Price River Formation, (Figure A4-3) Freethey and Cordy (1991), stated that, "water in these aquifers is more likely to be developed where the saturated thickness is large and the depth to the aquifer is less than 2,000 ft." They further stated that the margins of the Uinta Basin where these rocks are near the surface or outcrop is a possible location for development of groundwater with low enough TDS to be used for drinking water.

Wells in the Ferron Sandstone Member at the Drunkard's Wash coalbed methane field typically penetrate to depths ranging from 1,200 to 3,400 feet (Lamarre and Burns, 1996). An average water quality value of 13,120 mg/L TDS (Gwynn, 1998) for production waters that have been retained in catchment ponds suggests that these wells are not within a USDW. Gwynn (1998) however, does state that due to the ponding of the produced water in evaporation lagoons, the concentration of salts in these waters has probably increased from their original levels. This implies that these water quality data may not be useful in the confirmation of USDW qualifications. Quarterly Review (1993) reported that three wells producing gas and water from the Ferron Sandstone Member coalbeds in the Drunkard's Wash field yielded over 49,000 gallons of water per day with a TDS level of about 5,000 mg/L (sodium bicarbonate) during the first 2 to 3 months of operation. The Ferron Sandstone is hydrologically confined above and below by shale members of the Mancos Shale formation. Water produced from the Ferron Sandstone is thought to be connate water that was trapped in the sediment during coalification (Gloyn and Sommer, 1993). Hunt (Utah Division of Oil, Gas, and Mining, 2001) noted that there were no USDWs located immediately above the Ferron Sandstone Member due to the thick tongues of Mancos Shale that encapsulate the coal-bearing interval (Figure A4-2).

Beds targeted for methane gas exploration and production within the Blackhawk Formation are approximately 4,200 to 4,400 feet below the ground surface (Gloyn and Sommer, 1993). Coalbed gas production in the Castlegate Field accounted for less than 10 percent of the coalbed methane production in the Uinta Basin (Petzet, 1996). The average gas well producing from the coalbeds in the Blackhawk Formation (Castlegate field) yielded 318 barrels of water per day, and TDS levels of 5,489 mg/L have been measured in the produced waters (Gloyn and Sommer, 1993).

According to the State of Utah Department of Natural Resources (DNR), Division of Oil, Gas and Mining, the water quality in the Ferron and Blackhawk varies greatly with location, each having some TDS levels below and some above 10,000 mg/L (Utah DNR, 2002). In general, the quality of Blackhawk water is higher than that of Ferron water. The most recent Underground Injection Control application received for the Drunkard's Wash field (Ferron) showed a composite quality of input water to be about 31,000 mg/L TDS, and for the Castlegate field (Blackhawk) 9,286 mg/L TDS. At some locations, either formation member would not qualify as a USDW.

In the western part of the Uinta Basin, the Castlegate Sandstone, an aquifer, is separated from the Black Hawk Formation coalbeds by approximately 300 feet of alternating shale and sandstone (Utah DNR 2002). The Star Point Sandstone is located below approximately 400 feet of alternating sandstone and shale that underlies the bottom coal of the Black Hawk Formation. In some areas, the shale and sandstone underlying the Black Hawk coals are highly faulted. There is some potential that hydraulic fracturing fluids could be transported through natural fracture networks in these areas and reach the Star Point Sandstone. The relatively impermeable upper Blue Gate Shale Member of the Mancos Shale Further would prevent further downward migration.

In reference to the quality of water produced by the coalbed gas wells in both the Ferron Sandstone Member of the Mancos Shale and the Blackhawk Formation, Quarterly Review (1993) states: "Disposal of produced water does not appear to present a major environmental problem in the Uinta basin, unlike the San Juan and some other western basins. Rates are moderate, 200 to 300 barrels per day per well during early stages of production and TDS levels are not high (about 5,000 mg/L)." Because these TDS values are less than the 10,000 mg/L limit, both the Ferron Sandstone Member of the Mancos Shale and the Blackhawk Formation may qualify as USDWs.

Tabet (2001) suggests that coalbed methane extraction wells are not located in "producing" aquifers and that most of the potable water in the sparsely populated area is supplied by surface water and shallow alluvial aquifers.

4.3 Coalbed Methane Production Activity

Full-scale exploration in the Uinta Basin began in the 1990s (Quarterly Review, 1993). The most active operators at that time were PG&E Resources Company, the River Gas Corporation, Cockrell Oil Corporation, and Anadarko Petroleum Corporation. PG&E acquired the Castlegate Field, from Cockrell Oil (Gloyn and Sommer, 1993). Gas was produced from coalbeds in the Blackhawk Formation. The five wells initially drilled in the Castlegate Field were hydraulically fractured with 80,000 to 143,000 pounds of sand and unreported volumes of fluid. Other wells were to be fractured with a low-residue gel system to ensure breakdown within the reservoir (Quarterly Review, 1993).

The Castlegate field was off-line due to production water disposal problems (Tabet, 2001; and Hunt, Utah Division of Oil, Gas, and Mining, 2001). According, to the State of Utah DNR, Division of Oil, Gas and Mining, the field is now on production (Utah DNR, 2002).

The River Gas Corporation operates the Drunkard's Wash Unit, producing methane gas from coals within the Ferron Sandstone Member. The company reported that high fracture gradients hampered hydraulic fracturing stimulations using cross-linked borate gel with 250,000 pounds of proppant (Quarterly Review, 1993). Excessive proppant flowback resulted in one well where nitrogen foam was used for the fracturing. The Buzzard Bench Field, also producing gas from the Ferron Sandstone Member, was initially operated by Chandler & Associates, Inc. (Petzet, 1996) and is currently being managed by Texaco (Garrison et al., 1997).

A query of a database covering the Uinta Basin revealed that there are about 1,255 coalbed methane wells in production in the basin (Osborne, 2002). Gas Technology Institute (GTI) places the annual coalbed methane production in the Uinta Basin at 75.7 Bcf in 2000 (GTI, 2002).

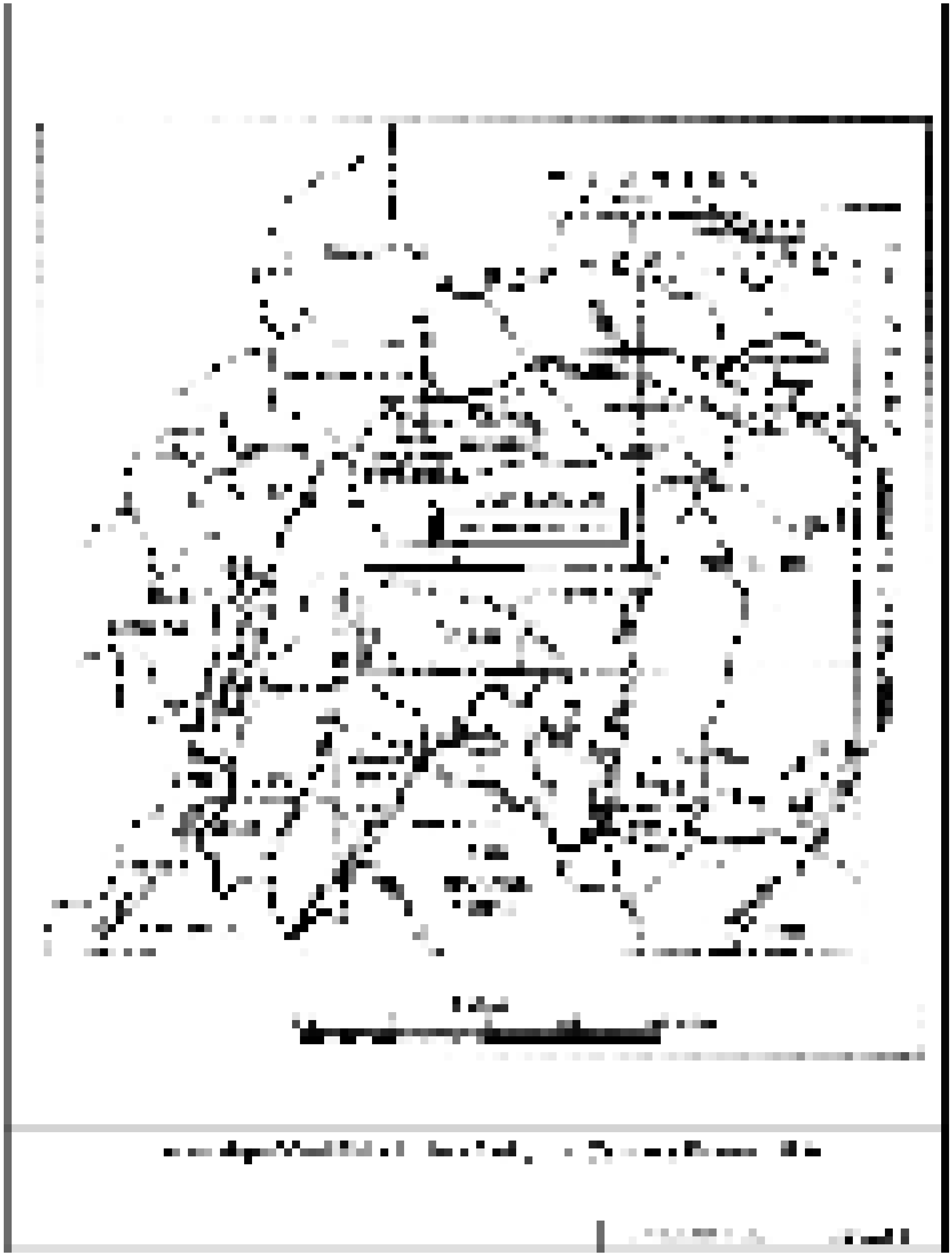
4.4 Summary

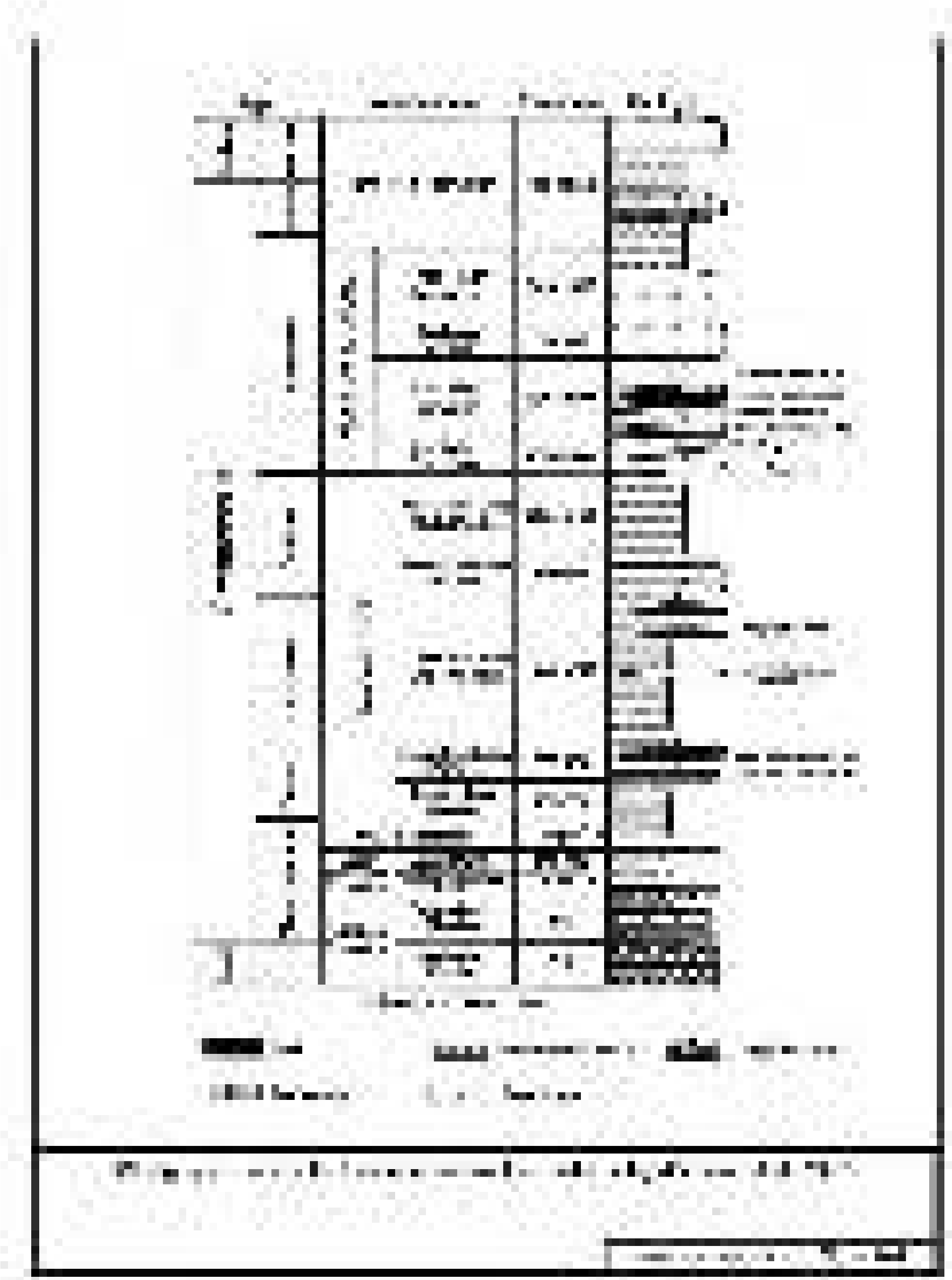
Waters from coalbed methane production in the Ferron Sandstone Member of the Mancos Shale in the Drunkard's Wash Unit are conflictingly reported to have TDS values of about 13,000 mg/L according to one source of information or to have levels of TDS of about 5,000 mg/L from another. However, the higher values were derived from water samples taken from evaporation lagoons and these high values might represent elevated concentrations of salts owing to evaporation. Consequently, if the more moderate TDS levels were correct, then the Ferron Sandstone would qualify as a USDW.

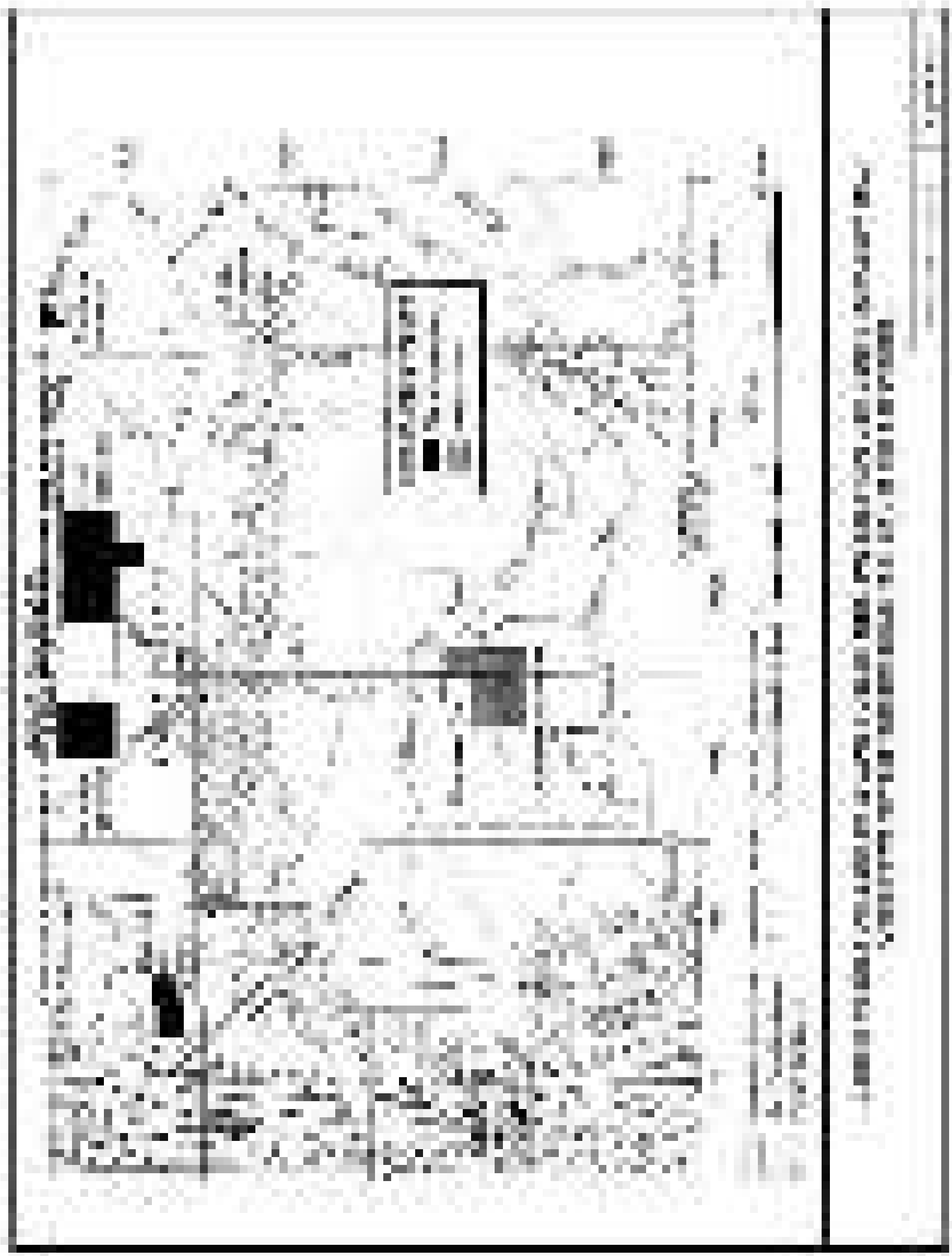
According to the State of Utah DNR, Division of Oil, Gas and Mining, the water quality in the Ferron and Blackhawk varies greatly with location, each having TDS levels below and above 10,000 mg/L (Utah DNR, 2002). In general, the quality of Blackhawk water is fresher than Ferron water. The most recent Underground Injection Control application received for the Drunkard's Wash field (Ferron) showed a composite quality of input water to be about 31,000 mg/L TDS, and for the Castlegate field (Blackhawk) 9,286 mg/L TDS. At some locations, neither formation member would qualify as a USDW.

The Drunkard's Wash and Castlegate coalbed methane extraction fields are located in a sparsely populated section of Utah. Tabet (Utah Geological Survey, 2001) suggests that coalbed gas extraction wells are not located in "producing" aquifers and that most of the potable water in the sparsely populated area is supplied by surface water and shallow alluvial aquifers.

The Blackhawk Formation is underlain by 300 feet of shale and sandstone that separate it from the Castlegate Sandstone aquifer. It is underlain by similar geologic strata, which separate it for the Star Point Sandstone. Only in highly faulted areas is there a reasonable possibility that hydraulic fracturing fluids could migrate down to the Star Point Sandstone.







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Attachment 5
The Powder River Basin

The Powder River Basin is located in northeastern Wyoming and southern Montana. The basin covers an area of approximately 25,800 square miles (Larsen, 1989), approximately 75 percent of which is within Wyoming (Figure A5-1). Fifty percent of the basin (Figure A5-2) is believed to have the potential for production of coalbed methane (Powder River Coalbed Methane Information Council, 2000). Much of the coalbed methane-related activity has been north and south of Gillette in northeastern Wyoming (Figure A5-2). The majority of the potentially productive coal zones range from about 450 feet to over 6,500 feet below ground surface (Montgomery, 1999). In addition to being an important resource for coalbed methane, the basin has also produced coal, petroleum, conventional natural gas, and uranium oxide (Law et al., 1991; Randall, 1991). Recent estimates of coalbed methane reserves in the Powder River Basin have been as much as 40 trillion cubic feet (Tcf) (PRCMIC, 2000) but more conservative estimates range from 7 to 12 Tcf (Montgomery, 1999). Annual production volume was estimated at 147 billion cubic feet (Bcf) in 2000 (GTI, 2002). In 2002, wells in the Powder River Basin produced about 823 million cubic feet (Mcf) per day of coalbed methane (DOE, 2002).

The information available indicates that hydraulic fracturing currently is not widely used in this region due to concerns about the potential for increased groundwater flow into the coalbed methane production wells and collapse of open hole wells in coal upon dewatering. According to the available literature, where hydraulic fracturing has been used in this basin, it has not been an effective method for extracting methane.

5.1 Basin Geology

The Powder River Basin is a thick sequence of sedimentary rock formed in a large downwarp within the Precambrian basement. The basin is bounded on the east by the Black Hills uplift, on the west by the Big Horn uplift and Casper Arch, on the south by the Laramie and Hartville uplifts and, on the north, it is separated from the Williston Basin by the Miles City Arch and the Cedar Creek Anticline (Larsen, 1989) (Figure A5-1). The long axis of the basin is aligned in a generally southeast to northwest direction, and it is as much as 18,000 feet deep (Randall, 1991) (Figures A5-1 and A5-3). Sediments range from Paleozoic at the bottom through Mesozoic to Tertiary at the top (DeBruin et al., 2000). The basin is a large asymmetrical syncline with its axis (deepest part) near the west side of the basin (Figure A5-3). From outcrops along the eastern edge of the basin, the sediments slope gently (1.5^{o}, about 100 feet per mile) downward to the southwest and then bend steeply upward (10 to 45^{o}) to outcrop in a monocline along the western edge of the basin.

Several periods of deposition by marine and fluvial-deltaic processes have occurred within the basin during the Cretaceous and Tertiary periods. These Cretaceous and lower

Tertiary rocks have a total thickness of up to 15,000 feet (Montgomery, 1999). Coal is found in the Paleocene Fort Union and Eocene Wasatch Formations (Figure A5-4). The Wasatch Formation occurs at land surface in the central part of the basin and is covered by alluvium or White River Formation in some places (Figure A5-4). Most of the coalbeds in the Wasatch Formation are continuous and thin (six feet or less) although, locally, thicker deposits have been found (DeBruin et al., 2000). The Fort Union Formation lies directly below the Wasatch Formation and can be as much as 6,200 feet thick (Law et al. 1991). The Fort Union Formation outcrops at the ground surface on the eastern side of the basin, east of the City of Gillette and on the western side of the basin, north and south of Buffalo. The coalbeds in this formation are typically most abundant in the upper Tongue River Member (Figure A5-4). This member is typically 1,500 to 1,800 feet thick, of which up to a composite total of 350 feet of coal can be found in various beds. The thickest of the individual coalbeds is over 200 feet (Flores and Bader, 1999). The coalbeds are interspersed with sandstone, conglomerate, siltstone, mudstone and limestone (Montgomery, 1999).

Most coalbed methane wells in the Powder River Basin are in the Tongue River Member of the Fort Union Formation, in the Wyodak-Anderson coal zone, which contains up to 32 different coalbeds according to some authors (Ayers, 1986), including the Big George in the central part of the basin (Flores and Bader, 1999). The Wyodak is one of the thick coalbeds that are targeted for coalbed methane development. This coalbed is also called the Wyodak-Anderson or the Anderson, and it can be subdivided further into several other coalbeds. These coalbeds are the Canyon, Monarch, and Cook. All of these coalbeds are coalbed methane targets. Most coalbeds are found within 2,500 feet of the ground surface.

The Wyodak or Wyodak-Anderson coalbed in the Wyodak-Anderson coal zone is prominent in the eastern portion of the Powder River Basin near the City of Gillette (Figures A5-3, A5-5 and A5-6). The Wyodak has been identified as the largest single coalbed in the country (Montgomery, 1999). The coal is close to the ground surface and mining of the coal is common. The Wyodak coalbed gets progressively deeper and thicker toward the west. This bed ranges from 42 to 184 feet thick. Most of the coalbed methane wells in the Powder River Basin are within the Wyodak coal zone near the City of Gillette.

The Big George Coalbed is located in the central and western portion of the Powder River Basin (Figure A5-7). Although the Big George is stratigraphically higher than the Wyodak, owing to the structure of the basin, the Big George, in the center portion of the basin, is deeper than the Wyodak at the eastern margin of the basin (Tyler, et al., 1995). To date, the Big George has not been developed for coalbed methane production to the same extent as the Wyodak-Anderson coal zone. This is due to a combination of factors including greater depth to coal, more groundwater, and longer distances to available transmission pipelines. However, as of December 2001, there were about 850 coalbed methane wells drilled into the Big George with a large number of wells planned for the future (Osborne, 2002).

A third significant coal zone, the Lake De Smet coal zone in the Wasatch Formation, is up to 200 feet thick and is located in the Lake De Smet area (Figure A5-8), 55 miles southwest of Recluse on the western side of the basin (Larsen, 1989). It has not yet been widely used for coalbed methane production.

Most of the coal in the Powder River Basin is subbituminous in rank, which is indicative of a low level of maturity. Some lignite, lower in rank, has also been identified. The thermal content of the coals found in the Powder River Basin is typically 8,300 British thermal units per pound (Randall, 1991). Coal in the Powder River Basin was formed at relatively shallow depths and relatively low temperatures. Most of the methane generated under these conditions is biogenic, which means that it was formed by bacterial decomposition of organic matter. Thermogenic formation (formed under high temperature) was not significant in most locations within the Powder River Basin. Consequently, coal in the Powder River Basin contains less methane per unit volume than many other coal deposits in other parts of the country. Coal in the Powder River Basin has been found to contain 30 to 40 standard cubic feet of methane per ton of coal compared to 350 standard cubic feet of methane per ton in other areas (DeBruin et al., 2000). The gas is typically more than 95 percent methane, the remainder being mostly nitrogen and carbon dioxide. This resource was overlooked for many years because it was thought to be too shallow for the production of significant amounts of methane (Petzet, 1997). However, the relatively low gas content of Powder River Basin coal is compensated by the thickness of the coal deposits. Because of the thickness of the deposits and their accessibility, commercial development of the coalbed methane has been found to be economical.

The Powder River Basin contains approximately 60 percent of the coalbed methane reserves in the State of Wyoming (DeBruin et al., 2000). Recent estimates of coalbed methane reserves in the Powder River Basin have been as much as 40 Tcf (PRCMIC, 2000) but more conservative estimates range from 7 to 12 Tcf (Montgomery, 1999). As of December 1999, monthly production exceeded 7 Bcf from 1,657 wells (DeBruin et al., 2000). Wells typically produce 160,000 cubic feet of gas per day (DeBruin et al., 2000). Annual production volume was estimated at 147 Bcf in 2000 (GTI, 2002). In 2002, wells in the Powder River Basin produced about 823 Mcf per day of coalbed methane (DOE, 2002). Coalbed methane has been developed along both the east and west flanks of the basin where the coalbeds are buried but relatively shallow. Many existing wells are awaiting connection to the distribution system and still more wells are being drilled. The estimated lifetime production from these wells is 300 to 400 Mcf per well (Petzet, 1997).

The amount of coalbed methane produced from each well is highly variable, and the volume of gas depends on the quality and thickness of the coal, the frequency of natural cleats in the coal, and the amount of water present. Other factors, such as well completion techniques and well stimulation techniques, also control the amount of gas produced from a well. Maximum coalbed methane flow from a well is typically achieved after one to six months of dewatering (Montgomery, 1999). Stable production is usually experienced for one to two years before production begins to decline (Montgomery, 1999). Production often declines at a rate of 20 percent per year until the well is no longer economically useful (Montgomery, 1999). Several options exist at that point, including re-fracturing the well, completing the well in a deeper coal formation, converting the well to a water supply well, or abandoning the well.

5.2 Basin Hydrology and USDW Identification

A report prepared by the United States Geological Survey (USGS) showed that samples of water co-produced from 47 coalbed methane wells in the Powder River Basin all had total dissolved solids (TDS) levels of less than 10,000 milligrams per liter (mg/L) (Rice et al., 2000). Based on the water quality component of the underground source of drinking water (USDW) definition, which specifies that a USDW contain less than 10,000 mg/L of TDS, the Fort Union Formation coalbeds are within a USDW. The water produced by coalbed methane wells in the Powder River Coal Field commonly meets drinking water standards, and production waters such as these have been proposed as a separate or supplemental source for municipal drinking water in some areas (DeBruin et al., 2000). Sandstones in the sediments both above and below the coalbeds are also aquifers.

In 1990, Wyoming withdrew an average of 384 million gallons per day of groundwater for a variety of purposes, the majority of which was agriculture. Approximately 13 percent was used for potable water supplies. Approximately 22 percent was withdrawn by industry and mining (Brooks, 2001). The proportion of this 22 percent attributable to coalbed methane production is increasing rapidly, and a concern exists that such good quality water in a semiarid region should be conserved (Quarterly Review, 1993). In 1990, before the rapid expansion of coalbed methane extraction in the region, Campbell County was identified by the USGS as an area of major groundwater withdrawal.

Approximately 80 percent of Wyoming residents rely on groundwater as their drinking water source (Powder River Basin Resource Council, 2001). Few public water supply systems exist in the Powder River Basin due to relatively low population densities. The City of Gillette, the largest in the major coalbed methane development area (Figure A5-2), uses groundwater from two sources identified as “in-town wells”, and the “Madison Well Field”. The city has experienced considerable drawdown and reduced production from their in-town wells that are completed in the Fort Union and Lance/Fox Hills aquifers (Brooks, 2001). It is unclear how much of the drawdown is attributable to withdrawals for water supply as a consequence of population growth and how much is attributable to nearby coalbed methane production. Between 1995 and 1998, the city restored and/or replaced several of its wells. The Madison Well Field produces water from the Madison Formation and is approximately 60 miles east of the city. There are no coalbed methane wells in the vicinity of the Madison Well Field (Brooks, 2001).

Regional groundwater flow in the basin is reported to be toward the northwest (Martin et al., 1988 in Law, 1991), with recharge occurring in the east along the Rochelle Hills.

Cleats and other fractures within the coalbeds create high hydraulic conductivities and facilitate the flow of groundwater and high water production within the coalbeds (Montgomery, 1999). The coalbeds are largely hydraulically confined by underlying shale and by basinward pinch-out. Surficial water and rainwater can enter the Fort Union coals from land surface at the eastern edge of the basin and at the Black Hills uplift. This flow inward from outcrop areas at higher elevations on the edge of the basin may have created artesian conditions in the deeper central portions of the basin. However, this view may not be entirely correct. For example, coalbed research (Law et al., 1991) hypothesizes that the sodium bicarbonate water in the Fort Union coal near the central part of the basin may not be derived from meteoric recharge, but rather from interstitial waters of the original peat deposits. Furthermore, Martin et al. (1988, as cited in Law et al., 1991) concluded on the basis of isotopic composition of water samples that only part of the water near outcrops was of meteoric origin. Although artesian pressure in the center of the basin has been thought to be evidence that the center of the basin is fed from meteoric recharge at the basin margins, the apparent artesian pressure (flowing wells) could be explained by the airlift effect of methane coming out of solution within the rising well water column.

Because the coalbeds are productive aquifers, they also require more dewatering of coalbed methane wells for methane production. Groundwater production, in terms of volume of water produced, was a major factor considered in the selection of sites for early coalbed methane wells and may still guide development of sites in some parts of the Powder River Basin. Wells in the eastern portion of the basin have been found to contain less water due to their location above the water table within the eastern anticlinal updip of the formation and, in some areas, due to the presence of nearby mines that dewater the aquifer. Drawdowns of up to 80 feet have been measured in wells near active mines; however, water levels have been reported to be unaffected at distances of more than three miles from mines (Randall, 1991). The Bureau of Land Management in conjunction with the State Engineer’s Office has been conducting ongoing research on the effects of coalbed methane production on drawdown (Wyoming Geological Association, 1999).

5.3 Coalbed Methane Production Activity

Coalbed methane activity in Wyoming occurs predominantly in Campbell, Sheridan and Johnson Counties (DeBruin, 2001). Wells are spaced from 40 to 80 acres per well, as determined by the State. Permits are required under both state water well regulations and state gas well regulations before drilling can commence. A discharge permit from the Wyoming Department of Environmental Quality is also required for the water that is removed from the well. Coalbed methane production wells in the Powder River Basin are typically 400 to 1,500 feet deep and can be as shallow as 150 feet (PRCMIC, 2000). By comparison, conventional gas and oil wells installed in the area are typically 4,000 to 12,000 feet deep (PRCMIC, 2000). Plans for construction of approximately 4,000 new coalbed methane production wells in the Montana portion of the Powder River Basin await completion of an in-depth environmental study (DeBruin, 2001).

Commercial development of methane directly from the coal seams began approximately in 1986. There were only 18 wells producing coalbed methane in the Powder River Basin by 1989. The number grew slowly through the early 1990s with 171 wells producing approximately 8 Bcf of gas per year. The rate of development of the resource accelerated greatly from 1997 to 1999. In 1999, there were 1,657 coalbed methane wells operating in the Powder River Basin, producing approximately 58 Bcf per year (Figure A5-9) of coalbed methane. As of November 2000, there were about 4,270 wells in Wyoming producing 15 Bcf of coalbed methane in that month alone (Osborne, 2002). By November 2001, monthly coalbed methane production had climbed to 23.5 Bcf from 7,870 producing wells in Wyoming (Osborne, 2002). In Montana, 246 active wells produced 872,008 Mcf of coalbed methane in December, 2001 (Osborne, 2002). The Powder River Basin has become the most active coalbed methane exploration and production area in the country (DeBruin et al., 2000). Despite all of the activity, less than 5 percent of the land underlain by coal in the Powder River Basin had been explored for the presence of coalbed methane as of the year 2000 (PRCMIC, 2000).

During the early years of coalbed methane development in the Powder River Basin (1980s to early 1990s), gas exploration and development companies completed wells with and without hydraulic fracture techniques. Larsen (1989) indicated that early wells were completed without fracturing treatments, particularly wells targeting gas reserves in coals interspersed between sandstone layers. However, the Quarterly Review (1989) reported that in one well, Rawhide 15-17, located north of Gillette, Wyoming, an “open frac” hydraulic fracturing was performed using 13,000 lbs of 12/20-mesh sand in 3,500 gallons of gelled water. Several wells installed in the early 1990s by Betop, Inc. were fractured using 4,000 to 15,000 gallons of a solution with 2 percent potassium chloride (KCl) in water. Sand was used to prop the fractures open in five of these wells (Quarterly Review, 1993). However, hydraulic fracturing experienced little success in this basin. Fractured wells produced poorly because the permeable, shallow subbituminous coals collapsed under the pressure of the overburden after they were dewatered (Lyman, 2001).

The Powder River Basin contains coals of high permeability. The permeability is so high in many areas that drilling fluid (typically water) is lost when drilling the coalbeds. Many times drilling mud is substituted to prevent loss of circulation (DeBruin, 2001). Because of this high permeability, most coalbed wells in the Fort Union Formation can be drilled and completed without the use of hydraulic fracturing (DeBruin, 2001; Quarterly Review, 1993). This has been confirmed by USGS officials in Wyoming (Brooks, 2001). Hydraulic fracturing is also avoided to prevent fracturing of impermeable formations adjacent to the coal, such as shales, that prevent the migration of groundwater. It is thought that fracturing the shale would increase the amount of water flowing into the wells. When fracturing has been done, it has been with water or sand/water mixtures. Unspecified “modest” improvements in coalbed methane gas flow have been observed (Quarterly Review, 1993).

In the Powder River Basin, two different coalbed methane sources are commonly developed: (1) gas extraction from methane-charged dry sand layers overlying or interbedded with the coals, and (2) conventional methane extraction from the water saturated coal seams. In the eastern (up dip) portion of the basin, the coals in the Wyodak-Anderson seam are relatively shallow and interbedded with sands (Montgomery, 1999) (Figure A5-6). In up dip areas above the water table, wells require minimal dewatering for coalbed methane production because there is little to no water in the sands (Quarterly Review, 1989; Montgomery, 1999). Coal mining operations near Gillette have lowered the water table in the vicinity of the mines, thereby dewatering nearby coalbeds and allowing desorption of methane gas from the coal. The sands are penetrated using open-hole techniques, generally without any fracture treatments (Quarterly Review, 1989). Further west, down dip (Figure A5-6), the coalbed methane producing sands and coals of the Fort Union Formation are separated from the overlying Wasatch Formation by a poorly permeable shale of limited areal extent (Quarterly Review, 1989; Quarterly Review, 1993). Further west, down dip (Figure A5-6) in this more water-saturated part of the basin, coalbed methane wells are also completed as open-hole wells.

The practice of open-hole drilling is commonly used in this region. In this practice, a portion of the borehole in the coal is drilled without any casing or well screen. Most other regions of the country where coalbed methane is recovered use a perforated casing throughout the target coal interval. The open coal zone is then cleaned out with water, and the surrounding coal formation is sometimes fractured to improve recovery of the methane. A submersible pump is set at the bottom of the target zone with tubing to the ground surface to remove groundwater from the well. The methane gas travels up the space between the water tubing and the casing. The well is capped to control the flow of methane gas. Wells are often dewatered for several months before producing optimal quantities of methane gas.

Side jetting has also been performed with some success; however, dynamic open-hole cavitation had not been attempted as of 1993. Side jetting is the process by which water and air are injected at high pressure to enlarge the boring in the coal seam. The cavitation process uses dynamic pressure changes to break apart the coal and to widen the boring within in the coal seam (Quarterly Review, 1993).

Production of coalbed methane from water-saturated coalbeds below the water table first requires partial dewatering of the coal to allow desorption of methane from the coal. Production from water-bearing coal seams can yield significant volumes of water; enough to make it difficult or infeasible to dewater the formation sufficiently to initiate coalbed methane flow (Montgomery, 1999). Tests on 11 wells reported by Crockett (2000) indicate that coalbed methane is desorbed from coal as a consequence of decreased hydrostatic pressure caused by pumping groundwater. One well started desorbing at 92 percent of the original reservoir pressure. “Most drilling to date has attempted to remain near or above the existing water table to minimize water production” (Montgomery, 1999). Modifications to well spacing and pumping configuration have been cited by Montgomery (1999) as showing some promise for allowing greater production from the water-saturated coal seams in the future. Because the water in the deeper coal seams may be original interstitial water, and recharge from meteoric water might not be an important factor (Montgomery, 1999), dewatering of these coals for the purposes of coalbed methane production might become economically feasible.

Disposal of water produced by coalbed methane wells is an issue at many well locations. Coalbed methane wells are generally pumped constantly, removing as much as 168,000 gallons per day of water from deeper formations (Randall, 1991). Averages of 17,000 gallons per day per well are more common (Powder River Basin Resource Council, 2001). Water produced during the dewatering of coalbed wells is generally discharged to stock ponds, water impoundments (reservoirs), drainages with ephemeral and intermittent streams, and surface waters. A National Pollution Discharge Elimination System permit is required for surface discharge of production water. The water is generally of potable quality in the center of the basin, becoming more saline to the north and south. It is sometimes used for irrigation and watering livestock (DeBruin, 2001). TDS levels are typically less than 5,000 parts per million. The water's salt content is primarily sodium bicarbonate (Quarterly Review, 1993). Average analytical results from 47 USGS water quality analyses of untreated, co-produced water from coalbed methane wells in the Powder River Basin are displayed in Table A5-1 below.

Table A5-1. Average Water Quality Results from Produced Waters (Rice et al., 2000)

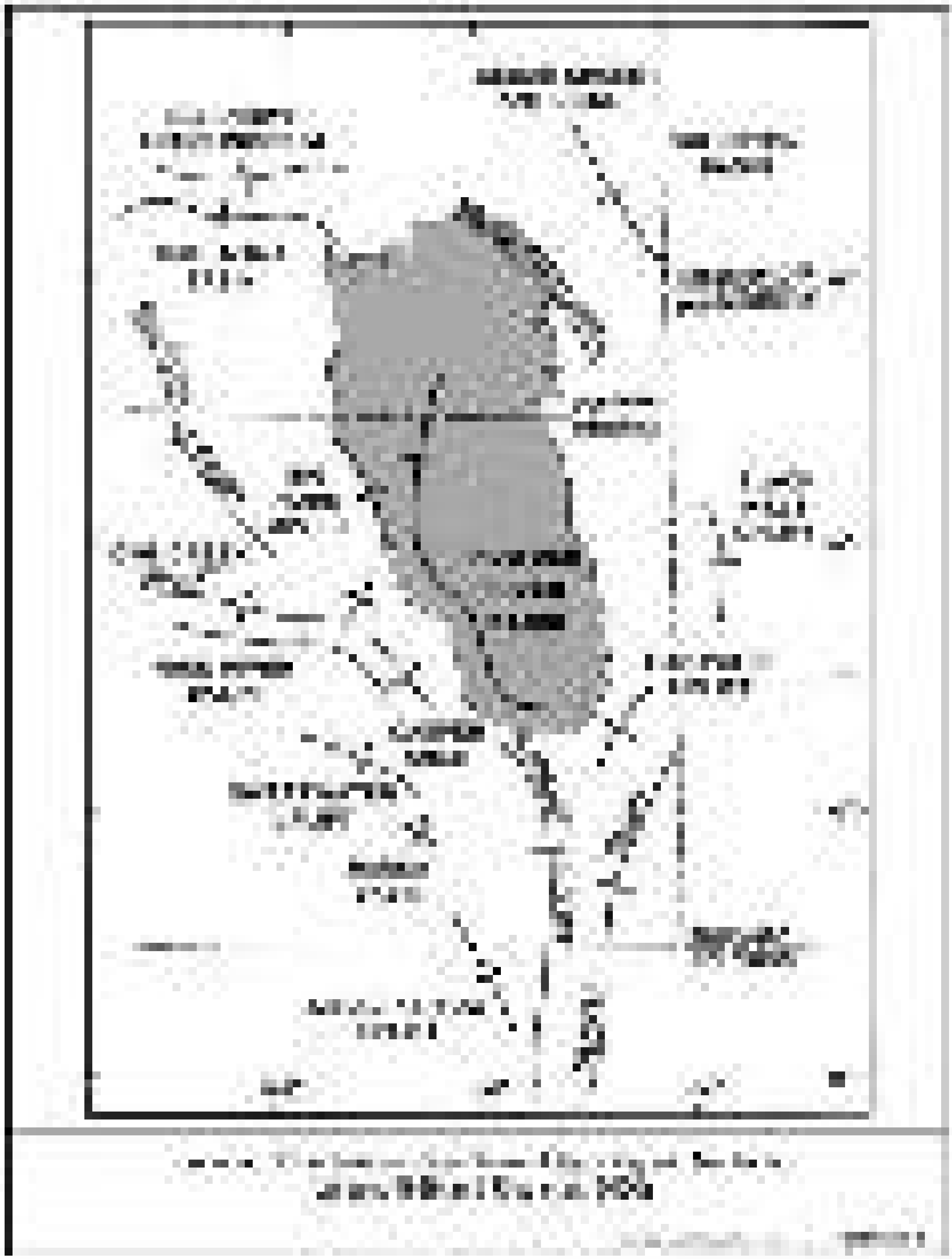
Parameter	Result	Units
pH	7.3	N/A
temperature	19.6	^{o}C
specific conductance	1,300	microsiemens
TDS	850	mg/L
fluoride	0.92	mg/L
chloride	13.0	mg/L
sulfate	2.4	mg/L
bromide	0.12	mg/L
alkalinity (as HCO_3)	950	mg/L
ammonium	2.4	mg/L
calcium	32	mg/L
potassium	8.4	mg/L
magnesium	16	mg/L
sodium	300	mg/L
barium	0.62	mg/L
iron	0.8	mg/L

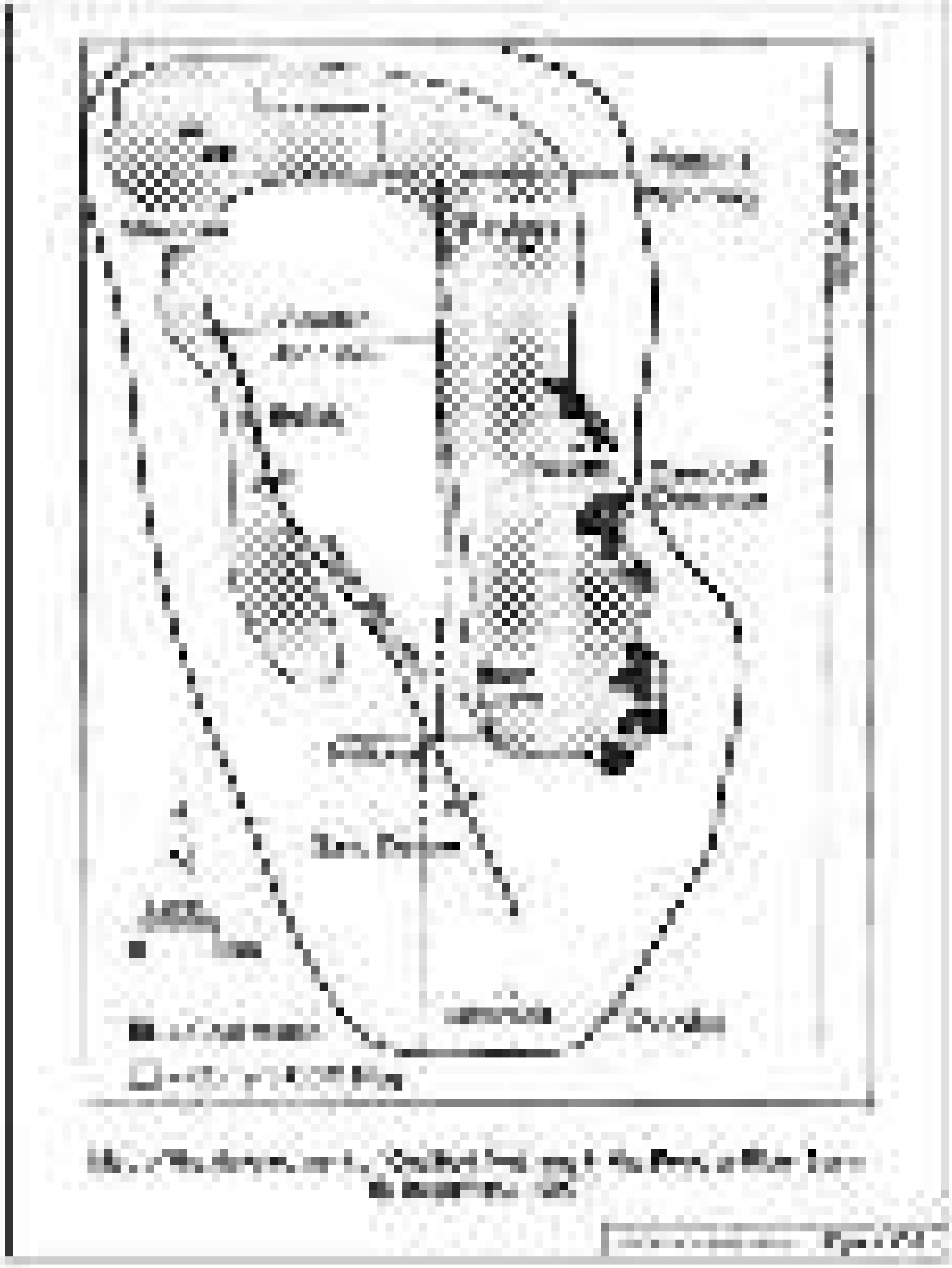
As a result of the rapid growth in the coalbed methane industry, the Wyoming State Engineer's Office (SEO) requested funding for drilling, equipping, and monitoring of observation wells, and the installation of surface water measuring devices to be located in coalbed methane production areas. These monitoring facilities would become part of the SEO statewide observation well network to monitor changes in groundwater levels and stream flow over time. As of 1999, work was underway, but no report of results had yet been made available.

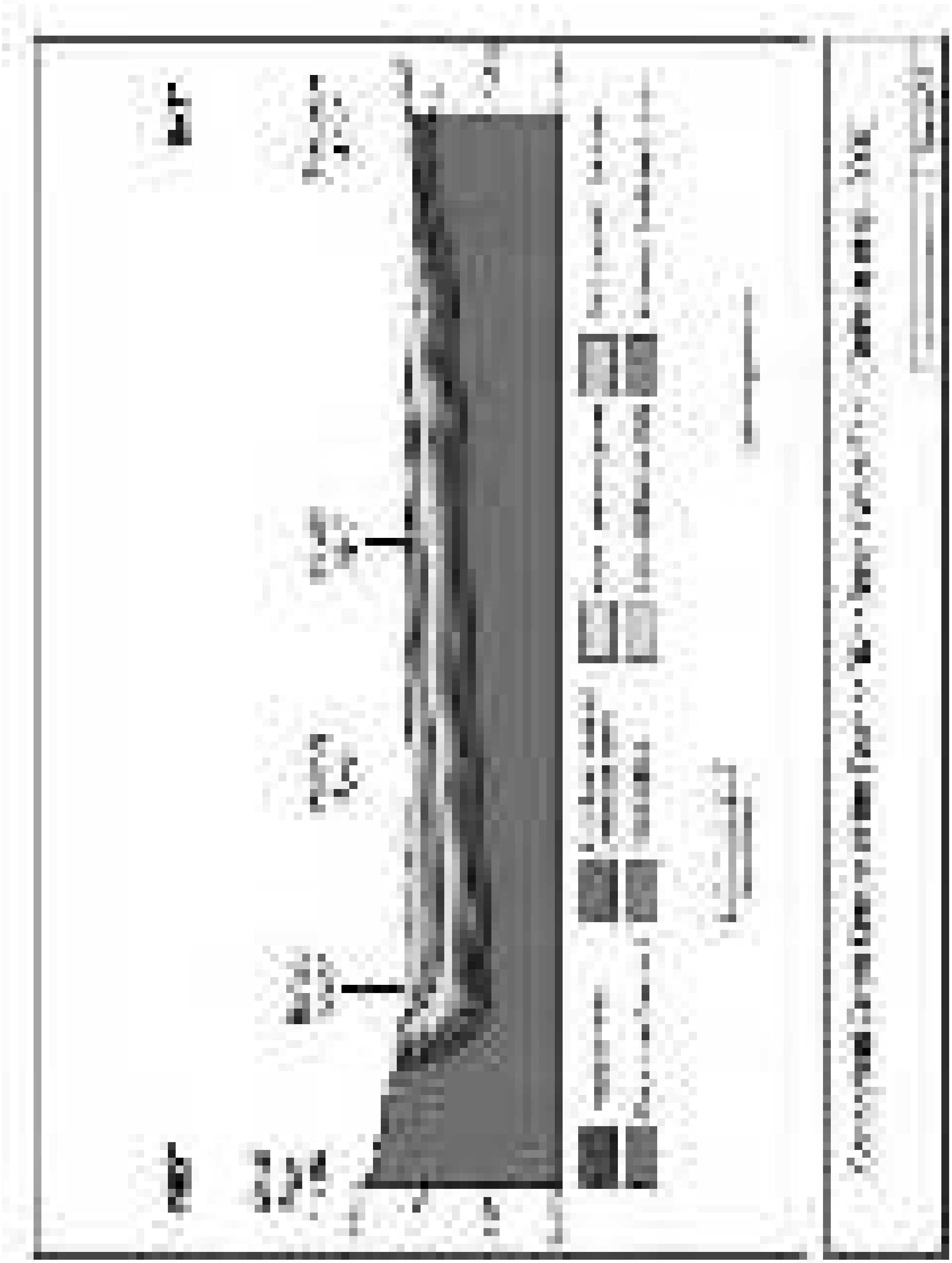
5.4 Summary

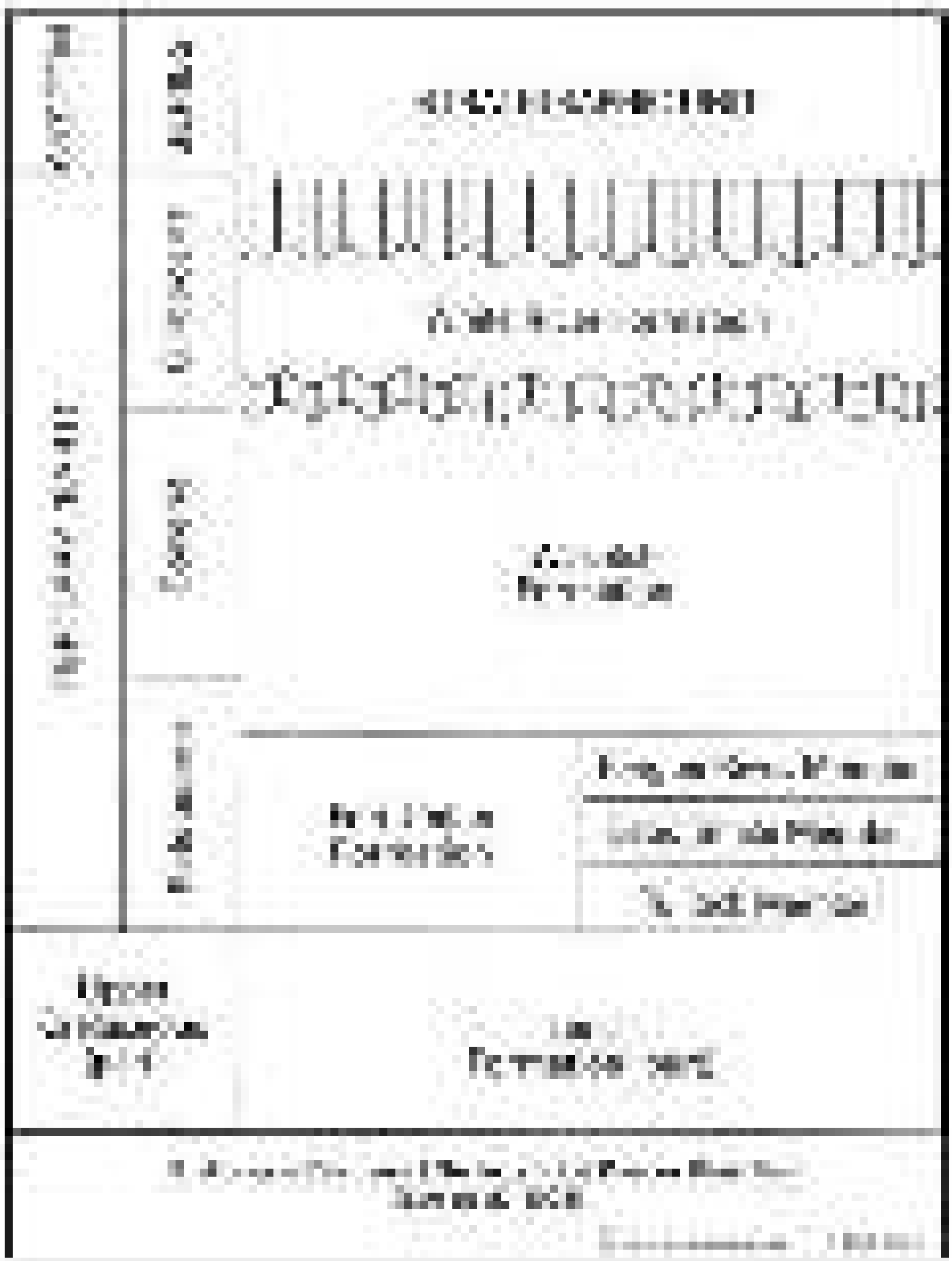
Based on the information for the Powder River Basin, the coalbeds that are being developed, or which may be developed, for coalbed methane in the Powder River Basin are also USDWs. Coalbeds in this basin are interspersed with sandstone and shale at varying depths. The Fort Union Formation that supplies municipal water to the City of Gillette is the same formation that contains the coals that are developed for coalbed methane. The coalbeds contain and transmit more water than the sandstones. The sandstones and coalbeds have been used for both the production of water and the production of coalbed methane. TDS levels in the water produced from coalbeds meet the water quality criteria for USDWs.

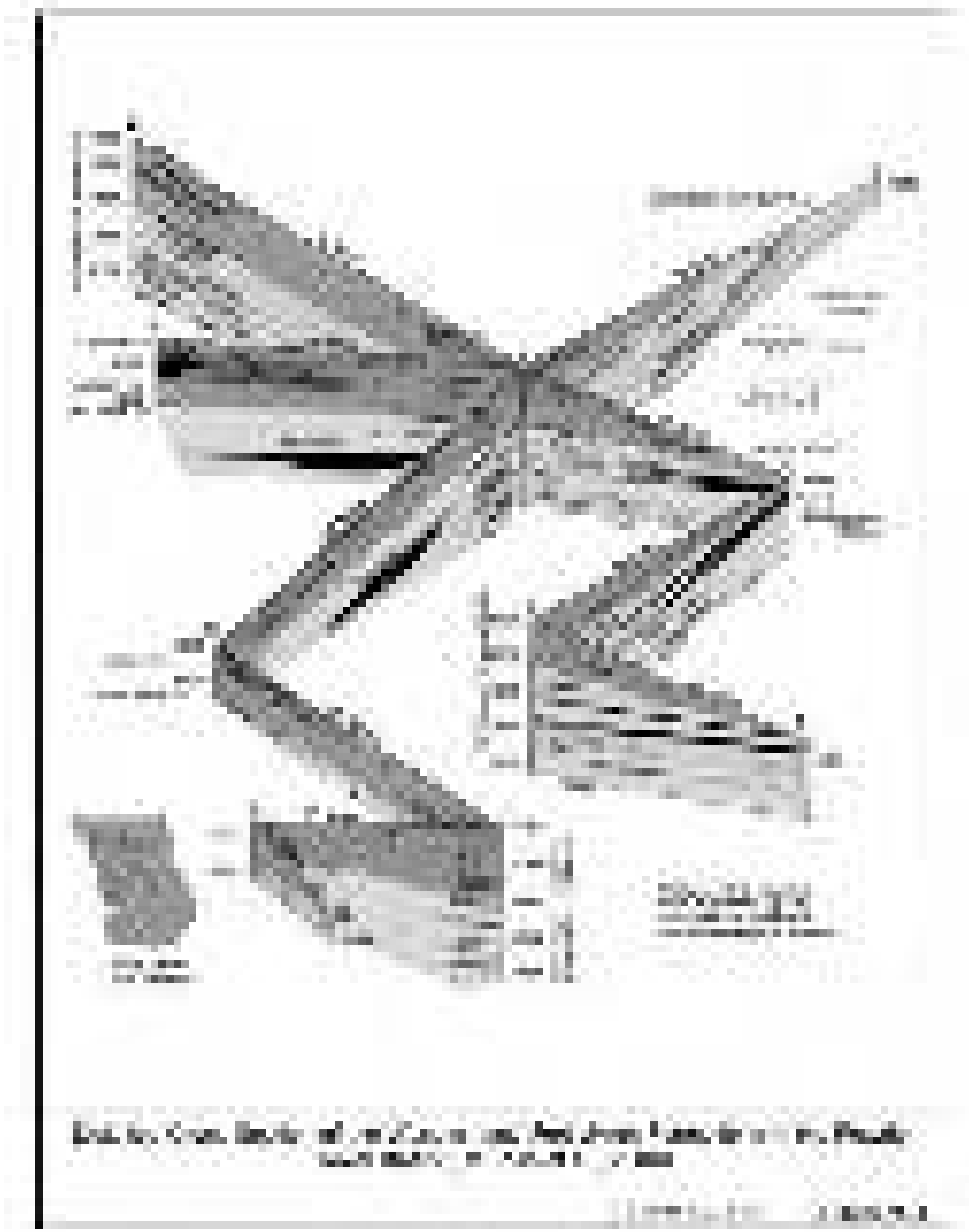
The information available indicates that currently hydraulic fracturing is not widely used in this region due to concerns about the potential for increased groundwater flow into the coalbed methane production wells and the consequent collapse of open hole wells in coal upon dewatering. According to the available literature, where hydraulic fracturing has been used in this basin, it has not been an effective method for extracting methane. Hydraulic fracturing has been conducted primarily with water, or gelled water and sand, although the recorded use of a solution of KCl was identified in the literature.

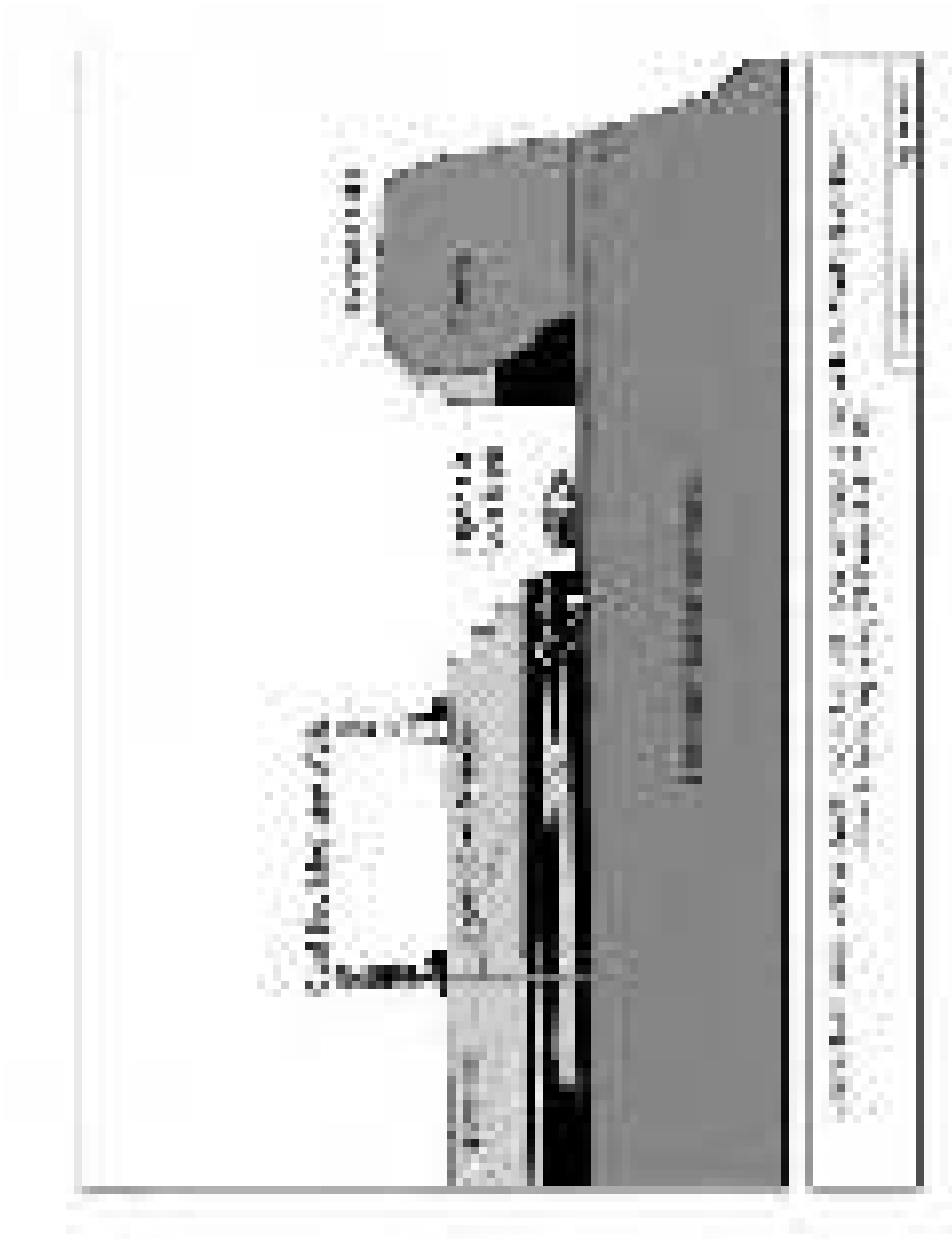


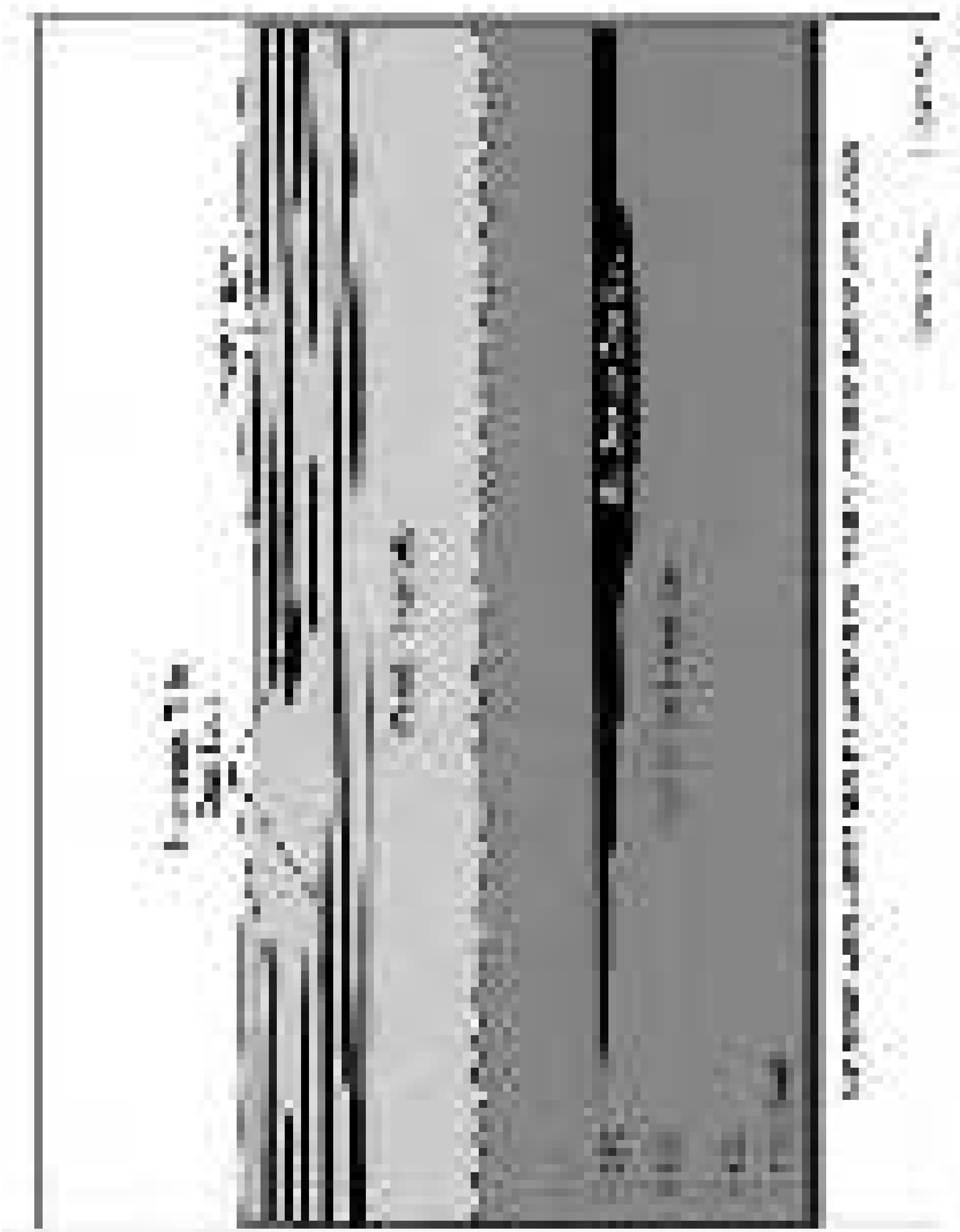


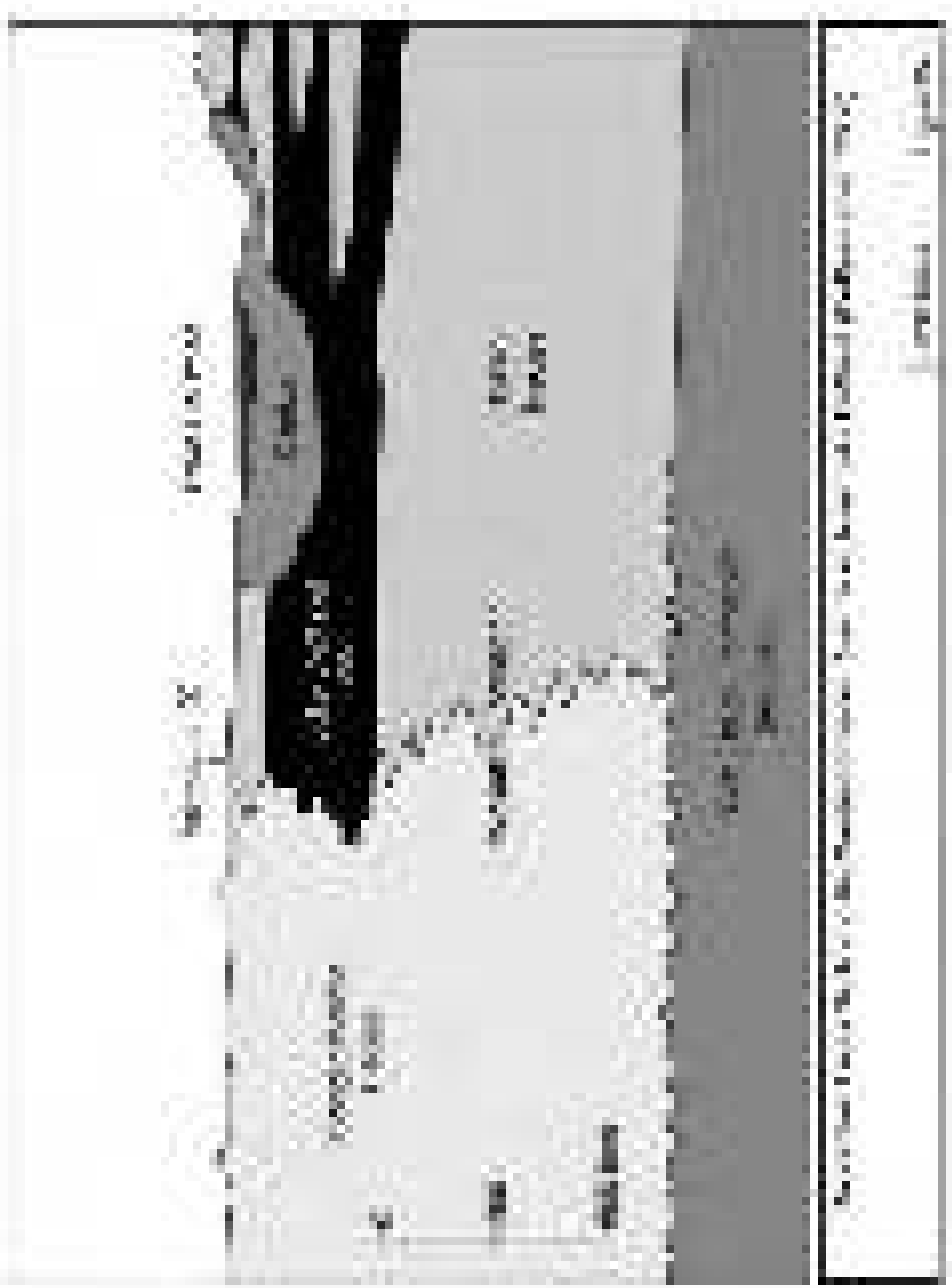


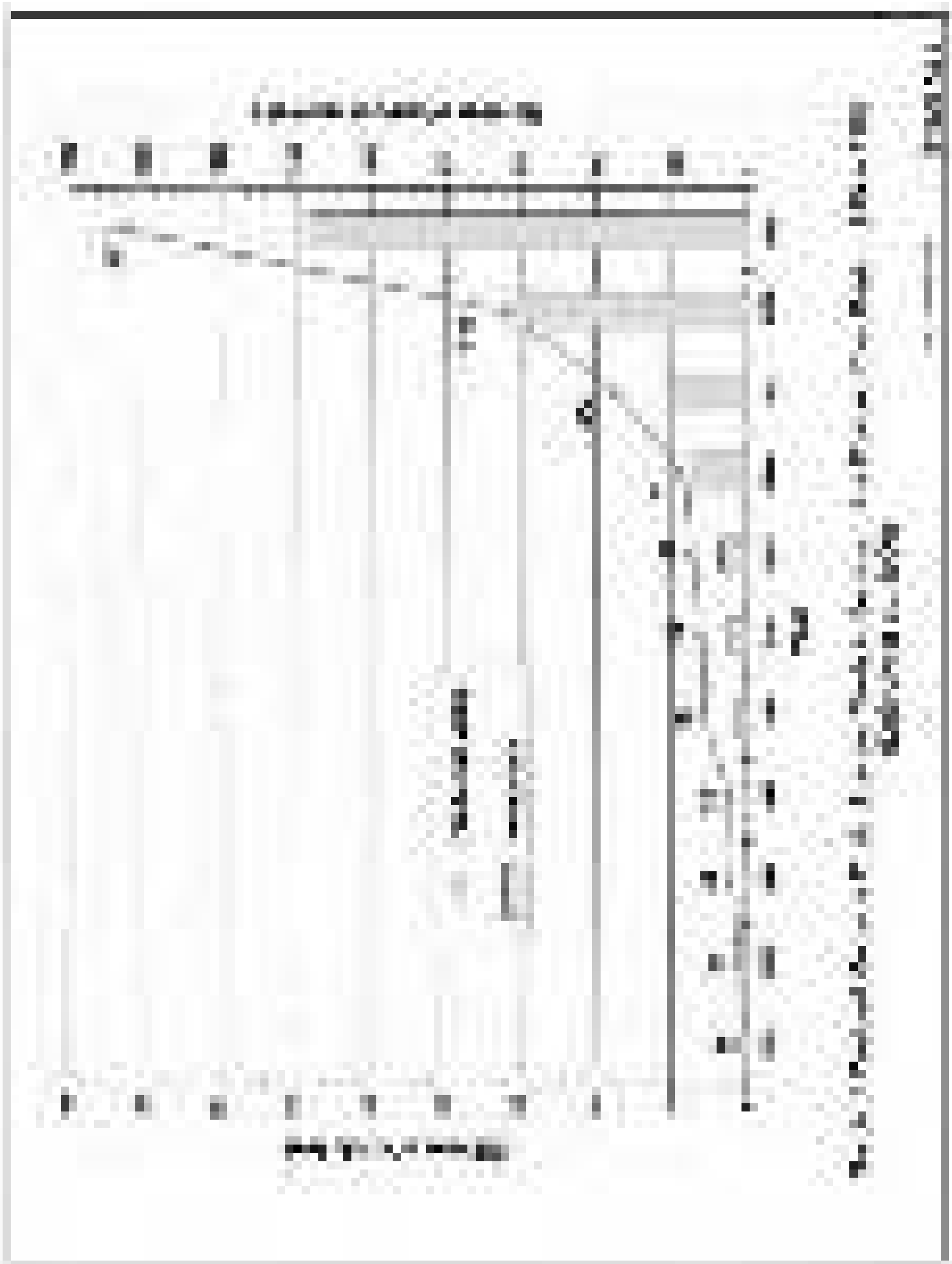












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Attachment 6
The Central Appalachian Coal Basin

The Central Appalachian Coal Basin is the middle basin of three basins that comprise the Appalachian Coal Region of the eastern United States. It includes parts of Kentucky, Tennessee, Virginia, and West Virginia (Figure A6-1). It covers approximately 23,000 square miles, contains six major Pennsylvanian age coal seams, and contains an estimated 5 trillion cubic feet (Tcf) of coalbed methane (Zebrowitz et al., 1991; Zuber, 1998). These coal seams typically contain multiple coalbeds that are widely distributed (Zuber, 1998). The coals seams, from oldest to youngest (West Virginia/Virginia name), are the Pocahontas No. 3, Pocahontas No. 4, Fire Creek/Lower Horsepen, Beckley/War Creek, Sewell/Lower Seaboard, and Iager/Jawbone (Kelafant et al., 1988). The Pocahontas coal seams include the Squire Jim and Nos. 1 to 7 and Nos. 3 and 4 are the thickest and most areally extensive. The majority of the coalbed methane (2.7 Tcf) occurs in the Pocahontas seams (Kelafant et al., 1988). The highest potential for methane development is in a small, 3,000 square mile area in southwest Virginia and south central West Virginia, where target coal seams achieve their greatest thickness and occur at depths of about 1,000 to 2,000 feet (Kelafant et al., 1988). The Gas Technology Institute (GTI) reported that the entire basin’s annual production was 52.9 billion cubic feet (Bcf) of gas in 2000 (GTI, 2002).

6.1 Basin Geology

The Central Appalachian Basin is characterized structurally by broad, open, northeast-southwest trending folds that typically dip less than five degrees (Kelafant et al., 1988) (Figure A6-2). The only documented exception to this is the Pine Mountain Overthrust Block in the southeast portion of the basin (Kelafant et al., 1988). Faults and folds associated with this 25 mile-wide and 125 mile-long structural feature are more intense as evidenced by overturned beds and even brecciated zones in some locations (Kelafant et al., 1988). The overthrust block is believed to have been transported about five miles from the southeast to the northwest (Kelafant et al., 1988). The two dominant joint patterns within the coals are most likely due to the basin having undergone two distinct patterns of structural deformation. These deformations include the Appalachian Orogeny and the tectonic event associated with development of the Pine Mountain overthrust (Kelafant et al., 1988).

The regional dip of coal-bearing Pennsylvanian strata is to the northwest at a rate of 75 feet per mile (Kelafant et al., 1988). Sedimentation within the Central Appalachian Basin was influenced somewhat by the Rome Trough, an Early Cambrian graben structure. Sediment deposition during early Pennsylvanian time (about 320 million years ago) occurred to the southeast of the Rome Trough in a rapidly but intermittently subsiding basin (Kelafant et al., 1988). As this tectonic activity began to abate in the Central Appalachian Basin, subsidence to the northeast of the Rome Trough began to form the Northern Appalachian Basin. However, subsidence rates in the Northern Appalachian Basin were comparatively slower, enabling the formation of more regionally extensive coalbeds (Kelafant et al., 1988).

There are three coal-bearing formations in the Central Appalachian Basin (Kelafant et al., 1988). From deepest to shallowest, they are the Pocahontas Formation, the New River/Lee Formation, and the Kanawha/Norton Formation. Each formation [Pennsylvanian in age (approximately 320 to 290 million years old)] is part of the Pottsville Group, and has varying nomenclature from state to state (Kelafant et al., 1988).

The Pocahontas Formation directly overlies the Mississippian Bluestone Formation, and was deposited in an unstable basin that was rapidly subsiding to the southeast (Kelafant et al., 1988). This is reflected in the thickness of the formation, which is thickest in the southeast and thins to the northwest. It also thins to the south and west due to erosion caused by the basal sandstone member of the overlying New River/Lee Formation (Kelafant et al., 1988). The Pocahontas Formation reaches its maximum thickness of 750 feet near Pocahontas, Virginia (Kelafant et al., 1988). The formation consists mostly of massively bedded, medium-grained subgraywacke, which can be locally conglomeratic (Kelafant, 1988). Gray siltstones and shales are interbedded within the sandstone (subgraywacke) unit, and coal seams comprise about two percent of the total thickness of the Pocahontas Formation (Kelafant et al., 1988).

The New River/Lee Formation conformably overlies the Pocahontas Formation in the northeastern portions of the basin (i.e., there are no time gaps in the depositional record), but there is an unconformity in the east-central portion of the basin (Kelafant et al., 1988). In the southern portion of the basin, the New River/Lee Formation unconformably overlies the Bluestone Formation. It is difficult to correlate this formation across state boundaries as nomenclature varies (Kelafant et al., 1988). The overall thickness of the formation decreases from east to west, with the thickest portion (1,000 feet) in parts of Virginia and West Virginia, lessening to fewer than 100 feet along the Ohio River in Kentucky (Kelafant et al., 1988). Coalbeds encountered in the New River/Lee Formation include the Fire Creek/Lower Horsepen, Beckley/War Creek, Sewell/Lower Seaboard, and the Iager/Jawbone (Kelafant et al., 1988). These coalbeds thin and pinch-out towards the south and west; therefore, there are no equivalent coalbeds in Kentucky and Tennessee (Kelafant et al., 1988).

The Kanawha/Norton Formation varies from a maximum thickness of 2,000 feet in West Virginia to less than 600 feet in portions of Dickenson and Wise Counties, Virginia (Kelafant et al., 1988). The formation is composed of irregular, thin- to massively-bedded subgraywackes interbedded with shale. Several thin carbonate units also occur within the formation as well as over 40 multi-bedded coalbeds.

All coal seams within the basin occur within the Pennsylvanian Pottsville Group (Figure A6-3). Specific stratigraphic nomenclature varies from state to state within the basin. (Names used in this summary are consistent with the West Virginia/Virginia nomenclature).

The Pocahontas No. 3 coal seam ranges in depth from outcrop along the northeastern edge of the basin to about 2,500 feet, with a thickness ranging up to seven feet (Kelafant et al., 1988). Depths to the Pocahontas No. 4 coal seam are somewhat similar to those for the Pocahontas No. 3 coal seam, as the No. 4 seam overlies the No. 3 seam by roughly 30 to 100 feet. The thickness of the No. 3 coal seam varies, with a maximum of approximately seven feet (Kelafant et al., 1988). The Fire Creek/Lower Horsepen coalbed ranges in depth from roughly 500 feet over half of its area, to a maximum depth of approximately 1,500 feet, with a maximum thickness of roughly six feet (Kelafant et al., 1988). The Beckley/War Creek coalbed is approximately two to five feet thick, and reaches to a maximum depth of about 2,000 feet (Kelafant et al., 1988). The Sewell/Lower Seaboard coalbed is fairly shallow, less than 500 feet in depth over half the area it covers, reaching to a depth over 1,000 feet in one small area. While this coal ranges in thickness from two to six feet, it averages about two feet in West Virginia and one foot in Virginia (Kelafant et al., 1988). The youngest targeted coal seam, the Iaeger/Jawbone, is generally less than 500 feet in depth, reaching its maximum depth of over 1,000 feet in two Virginia Counties. The thickness of the Iaeger/Jawbone coal ranges from two to six feet (Kelafant et al., 1988). Figures A6-4 through A6-9 are isopach maps for the six major coal groups of the Appalachian Coal Basin (adapted from Kelafant, et al., 1988).

6.2 Basin Hydrology and USDW Identification

The primary aquifer in the Kentucky portion of the Central Appalachian Basin is a Pennsylvanian sandstone aquifer underlain by limestone aquifers (National Water Summary, 1984). Water wells are typically 75 to 100 feet deep in the Pennsylvanian aquifer and commonly produce one to five gallons per minute of water (National Water Summary, 1984). The basin is located in a portion of the Cumberland Plateau physiographic province in Tennessee (National Water Summary, 1984). The primary aquifer in this area is a Pennsylvanian sandstone aquifer, comprising water-bearing sandstone and conglomerate subunits with interbedded shale and coal (National Water Summary, 1984). Water wells are typically 100 to 200 feet deep and usually produce 5 to 50 gallons per minute of water (National Water Summary, 1984). In Virginia, the basin is located in a portion of the Appalachian Plateau physiographic province. The primary aquifer in this region is the Appalachian Plateau Aquifer, a consolidated sedimentary aquifer consisting of sandstone, shale, siltstone, and coal (National Water Summary, 1984). Water wells are typically 50 to 200 feet deep, and commonly produce one to 50 gallons per minute of water (National Water Summary, 1984). In West Virginia, the basin is in a portion of the Appalachian Plateaus physiographic province of that state. The primary aquifers in this area are Lower Pennsylvanian aquifers, which include the Pottsville Group (National Water Summary, 1984). Wells are commonly 50 to 300 feet deep and typically produce one to 100 gallons per minute of water (National Water Summary, 1984).

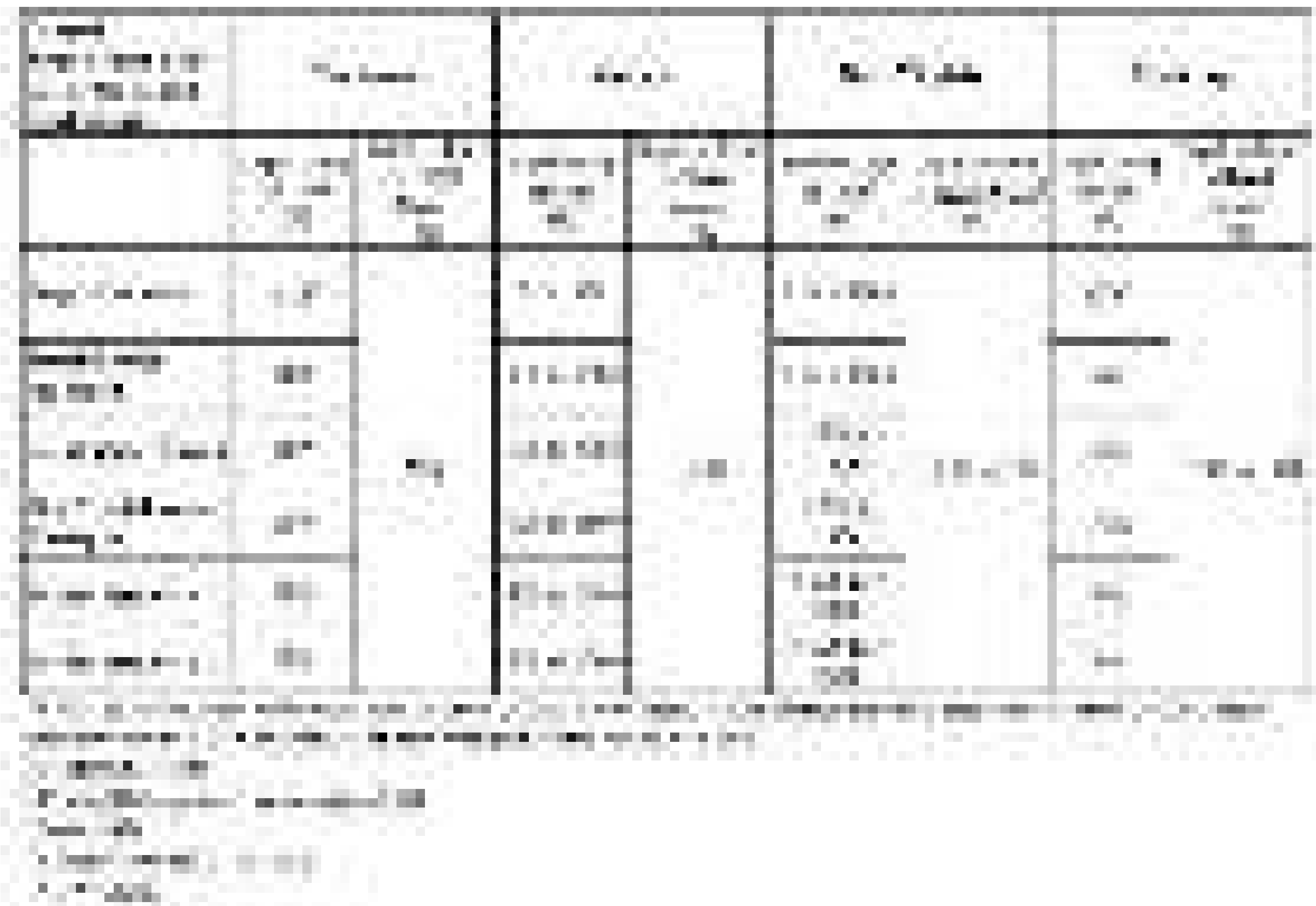
Produced water volumes from coal seams within the Central Appalachian Basin are relatively small, typically only several barrels or less per day per well, with high total dissolved solid (TDS) levels, usually greater than 30,000 milligrams per liter (mg/L) (Quarterly Review, 1993). Half the states (Kentucky and Ohio) within the Central Appalachian Basin have maps to locate the undulating interface between saline and freshwater aquifers. The remaining states (Tennessee and Virginia) have no maps defining this interface. Mike Burton (2001), a geologist with the Oil and Gas Office of the Tennessee Geology Division (TGD), reports that the state has no data relating to coalbed methane, which suggests that little or no coalbed methane extraction occurs inside Tennessee's borders (Burton, 2001). Luke Ewing (Ewing, 2001) of the TGD reported that the state had no aquifer maps. Scotty Sorles (Sorles, 2001) of Tennessee's Underground Injection Control Program mentioned that within the state, produced water disposal methods vary on a site-by-site basis. Depending on site characteristics, all injected waters must either be returned to the formation from which they came, or be treated to drinking water levels prior to injection elsewhere (Sorles, 2001).

Robert Wilson, Director of the Virginia's Division of Gas & Oil, stated that there is no mapping program for underground sources of drinking water (USDWs) or for the fresh/saline groundwater interface in Virginia. He reported that the most potable water is found far above the coal zones used for coalbed methane extraction, with fresh water typically found at less than 300 feet deep. He believes most drinking water in southwestern Virginia comes from wells in fractured bedrock aquifers or shallow coal aquifers, or, in some areas, directly from springs. Mr. Wilson also stated that some coalbed methane exploration has moved to shallower coal seams. The Commonwealth of Virginia has instituted a voluntary program concerning depths at which hydraulic fracturing may be performed (Virginia Division of Oil and Gas, 2002). This program involves an operator's determination of the elevations of the lowest topographic point and the deepest water well within a 1,500-foot radius of any proposed extraction well (Wilson, 2001). Hydraulic fracturing should occur at least 500 feet deeper than the lower of these two points (Wilson, 2001).

According to Mr. Tony Scales of the Virginia Department of Mines, Minerals and Energy, coal seams are the most permeable layers in the geologic subsurface in Virginia. For this reason, many private wells in the coalbed methane-producing counties are finished within the coalbeds. Mr. Scales stated that impacts to water supplies have occurred if the coal seams have been punctured by coalbed methane well drilling. The puncture hole acts as a conduit for the flow of water out of the coals and into lower formations. The puncture hole also allows methane to rise up to the surface (Virginia Department of Mines, Minerals, and Energy, 2002).

The following table contains information concerning the relative locations of the base of the zone of fresh water and potential methane-bearing coalbeds in the Central Appalachian Coal Basin. The table provides useful information that can help in determining whether coalbeds being used or slated for methane development lie within USDWs. Note that the 10,000 mg/L level of TDS in groundwater is the water quality criterion for a USDW. The depth to the USDW will thus lie well below the fresh water/ saline water interface. The area of focus for coalbed methane exploration in the basin only covers parts of Virginia and West Virginia (Figure A6-1). In Virginia, the depth to the base of fresh water is approximately 300 feet, whereas the depths to the bases of USDWs are greater. Thus, as can be seen in Table A6-1, methane-producing coalbeds could lie within USDWs in Virginia. West Virginia's interface between fresh and saline water (Foster, 1980) is based on a qualitative assessment, and is estimated at 280 to 730 feet. Again, the depths to USDWs are greater, and thus the coalbeds of interest could lie within potential USDWs in West Virginia. Finally, in Kentucky the interface between fresh water and saline water is based on a TDS level of 1,000 mg/L (Hopkins, 1966). Although the depths to methane-producing coalbeds in Kentucky are not listed in the Table A6-1, it is possible that, as in Virginia and West Virginia, such depths could be lower than the base of USDWs in Kentucky.

Table A6-1. Relative Locations of USDWs and Methane-Bearing Coalbeds



6.3 Coalbed Methane Production Activity

Coalbed methane operators in the Central Appalachian Basin include Equitable Resources, CONSOL (Consolidation Coal Company), and Pocahontas Gas Partnership, all located in Virginia (Zuber, 1998). GTI reported that the entire basin's annual production was 52.9 Bcf of gas in 2000 (GTI, 2002).

The Nora Field in southwestern Virginia is one of the better known coalbed methane production fields. Equitable Resources operates the Nora Field in southwestern Virginia. According to the

Virginia Division of Gas and Oil, over 700 coalbed methane wells were drilled in the Nora Field in 2002 and more than 1,800 coalbed methane wells were drilled in southwestern Virginia's Buchanan County (VA Division of Gas and Oil, 2002). Foam or water is used as the fracturing fluid and about 70,000 to 100,000 pounds of sand per well serve as proppant (Zuber, 1998). CONSOL and Pocahontas Gas Partnership produce coal methane from coal mine developments in Buchanan County, in southwestern Virginia (Zuber, 1998).

Many other smaller test projects were carried out in the basin in the 1970s, including the New River Coal Company/Lick Run Mine Project, Department of Energy (DOE)/Clinchfield Coal Company Project, U.S. Bureau of Mines (USBM)/Occidental Research/Island Creek Coal Company Project, Gas Research Institute/Wyoming County Co-op Project, USBM Federal No. 1 Project, and the Consolidation Coal Company/ Kepler Mine Project (Hunt and Steele, 1991). These projects were very small (five wells or fewer) and achieved limited success in terms of production. During development of some wells in the DOE/Clinchfield Coal Company project and the USBM Federal Project No. 1, fracture treatments "screened out" (i.e., the proppant placement failed), affecting those coalbed methane wells' production viability.

No coalbed methane production occurred in Tennessee between 1995 and 1997 (Lyons, 1997). Three coalbed methane wells produced gas from 1957 to 1980 in Harlan County, Kentucky, and only one test well was in production in the early 1990s in eastern Kentucky (Lyons, 1997). The Kentucky Department of Mines and Minerals website (2002) indicated that 1,338 gas wells were in operation in Kentucky at the end of 2000, but no indication was given whether these were coalbed methane wells or conventional gas wells.

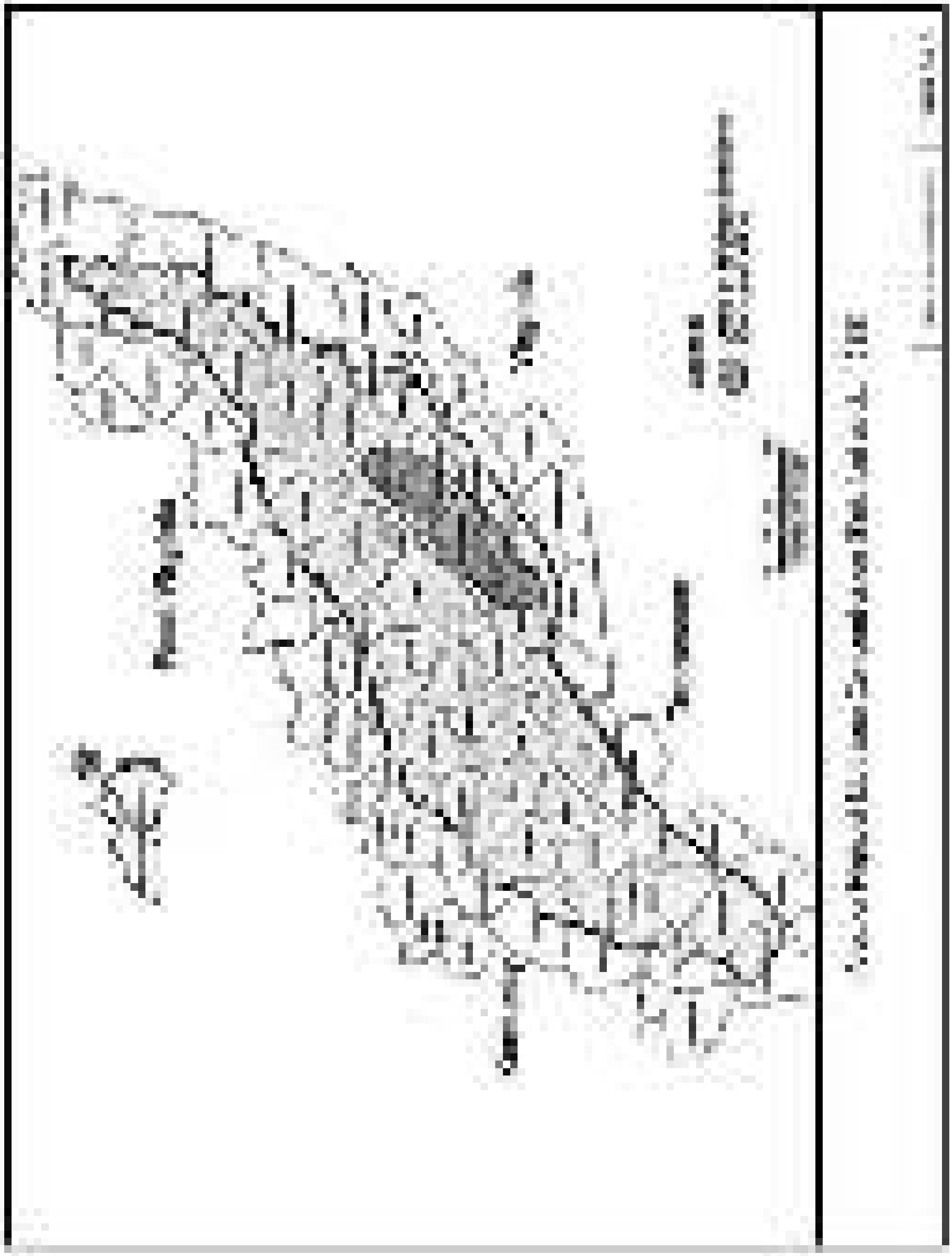
In August 2001, EPA attended a hydraulic fracturing field visit in the Central Appalachian coal basin in Virginia. Pocahontas Oil & Gas, a subsidiary of Consol Energy, Inc., invited EPA personnel to a well location where a hydraulic fracturing treatment was being performed by Halliburton Energy Services, Inc. This treatment employed a variety of fluids and additives to create fractures in select coal seams at various depths. The main fracturing fluid was nitrogen foam (70% nitrogen / 30% water mixture). Prior to injection of the foam, 6 barrels of 15 percent hydrochloric acid were introduced into the well to dissolve the grout surrounding the injection perforations. Once the fracture was propagated to its maximum extent, 16/30 sand suspended in a 10-pound linear gel was injected to prop the fracture open. All the fluids and additives used were produced by Halliburton, including a scale inhibitor and a microbicide additive. Halliburton staff stated that typical fractures range in length from 300 to 600 feet from the well bore in either direction, but that fractures have been known to extend from as few as 150 feet to as many as 1,500 feet in length. According to the fracturing engineer on-site, fracture widths range from one eighth of an inch to almost one and a half inches (Virginia Site Visit, 2001).

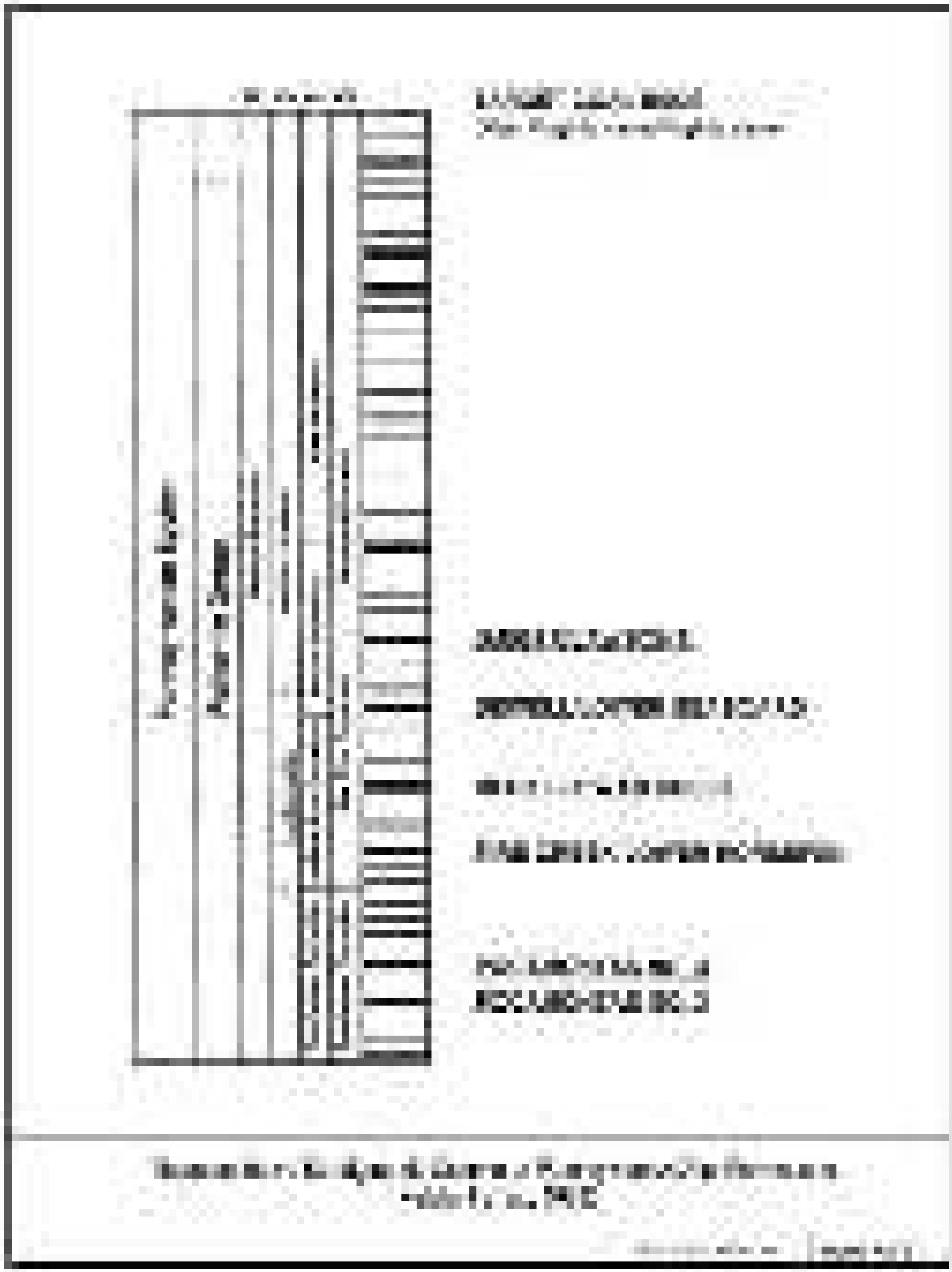
Once a well is drilled and fractured in Virginia, several weeks might elapse before fracturing fluid flowback is initiated because a pipeline system must be constructed to transport the produced coalbed methane away from the well. Flowback fracturing fluids are collected in lined pits and tanks and transported off-site for disposal. The State of Virginia does not regulate the use of any drilling or fracturing fluids (Wilson, 2001).

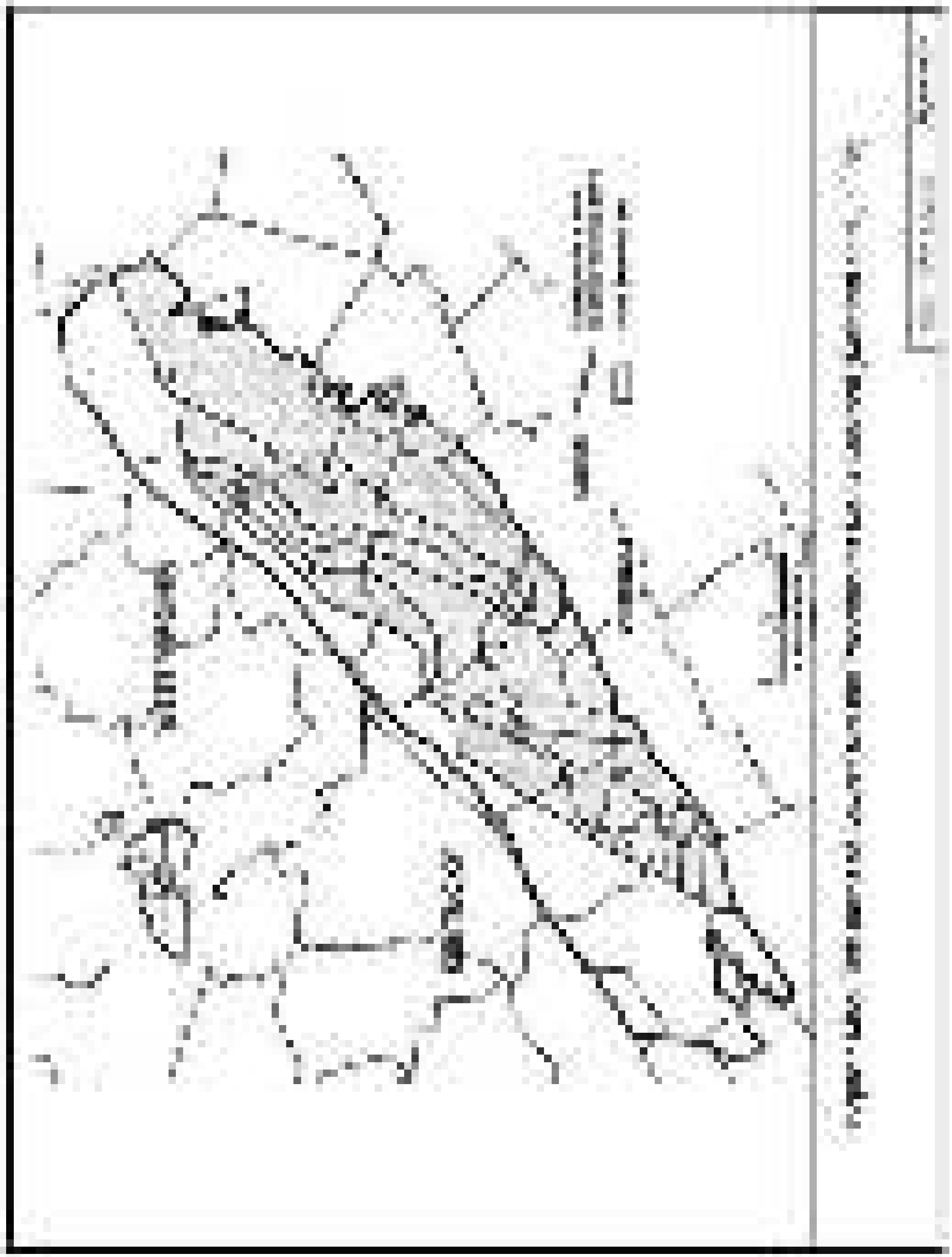
6.4 Summary

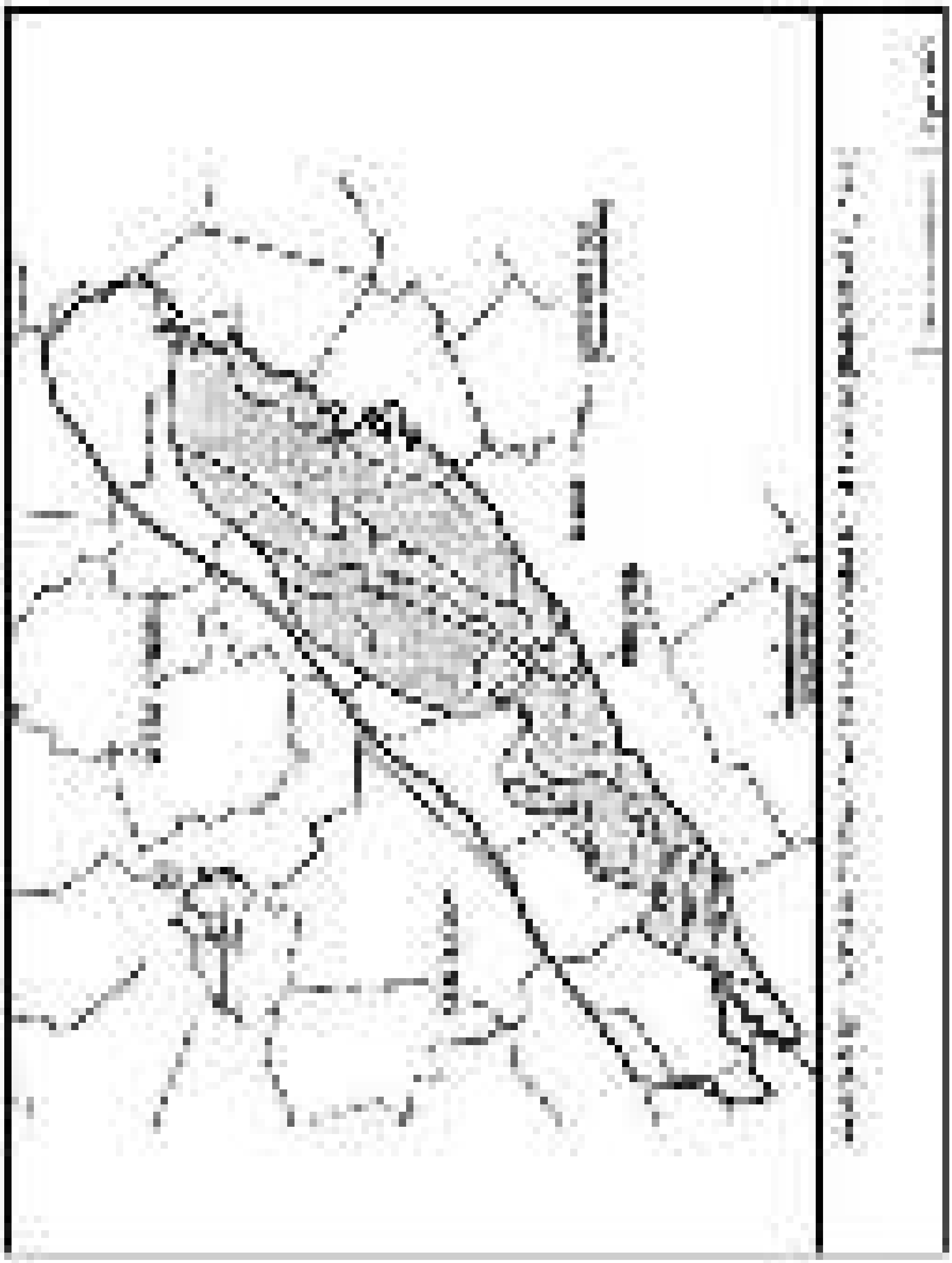
The area with the highest potential for coalbed methane production in the Central Appalachian Coal Basin is southwestern Virginia (Dickenson and Buchanan Counties) and southern West Virginia (Wyoming and McDowell Counties) (Figure A6-1). The coal seams achieve their greatest thickness in these regions and occur at depths of approximately 1,000 to 2,000 feet. Based on Table A6-1, methane-producing coal may lie within a USDW, providing the potential for impact of water supplies.

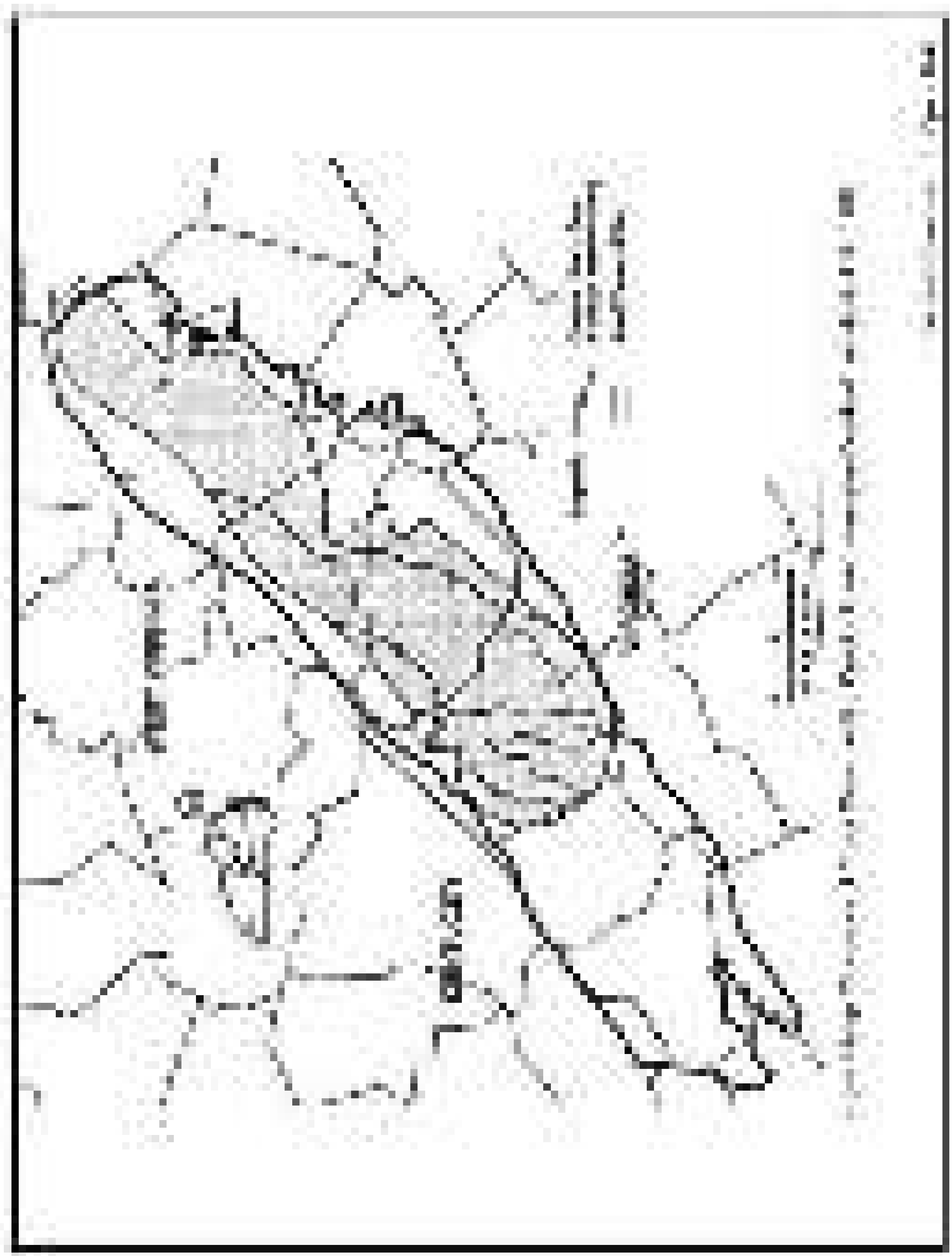
Hydraulic fracturing is common practice in this region. Foam and water are the fracturing fluids of choice and sand serves as the proppant. Because most of the coal strata dip, a coalbed methane well's location within the basin may determine if hydraulic fracturing during the well's development will likely affect water quality within the surrounding USDW. For instance, on the northeastern side of the basin, the depth to the Pocahontas No. 3 coalbed is less than 500 feet. This depth increases to over 2,000 feet in the western portion of the basin, in the direction of the coal seam dip. Therefore, a well tapping this coal seam in the western portion of the basin may be below the base of a USDW but a well tapping this coal seam in the eastern portion of the basin may be within a USDW. Additionally, the base of the freshwater is not a flat surface, but rather an undulating one. These factors indicate that the relationship between a coalbed and a USDW must be determined on a site-specific basis.

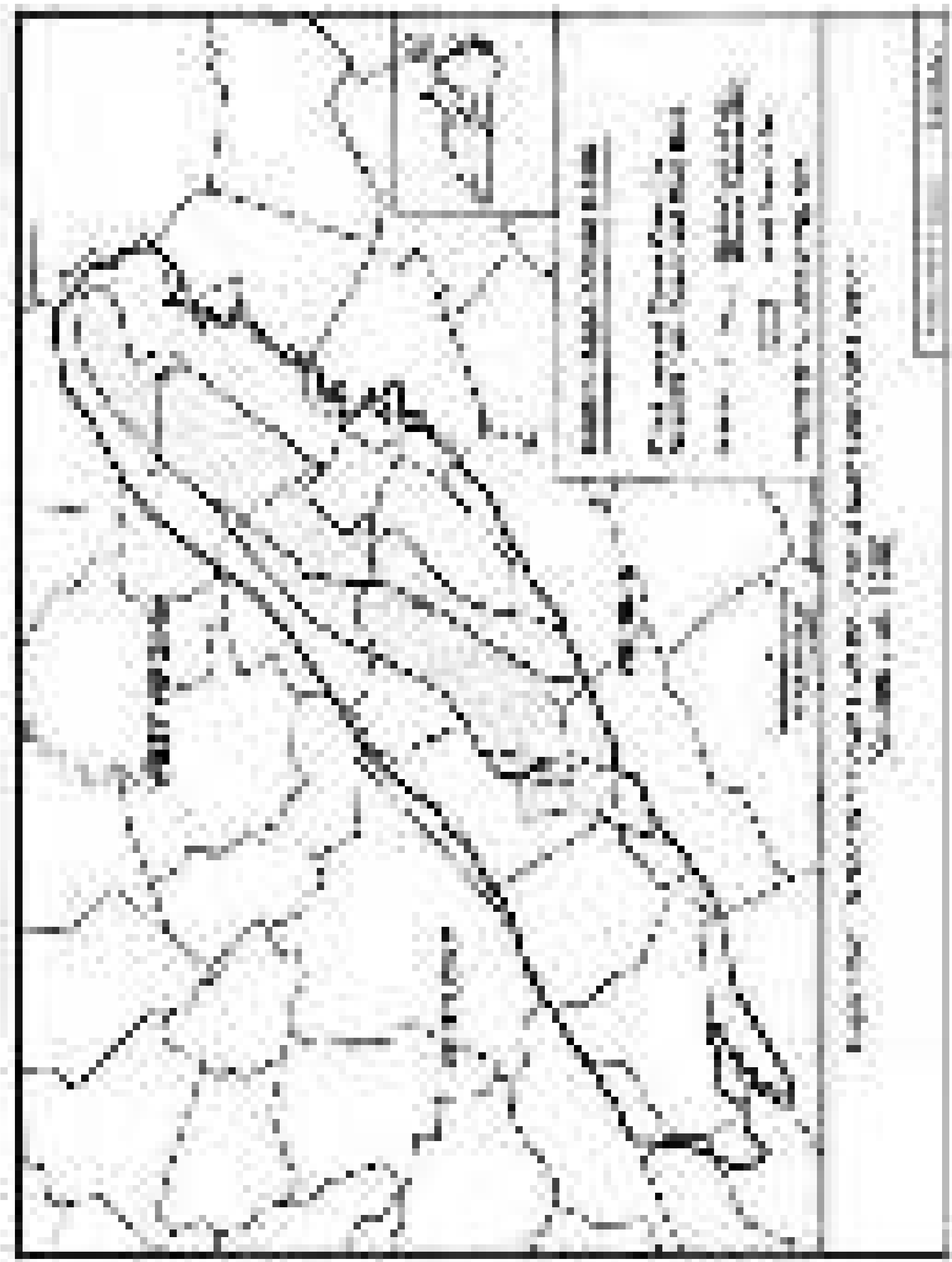


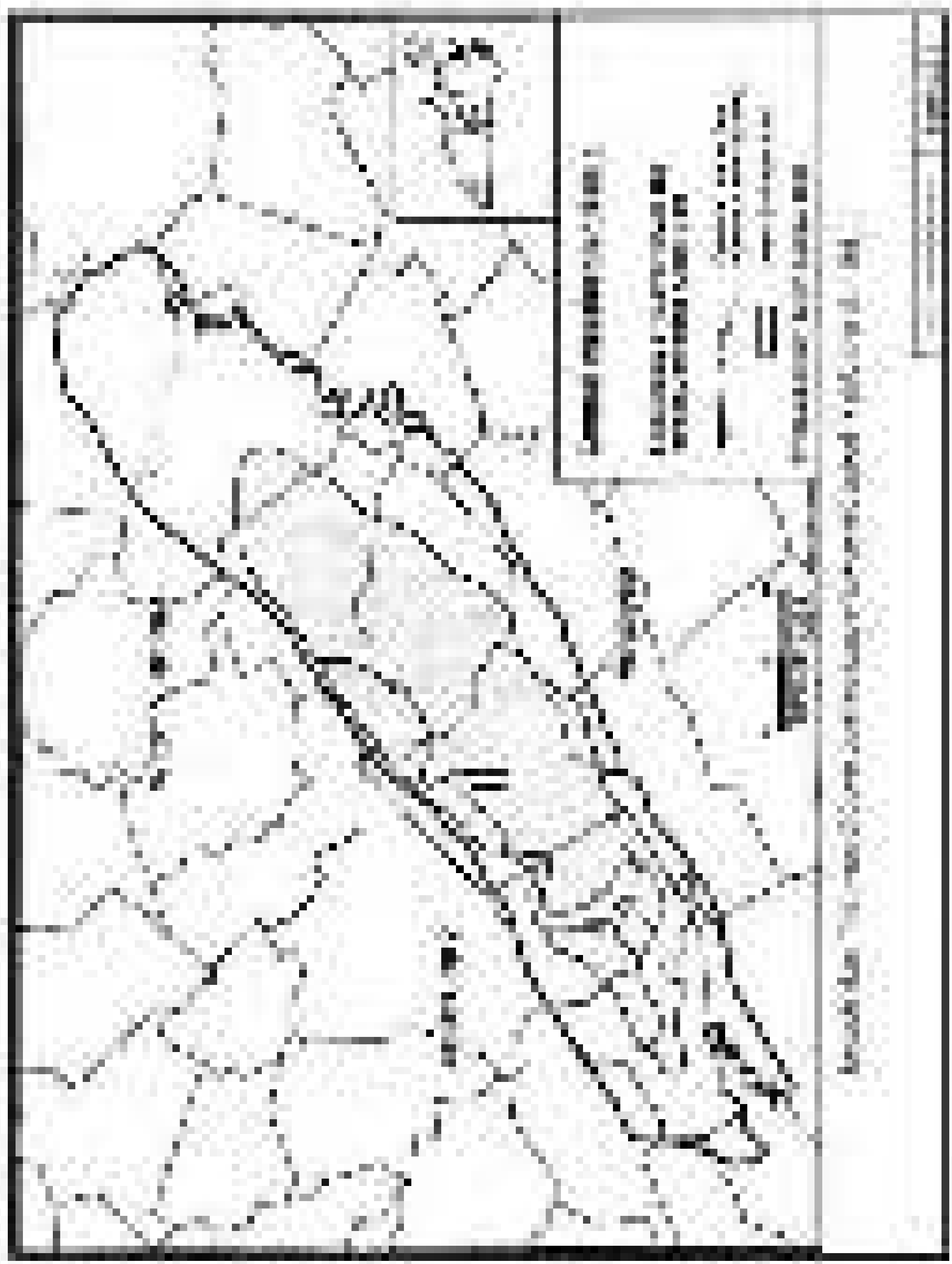












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Attachment 7
The Northern Appalachian Coal Basin

The Northern Appalachian Coal Basin is the northernmost of the three basins comprising the Appalachian Coal Region of the eastern United States, and includes parts of the States of Pennsylvania, West Virginia, Ohio, Kentucky, and Maryland (Figure A7-1). The basin trends northeast-southwest and the Rome Trough, a major graben structure, forms the southeastern and southern structural boundaries (Kelafant et al., 1988). The basin is bounded on the northeast, north, and west by outcropping Pennsylvanian-aged sediments (Kelafant et al., 1988). The basin lies completely within the Appalachian Plateau geomorphic province, covering an area of approximately 43,700 square miles (Adams et al., 1984 as cited in Pennsylvania Department of Conservation and Natural Resources, 2002). It consists of six Pennsylvanian age coal units, and contains an estimated 61 trillion cubic feet of coalbed methane (Kelafant et al., 1988). Coal seam depths range from surface outcrops to up to 2,000 feet below ground surface, with most coal occurring at depths shallower than 1,000 feet (Quarterly Review, 1993). Annual coalbed methane production stood at 1.41 billion cubic feet in 2000 (GTI, 2002).

7.1 Basin Geology

The six Pennsylvanian aged coal zones located within the Northern Appalachian Coal Basin are the Brookville-Clarion, Kittanning, Freeport, Pittsburgh, Sewickley, and the Waynesburg. These coal units are contained within the Pottsville, Allegheny, and the Monongahela Groups (Figure A7-3) (Kelafant et al., 1988).

In the Northern Appalachian Basin, the Pottsville stratigraphic group is generally 200 to 300 feet thick, and thins to the north and west into Ohio and Pennsylvania (Kelafant et al., 1988). The coals in this group are interbedded with fluvial and deltaic sands and shales and are capped by marine limestones and shales (Kelafant et al., 1988). Deposition of this group took place on irregular Mississippian terrain, forming thin and erratic coals (Kelafant et al., 1988).

The Allegheny Group reaches a maximum thickness of 200 to 300 feet in western Maryland and thins westward to about 150 to 200 feet in Ohio. Deposition of this group occurred as cyclothem-type sedimentation, resulting in a complex sequence of lenticular, thin- to massive-bedded subgraywacke, shale, and mudstone interbedded with clays and coal (Kelafant et al., 1988). Due to their alluvial and delta plain depositional environments, Allegheny coals, which include the Brookville/Clarion, Kittanning, and the Freeport, are 2 to 6 feet thick and aerially extensive (Kelafant et al., 1988). The coalbeds decrease in number from the eastern to the western edge of the basin (Kelafant et al., 1988).

The Monongahela Group was deposited primarily in lacustrine and swamp environments. Some of the most important economic coals of the basin were deposited in large lakes, such as the Pittsburgh and Sewickley coals (Kelafant et al., 1988). The Monongahela Group is thickest along the Monongahela River at 400 feet and thins to 250 feet in the southwest along the Ohio River. Shales, mudstones, and freshwater limestones are the major rock types of the Group (Kelafant et al., 1988). The Waynesburg coals are also contained within the Monongahela Group. In general, the coalbeds of the Monongahela Group are laterally extensive.

The total thickness of the Pennsylvanian-aged coal system averages 25 feet; however, better-developed seams within the coal zones can increase in thickness by up to twice the average (Quarterly Review, 1993). Within the Pennsylvanian Coal System, the deepest coal, the Brookville-Clarion, ranges in depth from surface exposures in anticlines to 2,000 feet below ground surface. The Kittanning Group reaches a maximum depth of 2,000 feet and is approximately 800 feet deep in more than half the area in which the group occurs. The distance between the Upper and Lower Kittanning is approximately 100 feet (Kelafant et al., 1988). Freeport coals are at a maximum depth of 1,800 feet in the central portion of the Northern Appalachian Coal Basin. The Upper and Lower Freeport are separated vertically by a distance of 40 to 60 feet (Kelafant et al., 1988). The Pittsburgh coals achieve a maximum depth of 1,200 feet and roughly half of the coals can be found at depths greater than 400 feet (Kelafant et al., 1988). Sewickley coals are deeper than 400 feet with the deepest coals located at 1,200 feet below ground surface (Kelafant et al., 1988). The final and youngest group discussed here, the Waynesburg group, is the shallowest, reaching a maximum depth of 800 feet in the center of the basin. Figures A7-4 through A7-9 (adapted from Kelafant et al., 1988) are isopach maps of sediment cover for the six major coal zones of the Appalachian Coal Basin.

7.2 Basin Hydrology and USDW Identification

The Northern Appalachian Basin is situated in the Appalachian Plateaus physiographic province of the region. The primary aquifer in this area is a Pennsylvanian sandstone aquifer underlain by limestone aquifers (National Water Summary, 1984). Water wells are typically 75 to 100 feet in depth in the Pennsylvanian aquifer and commonly produce one to five gallons per minute of water (National Water Summary, 1984). The primary aquifers in the Maryland portion of the basin are Appalachian sedimentary aquifers, which are mostly sandstones, shales, and siltstones with some limestone, dolomite, and coal. Water wells here are typically 30 to 400 feet in depth and usually produce 10 to 100 gallons per minute of water (National Water Summary, 1984).

In Ohio, the primary aquifers are sandstone aquifers, shaly sandstone and carbonate aquifers, and coarse-grained aquifers (comprised of alluvium and glacial outwash) associated with river valleys (National Water Summary, 1984). Water wells within these aquifers typically range from 25 to 300 feet in depth, and common water production rates vary between 1 and 500 gallons per minute (National Water Summary, 1984).

In Pennsylvania, the primary aquifers are sandstone and shale aquifers, with smaller unconsolidated sand and gravel aquifers surrounding river courses (National Water Summary, 1984). Well depths in the sandstone and shale aquifers in Pennsylvania are usually 80 to 200 feet in depth, and the wells typically produce 5 to 60 gallons per minute of water (National Water Summary, 1984).

In West Virginia, the primary aquifer is an Upper Pennsylvanian-aged aquifer consisting of the Dunkard, Monongahela, and Conemaugh Groups (National Water Summary, 1984). This aquifer consists of nearly horizontal beds of shale, sandstone, siltstone, coal, and limestone (National Water Summary, 1984). Water wells typically extend from 50 to 300 feet in depth in this area of West Virginia, and commonly produce 1 to 30 gallons per minute of water (National Water Summary, 1984).

Individual states containing portions of the basin have developed various maps and documents locating underground sources of drinking water (USDWs) and aquifers within their state boundaries, mostly as a part of their respective Underground Injection Control (UIC) Programs. EPA's Regional Office also has information concerning the location of these resources, as not all states within the Northern Appalachian Coal Basin have primacy over their UIC Program. Water quality data from eight historic Northern Appalachian Coal Basin projects show that estimated total dissolved solids (TDS) levels ranged from 2,000 to 5,000 milligrams per liter (mg/L) at depths ranging from 500 to 1,025 feet below ground surface (Zebrowitz et al., 1991), well within EPA's water quality criterion for a USDW of less than 10,000 mg/L of TDS (40 CFR §144.3).

Most states within the Northern Appalachian Basin, including Kentucky, Ohio, and West Virginia have mapped the interface between saline and freshwater aquifers. For Maryland and Pennsylvania, no maps have been identified that define the interface between saline and freshwater aquifers. In Maryland, a deep well drilled in southern Garrett County encountered the fresh/saltwater interface at a depth of 940 feet (Duigon and Smigaj, 1985). Groundwater in Pennsylvania deeper than 450 feet is not considered to be a USDW (Platt, 2001) because of the existence of non-water producing shale from 450 to 1000 feet, and TDS levels in water below this shale that are typically greater than 100,000 mg/L. The following table contains information concerning the relative location of potential USDWs and potential methane-bearing coalbeds in the Northern Appalachian Coal Basin.

As shown in Table A7-1, coalbeds with methane production potential in the Northern Appalachian Basin do occur within USDWs, indicating the potential for impact. West Virginia's interface line between fresh and saline water (Foster, 1980) is based on a qualitative assessment, Ohio's interface line is based on a TDS level of 3,000 mg/L (Sedam and Stein, 1970), and Kentucky's interface line is based on a TDS level of 1,000 mg/L (Hopkins, 1966). In Maryland, the fresh water distinction was probably made based on a TDS level of 1,000 mg/L, as the reference refers to sodium and chloride concentrations of 1,800 mg/L and 2,900 mg/L as "high levels" (Duigon and Smigaj, 1985).

Therefore, in these states, the depth to the 10,000 mg/L level of TDS in groundwater is potentially and likely deeper than the depths presented above (Table A7-1). This assumption is confirmed by a structure elevation map (Figure A7-6) of the Upper and Lower Freeport Sandstones of the Upper Allegheny Group (Figure A7-3) in Ohio. With the increasing depth of these stratigraphic units toward the basin center, much of the formation waters in these units south of the easternmost counties in Ohio contain TDS levels in excess of 10,000 mg/L (Vogel, 1982). Likewise, the Pittsburgh Group Coals in Pennsylvania range in depth from outcrop to 1,200 feet below ground surface (Figure A7-7). Over this length of “dip”, it is likely that the coals intersect drinking water aquifers before they reach depths where TDS levels exceed the 10,000 mg/L TDS water quality criterion of a USDW.

7.3 Coalbed Methane Production Activity

Coalbed methane has been produced in commercial quantities from the Pittsburgh coalbed of the Northern Appalachian Coal Basin since 1932 (Lyons, 1997), after the 1905 discovery of the Big Run Field in Wetzel County, West Virginia (Hunt and Steele, 1991). Coalbed methane production development in the Northern Appalachian Basin has lagged, however, due to insufficient reservoir knowledge, inadequate well completion techniques, and coalbed gas ownership issues revolving around whether the gas is owned by the mineral owner or the oil and gas owner (Zebrowitz et al., 1991). Annual coalbed methane production stood at 1.41 billion cubic feet in 2000 (GTI, 2002). As of October 2002, 185 coalbed methane wells were producing coalbed methane in Pennsylvania (Pennsylvania Department of Conservation and Natural Resources, 2002). Discharge of produced waters has also proven to be problematic (Lyons, 1997) for coalbed methane field operators in the Northern Appalachian Coal Basin.

Some operators in the Northern Appalachian Coal Basin and several test projects are discussed below. As of 1993, O’Brien Methane Production, Inc. had at least 20 wells in southern Indiana County, Pennsylvania (Quarterly Review, 1993). They received a water treatment and discharge permit that allowed O'Brien to discharge produced water into Blacklick Creek. The wells in O’Brien’s field were hydraulically fractured with water and sand. Nitrogen was being contemplated for future fracturing. O’Brien's operations have since been assumed by Belden and Blake. BTI Energy, Inc. also had a few coalbed wells in northern Fayette County, Pennsylvania. Two were completed in 1993 and the firm held permits for eight additional wells.

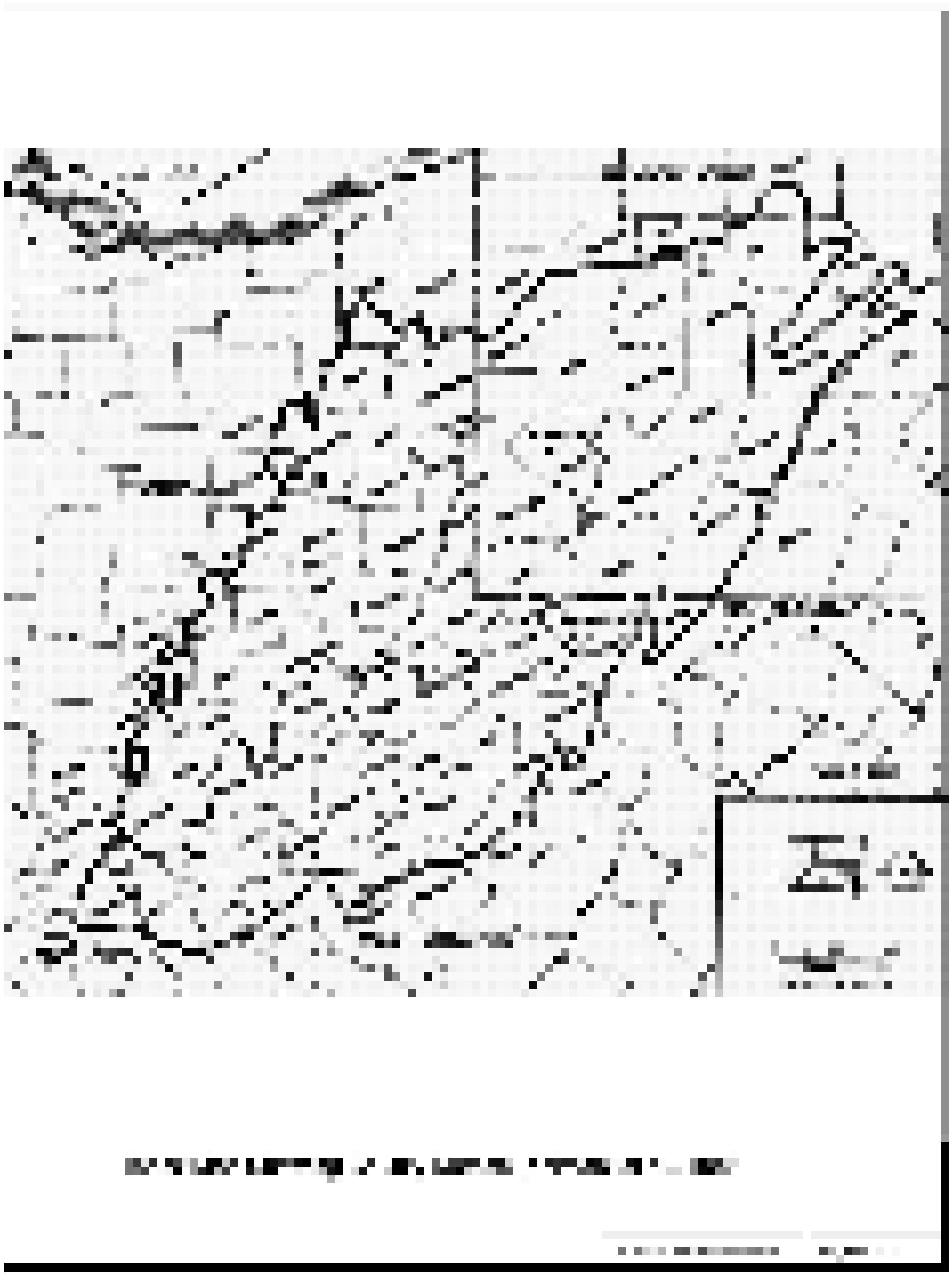
Other projects in the basin included the Lykes/Emerald Mines Project of the United States Bureau of Mines (USBM) and the Penn State University/Carnegie Natural Gas/U.S. Steel Wells Project, both in Greene County, Pennsylvania. Depths to the top of the Pittsburgh coals in Greene County range from 800 to 1,200 feet below ground surface (Kelafant et al., 1988). Hydraulic fracturing fluids included water and sand, and nitrogen foam and sand (Hunt and Steele, 1991). The Christopher Coal Company/Spindler Wells Project, which took place from 1952 to 1959, fractured one well with 12 quarts of nitroglycerin (Hunt and Steele, 1991). In the Vesta Mines Project of Washington County, Pennsylvania, the USBM used gelled water and sand to complete five wells in the Pittsburgh Seam (Hunt and Steele, 1991).

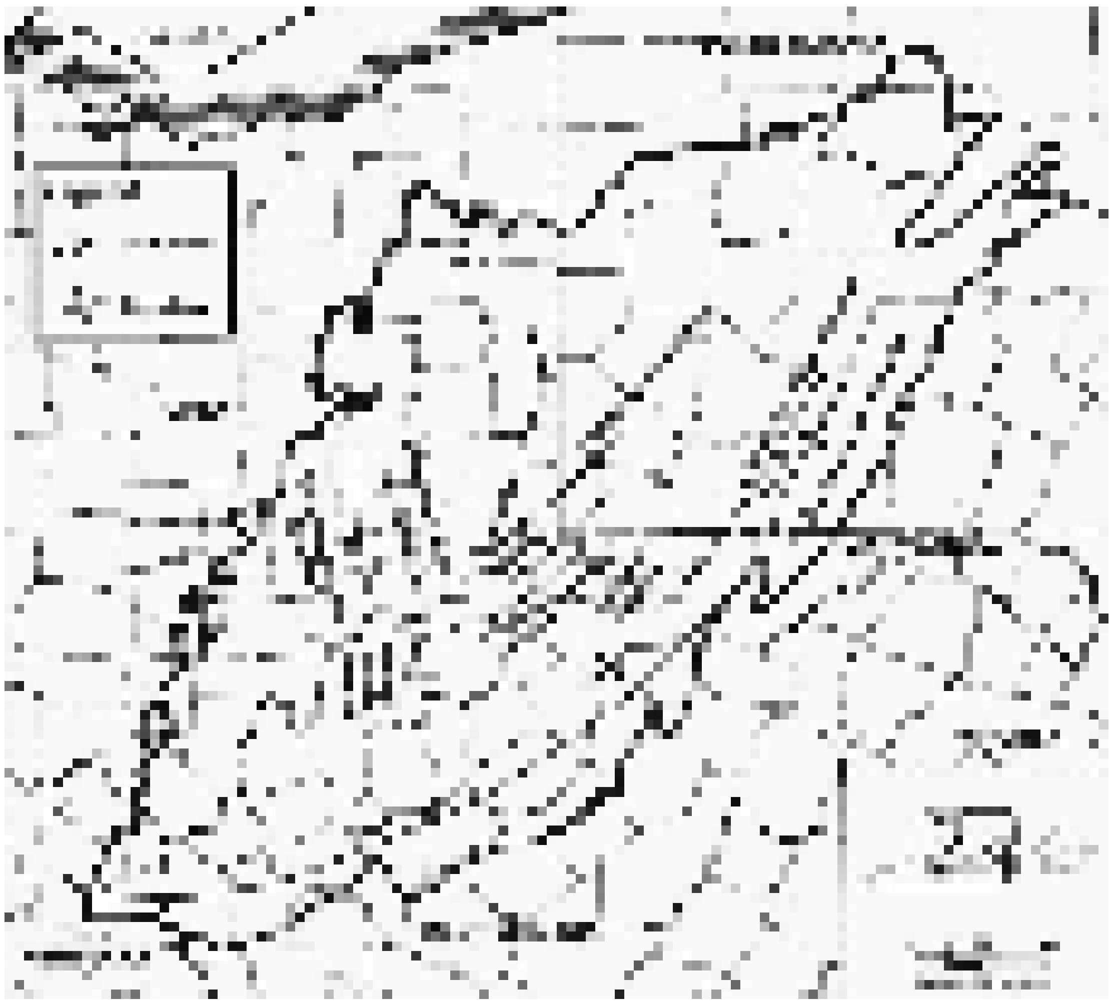
Within the State of Pennsylvania, there have been complaints of methane migrating into water supplies (Markowski, 2001). According to the Pennsylvania Department of Conservation and Natural Resources (2002), none of these complaints were linked specifically to hydraulic fracturing of coalbed methane wells. During a telephone interview (Markowski, 2001), Ms. Markowski stated methane contamination is due to the fact that many coalbed methane wells in southwestern Pennsylvania are completed in abandoned mine shafts. A puncture in the roof of the mineshaft provides a migration pathway for methane into overlying groundwater. These wells are known as gob wells, and are not usually hydraulically fractured or stimulated.

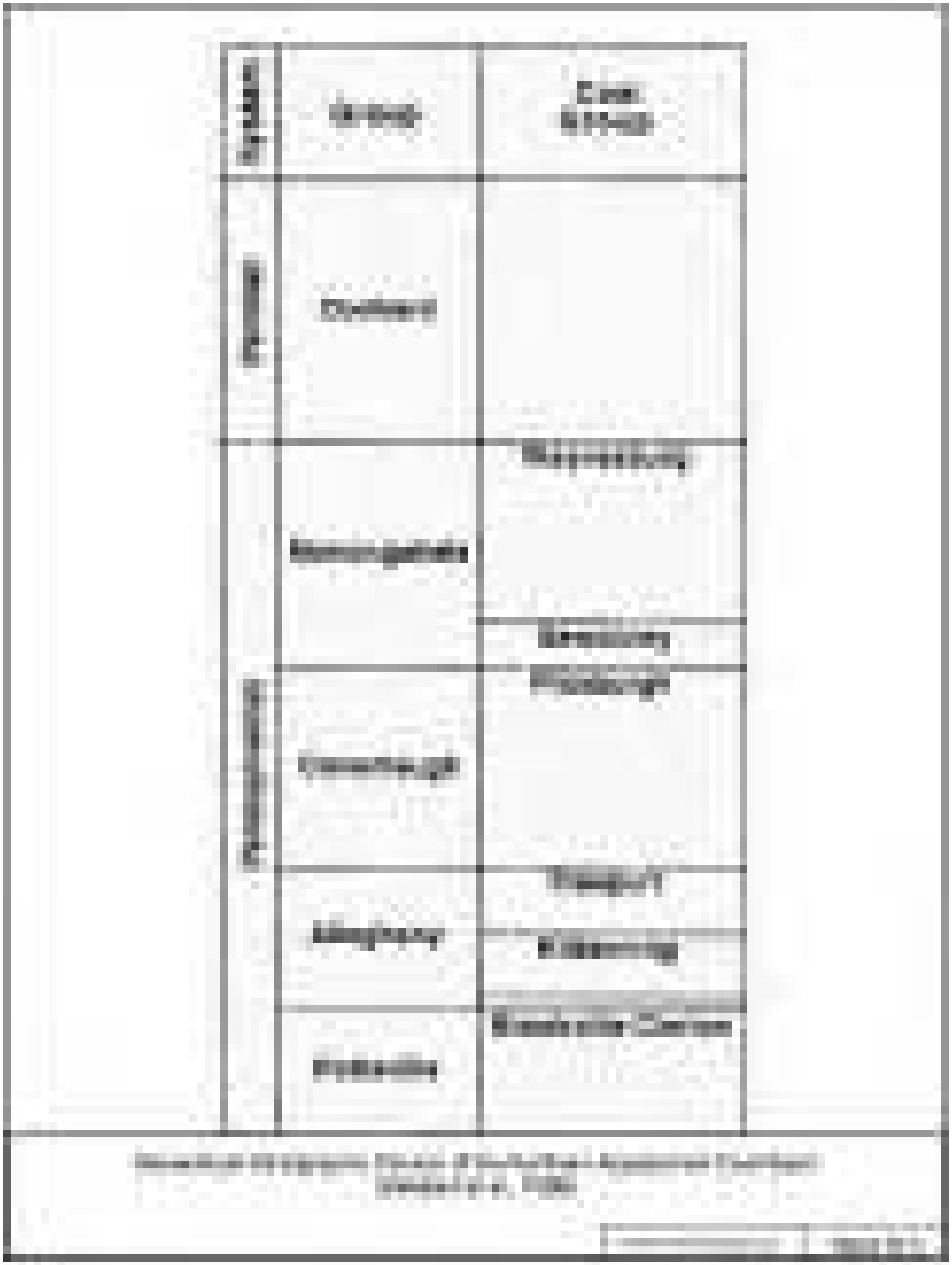
7.4 Summary

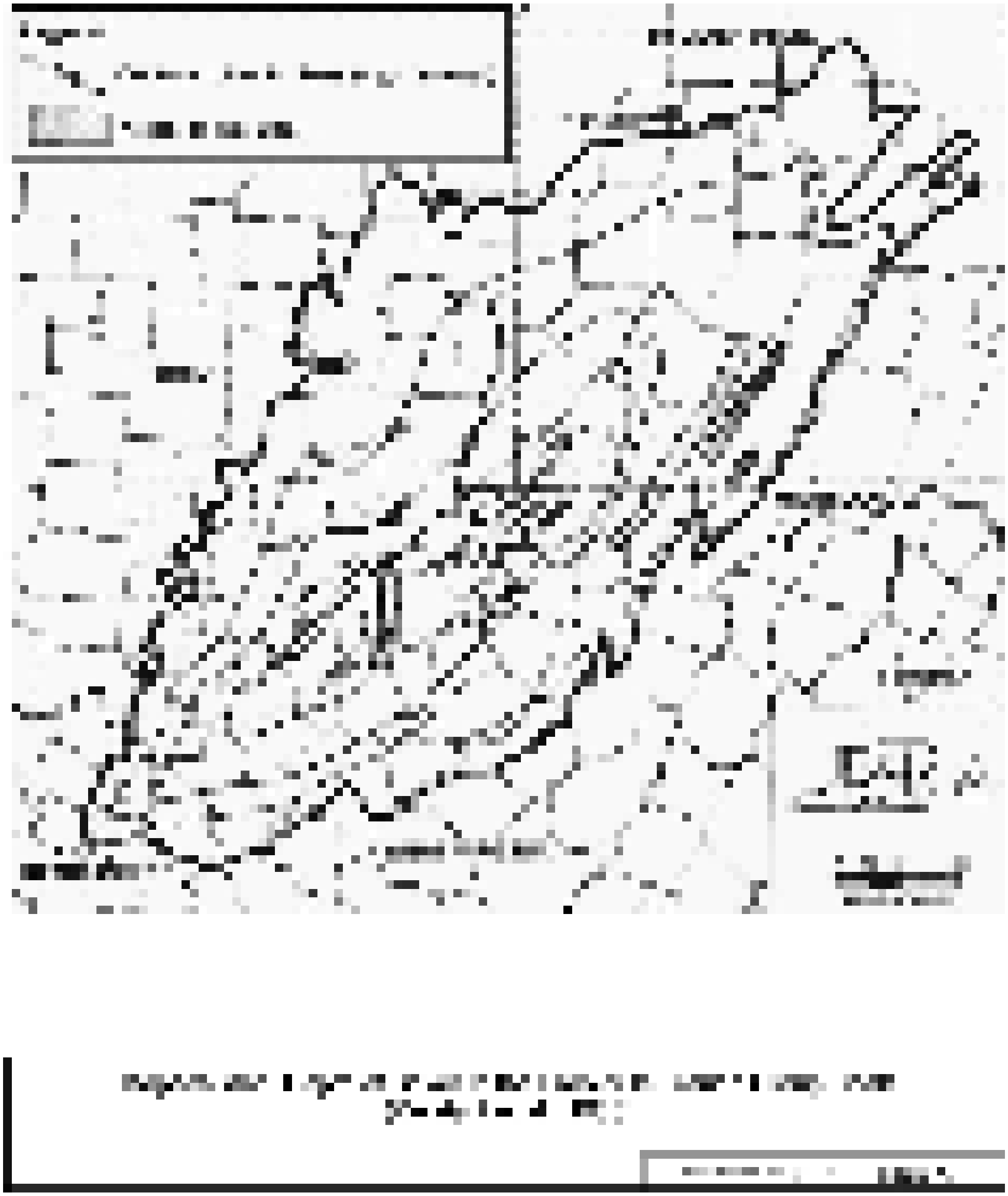
Based on available information, coal seams with methane production potential are located within USDWs throughout the Northern Appalachian Coal Basin, and hydraulic fracturing takes place in this basin. Because most of the coal strata dip, a well's location within the basin determines whether it is within a USDW, and whether the potential for impact exists. For example, in the Pittsburgh Coal Zone in Pennsylvania, the depth to the top of this coal zone varies from outcrop to about 1,200 feet in the very southwestern corner of the state. The approximate depth to the bottom of the USDW is 450 feet. Therefore, production wells operating down to approximately 500 feet could potentially be hydraulically connected to the USDW. However, those wells operating at depths greater than 900 feet would probably not be hydraulically connected to the USDW, unless a fracture extending beyond the coal layers to the shallower aquifer was to occur.

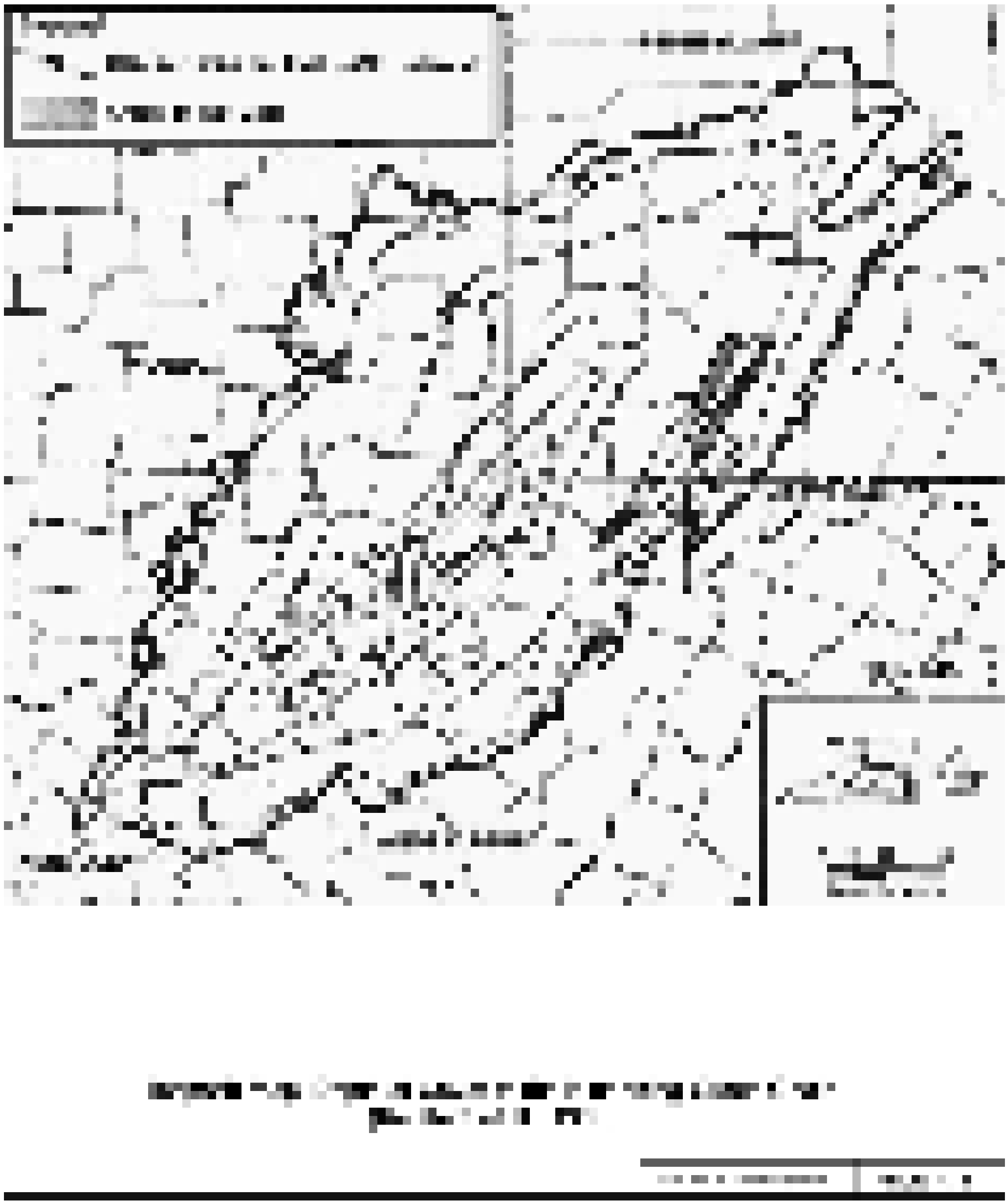
Milici (2002) indicated that the Pittsburgh Coal in Pennsylvania is mined out along its outcrop and the remaining coal resources are deeper (> 450 feet) in the basin. While this situation would greatly minimize the possibility of water quality impacts for this coal zone in Pennsylvania, the potential for contamination from the Pittsburgh coalbeds in other states within the basin still exists (see Table A7-1).

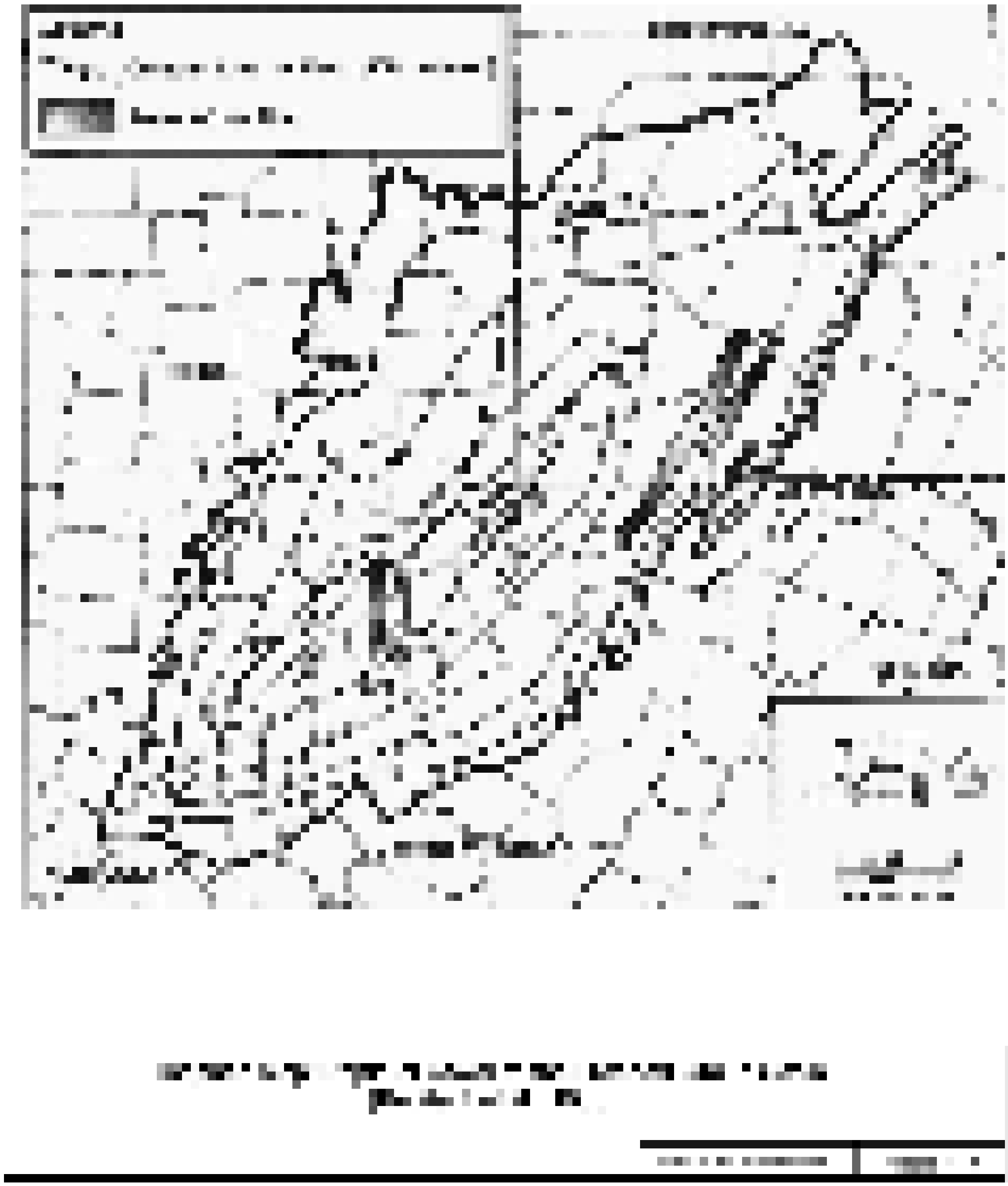


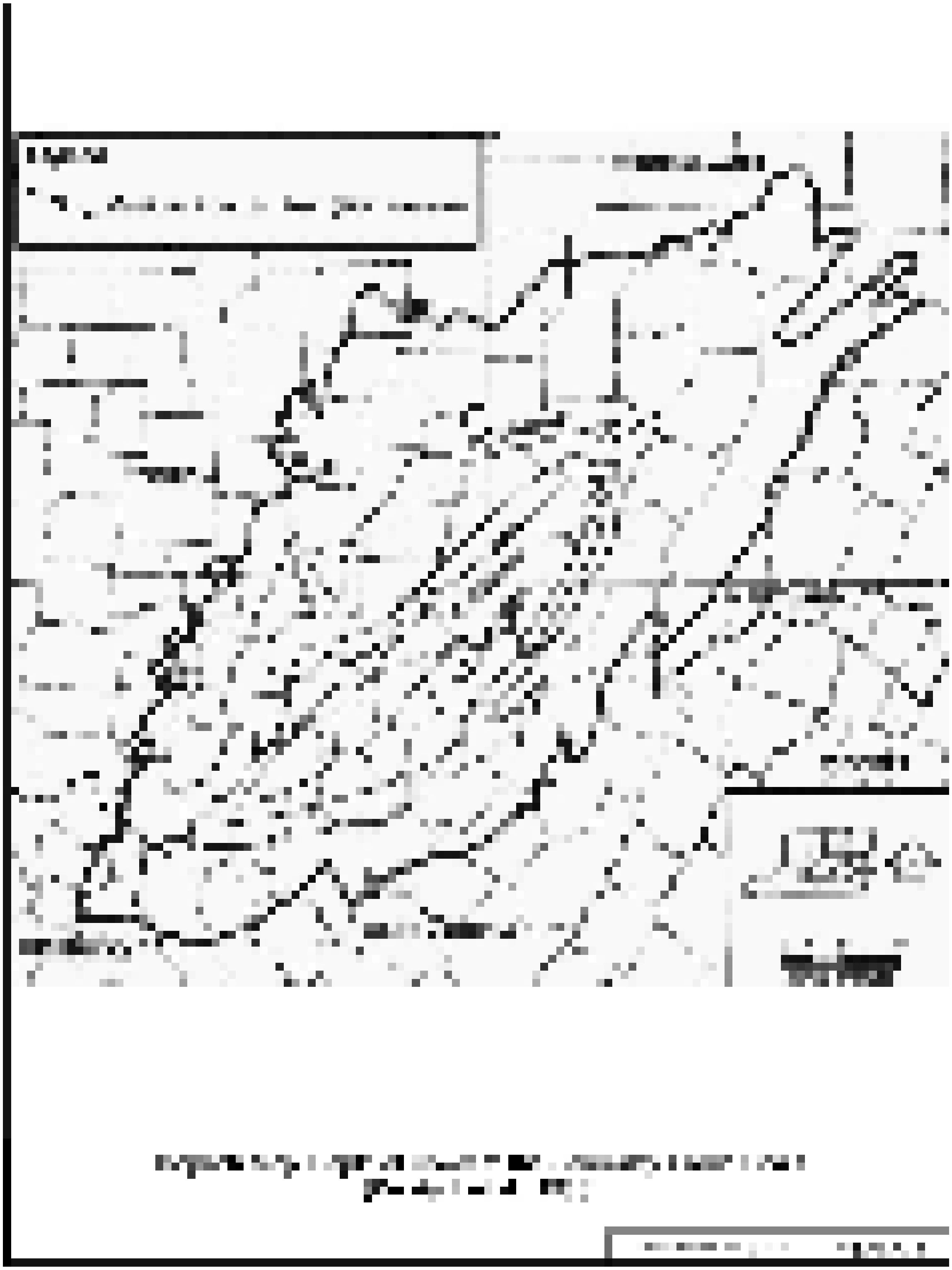


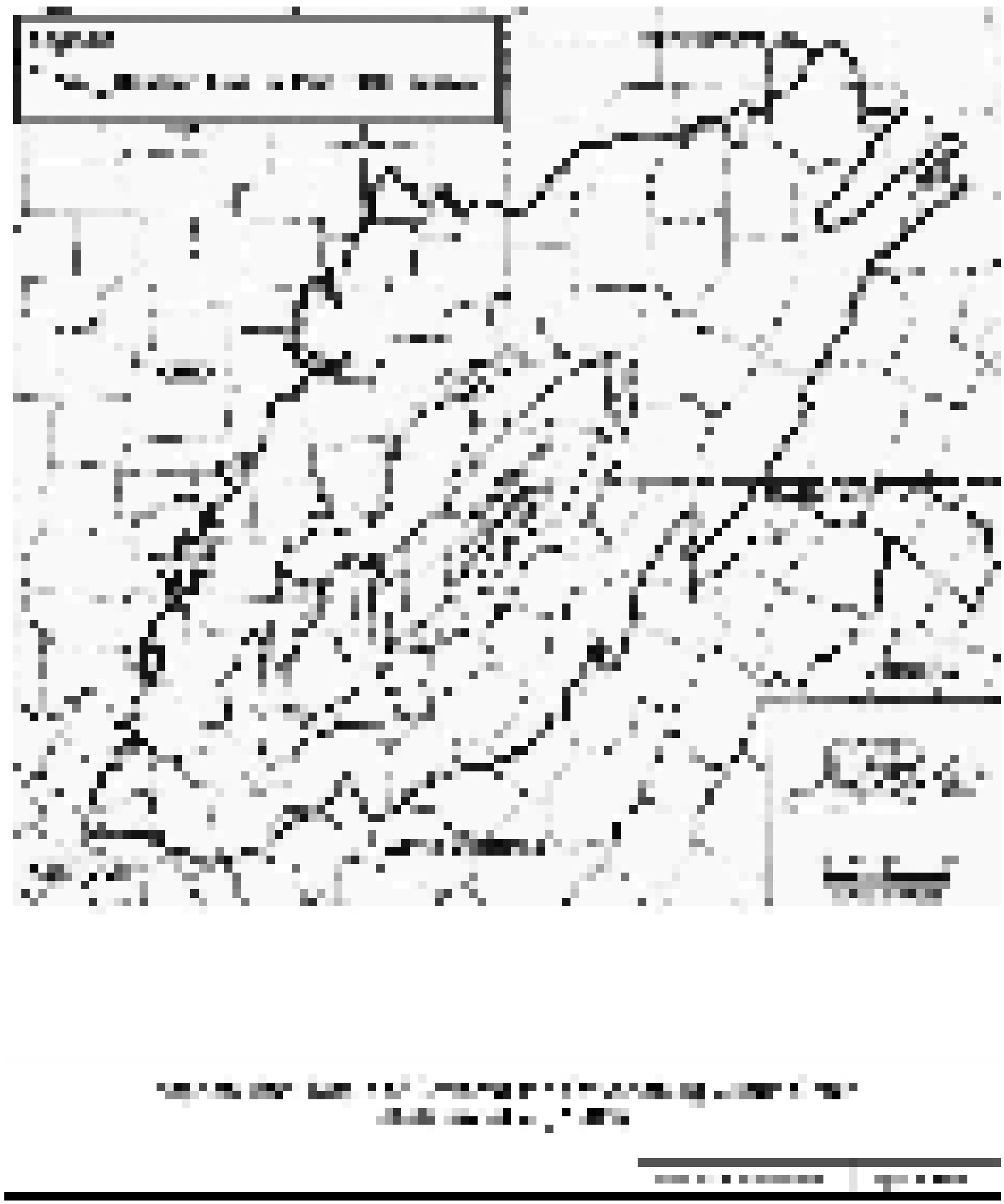












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Attachment 8
The Western Interior Coal Region

The Western Interior Coal Region comprises three coal basins, the Arkoma, the Cherokee, and the Forest City Basins, and encompasses portions of six states: Arkansas, Oklahoma, Kansas, Missouri, Nebraska, and Iowa (Figure A8-1). The Arkoma Basin covers about 13,500 square miles in Arkansas and Oklahoma, with an estimated 1.58 to 3.55 trillion cubic feet (Tcf) of gas reserves, primarily in the Hartshorne coals (Quarterly Review, 1993).

The Cherokee Basin is part of the Cherokee Platform Province, which covers approximately 26,500 square miles (Charpentier, 1995) in Oklahoma, Kansas, and Missouri. The basin contains an estimated 1.38 million cubic feet of gas per square mile (Stoeckenger, 1990) in the targeted Mulky, Weir-Pittsburg, and Riverton coal seams of the Cherokee Group (Quarterly Review, 1993). In total, the basin contains approximately 36.6 billion cubic feet (Bcf) of gas. However, the Petroleum Technology Transfer Council (1999) indicates that there are nearly 10 Tcf of gas in eastern Kansas alone. The Forest City Basin covers about 47,000 square miles (Quarterly Review, 1993) in Iowa, Kansas, Missouri, and Nebraska, and contains an estimated 1 Tcf of gas (Nelson, 1999). For the entire region, coalbed methane production was 6.5 Bcf in 2000 (Gas Technology Institute (GTI), 2002).

8.1 Basin Coals

The Arkoma Basin is the southernmost of the three basins comprising the Western Interior Coal Region, and is bounded structurally by the Ozark Dome to the north, the Central Oklahoma Platform and Seminole Uplift on the west, and the Ouachita Overthrust Belt to the south (Quarterly Review, 1993). Middle Pennsylvanian coalbeds occur within the Hartshorne and McAlester Formations (Figure A8-2), as well as the Savanna and Boggy Formations (Quarterly Review, 1993).

The Cherokee Basin is the central basin of the Western Interior Coal Region, and is bounded on the east and southeast by the Ozark Dome, on the west by the Nehama Uplift, and on the north by the Bourbon Arch (Quarterly Review, 1993). Principal coals occur in the Krebs and Cabaniss Formations of the middle Pennsylvanian Cherokee Group (Figure A8-3).

The Forest City Basin (Figure A8-4), the northernmost basin of the Western Interior Coal Region, is a shallow cratonic depression bounded by the Nemaha Ridge to the west, the Thurman-Redfield structural zone to the north, the Mississippi River Arch to the east, and the Bourbon Arch to the south (Bostic et al., 1993). Methane-bearing coals occur in the middle Pennsylvanian Cherokee and Marmaton Groups, with the Cherokee Group being of primary interest (Tedesco, 1992).

8.1.1 Arkoma Basin Coals

The Hartshorne coals of the Hartshorne Formation are the most important for coalbed methane production in the Arkoma Basin. Their depths range from 600 to 2,300 feet in two productive areas in southeastern Oklahoma (Quarterly Review, 1993). Iannacchione and Puglio (1979) estimated that 58 percent of the coalbed methane in the Hartshorne coals in southeastern Oklahoma occurs at 500- to 1,000-foot depths. These coals can reach depths of greater than 5,000 feet, and are three to nine feet thick (Quarterly Review, 1993). Depths to the top of the Hartshorne coal in southeastern Oklahoma range from 380 to 1,540 feet (Friedman, 1982). As of March 2000, there were 377 coalbed methane wells in eastern Oklahoma, ranging in depth from 589 to 3,726 feet (Oklahoma Geological Survey, 2001).

8.1.2 Cherokee Basin Coals

The primary coal seams targeted by operators in Kansas are the Riverton Coal of the Krebs Formation and the Weir-Pittsburg and Mulky coals of the Cabaniss Formation (Quarterly Review, 1993). The Riverton and Weir-Pittsburg seams are about 3 to 5 feet thick and range from 800 to 1,200 feet deep (Quarterly Review, 1993). The Mulky Coal, which ranges up to 2 feet thick, occurs at depths of 600 to 1,000 feet (Quarterly Review, 1993).

8.1.3 Forest City Basin Coals

Individual coal seams in the Cherokee Group in the Forest City Basin range from a few inches to about 4 feet thick, with some seams up to 6 feet thick (Brady, 2002; Smith, 2002). Cumulative maximum coal thickness within the Cherokee Group is about 25 to 30 feet (Brady, 2002; Smith, 2002). Depths to the top of the Cherokee Group coals range from surface exposures in the shallower portion of the basin in southeastern Iowa, to about 1,220 feet in the deeper part of the basin, in northeastern Kansas (Bostic et al., 1993). At one location in Nebraska, the depth to the Cherokee Group is about 1,396 feet, and the base is at a depth of 2,096 feet (Condra and Reed, 1959). Maximum thickness of the Cherokee and Marmaton Groups is about 800 feet in the southeastern tip of Nebraska (Burchett, unpublished paper).

8.2 Basin Hydrology and USDW Identification

8.2.1 Arkoma Basin Hydrology and USDW Identification

In Arkansas, the Arkoma Basin falls within the Interior Highlands physiographic province (Figure A8-5). According to the National Water Summary (1984), there is no principal aquifer in this area, only small alluvial aquifers bounding the Arkansas River (Figure A8-5). In these alluvial aquifers, water wells typically penetrate to depths of 100 to 150 feet, and common well yields are in the order of 1,000 to 2,000 gallons of water per minute (National Water Summary, 1984). In Oklahoma, the Arkoma Basin is contained within the Ouachita and Central Lowland physiographic province (Figure A8-6). Much like in Arkansas, there are no principal aquifers in this portion of the state, but there are smaller alluvium and terrace deposits along the Arkansas, North Canadian, and Canadian Rivers (National Water Summary, 1984) that serve as aquifers (Figure A8-6). Marcher (1969) also identifies these smaller deposits as the most favorable for groundwater supplies. Water well depths in the alluvium and terrace deposits of the Arkansas River in Oklahoma typically range from 50 to 100 feet (National Water Summary, 1984). Water well production rates in all three aquifers commonly range from 100 to 600 gallons of water per minute in alluvium, and 50 to 300 gallons of water per minute in terrace deposits (National Water Summary, 1984).

Bill Prior, a geologist with the Arkansas Geological Commission, stated that within Arkansas, the Arkoma Basin was in the Arkansas River Physiographic Province, which lacks a true aquifer. Most of the rocks within this physiographic province are tight sandstones and shales, and most communities within the province use surface water supplies (Prior, 2001). Doug Hansen of the Arkansas Geological Commission said that there were a few scattered bedrock wells within the Arkoma Basin (Hansen, 2001). Total dissolved solids (TDS) levels in the McAlester Formation in Arkansas (which contains the Hartshorne coals; Potts, 1987) range between 55 to 534 milligrams per liter (mg/L) at depths ranging from 32.4 to 190 feet below land surface (Cordova, 1963). The base of fresh water in the area is about 500 to 2,000 feet below ground surface (Cordova, 1963). However, Cordova (1963) does not define “fresh water;” therefore, it is difficult to determine if the depths reported by Cordova coincide with the base of an underground source of drinking water (USDW).

Water quality test results from the targeted Hartshorne seam in Oklahoma have shown the water to be highly saline (Quarterly Review, 1993). Ken Luza, a geologist with the Oklahoma Geological Survey, stated that a hydrologic atlas prepared by the Oklahoma Geological Survey delineated a 5,000 mg/L TDS water quality contour line in a portion of the state, including the Arkoma Basin (Marcher, 1969; Marcher and Bingham, 1971). Maps such as these atlas maps show that, based on water quality and rock type, very little of the area falls within a zone “most favorable for groundwater supplies” or “moderately favorable for groundwater supplies.” Most of the area falls within a zone designated as “least favorable for groundwater supplies” (Cardott, 2001). Pam Hudson, Manager of the

Geologic Section of the Oklahoma Corporation Commission, stated that the Commission has a series of maps, one for each county in Oklahoma, showing the depth to the 10,000 mg/L TDS line (Hudson, 2001). The water quality criterion for a USDW is a TDS level of less than 10,000 mg/L. The Oklahoma Corporation Commission maps are used to assist drillers in complying with state regulations that require oil and gas wells to be cased through USDWs.

The following table contains information concerning the relative location of potential USDWs and methane-bearing coalbeds in the Arkoma Basin.

Table A8-1 Relative Locations of USDWs and Potential Methane-Bearing Coalbeds, Arkoma Basin

Arkoma Coal Basin, States and Coal Group	**Arkansas**		**Oklahoma**	
	Depth to top of Coal [1] (ft)	Depth to base of Fresh Water [2,3] (ft)	Depth to top of Coal [1] (ft)	Depth to base of USDW [4] (ft)
Hartshorne Coals	0 to < 4,500	500 to 2000	> ~1000	< ~900

[1] Andrews et al., 1998
[2] Note: The base of "fresh water" is not the base of the USDW (depth to the base of the USDW is unknown or not available). Fresh water is within the USDW and the base of fresh water is above the base of the USDW. Cordova (1963) does not define "fresh water."
[3] Cordova, 1963
[4] Oklahoma Corporation Commission Depth to Base of Treatable Water Map Series (2001)

Based on Table A8-1, it can be determined that in Arkansas, there is a possibility for the Hartshorne Coals to be located within a USDW, allowing the potential for impacts. The potential for impacts from fracturing coalbeds below the USDW is not known. Cordova (1963) does not specify the TDS level used to determine the depth of the base of fresh water in the Arkansas Valley region; he merely states that it is the depth to salt water, and he does not provide a definition of "salt water." The position of a coalbed methane well within the basin would ultimately determine if coals and USDWs coincide, as the Hartshorne Coals are typically shallower on basin margins (Andrews et al., 1998) and progressively increase in depth toward the basin's center (where they are potentially too deep to be located within a USDW).

8.2.2 Cherokee Basin Hydrology and USDW Identification

The Cherokee Basin underlies parts of the States of Kansas, Missouri, and Oklahoma. In Kansas, the Cherokee Basin is part of the Central Lowlands and Ozark Plateaus physiographic provinces (Figure A8-7). While the majority of this area does not contain a principal aquifer, the Ozark and Douglas aquifers (Figure A8-7) are contained in the basin (National Water Summary, 1984). The confined Ozark Aquifer, composed of weathered and sandy dolomites, typically contains water wells that extend from 500 to

1,800 feet in depth, commonly yielding 30 to 150 gallons of water per minute (National Water Summary, 1984). The usually unconfined Douglas Aquifer is channel sandstone of Pennsylvanian Age (National Water Summary, 1984). Water wells are usually 5 to 400 feet deep in this aquifer and typically produce 10 to 40 gallons of water per minute (National Water Summary, 1984).

In Missouri, only a very small portion of the basin falls within the Osage Plains area of the Central Lowlands physiographic province (Figure A8-8). The principal aquifers in this portion of Missouri are the Ozark and Pennsylvanian-Mississippian age aquifers (National Water Summary, 1984) (Figure A8-8). Water well depths in the Ozark Aquifer typically range from 200 to 1,700 feet, and those in the Pennsylvanian-Mississippian age aquifers typically range from 100 to 400 feet in depth (National Water Summary, 1984). Common well yields are 15 to 700 gallons of water per minute and 1 to 15 gallons of water per minute in the Ozark and Pennsylvanian-Mississippian aquifers, respectively (National Water Summary, 1984). Only a very small portion of the Cherokee Basin, bounded from the Forest City Basin to its north by the Bourbon Arch, falls within the State of Missouri (Figure A8-9). Jim Vandike, Chief of Missouri's Water Resources Branch at the Missouri Geological Survey, stated that only two public water supplies obtain water from Pennsylvanian strata, and those wells were outside of the Cherokee Basin (Vandike, 2001).

In Oklahoma, the Cherokee Basin lies within the Central Lowland physiographic province (Figure A8-6). In addition to the alluvium and terrace deposit aquifers previously discussed in the Arkoma Basin aquifer descriptions, this area also contains the Garber-Wellington and Vamoosa-Ada Aquifers (Figure A8-6), which are unconfined to confined sandstone with shale and siltstone aquifers (National Water Summary, 1984). The Vamoosa-Ada Aquifer contains some conglomerate aquifers as well. Water well depths in these two aquifers usually range from 100 to 900 feet, and wells typically produce from 100 to 300 gallons of water per minute (National Water Summary, 1984). At least half of the area of this basin in Oklahoma does not contain a principal aquifer (National Water Summary, 1984).

In Kansas, Al Macfarlane, of the Kansas Geological Survey, stated that the Ozark Aquifer was located in the Cherokee Basin in Kansas (Macfarlane, 2001). An Ozark Aquifer Extent map indicates that the "usable" part of the aquifer (defined as having less than 10,000 mg/L of TDS per Macfarlane; no definition of "usable" is provided by the map) covers the three southeastern-most counties (Bourbon, Crawford, and Cherokee) of the state (Figure A8-7) and parts of the adjacent four counties (Linn, Allen, Neosho, and Labette) (DASC Ozark Aquifer Extent Map, 2001c). Because the land surface elevation in that portion of the state is roughly 850 feet above sea level (DASC Kansas Elevation Map, 2001b) and the elevation of the base of the Ozark Aquifer is roughly 900 feet below sea level (Ozark Aquifer Base Map, 2001c), the base of the Ozark aquifer is roughly 1,750 feet below ground surface. Groundwater samples taken from lower Paleozoic aquifers in Kansas show TDS levels ranging from <500 to 5,000 mg/L (Figure A8-10) (Macfarlane and Hathaway, 1987), well within the range for a USDW.

Table A8-2 contains information concerning the relative location of potential USDWs and methane-bearing coalbeds in the Cherokee Basin. The table shows that all or part of the targeted coal seams could be coincident with a USDW, allowing the potential for impacts. Most past coalbed methane production activity within the Cherokee Basin took place in Kansas (Quarterly Review, 1993). However, coalbed methane production activity within the Cherokee Basin in Oklahoma has increased markedly in recent years (Hudson, 2001).

Table A8-2 Relative Locations of USDWs and Potential Methane-Bearing Coalbeds, Cherokee Basin

Coal Group	**Kansas**		**Missouri**		**Oklahoma**	
	Depth to top of Coal [1] (ft)	Depth to base of Fresh Water (USDW) [2] (ft)	Depth to top of Coal [1] (ft)	Depth to base of Fresh Water [3] (ft)	Depth to top of Coal [1] (ft)	Depth to base of Fresh Water (ft)
Mulky	600 to 1000	~ 1750	600 to 1000	N/A[4]	600 to 1000	N/A[4]
Weir-Pittsburg	800 to 1200		800 to 1200		800 to 1200	
Riverton	800 to 1200		800 to 1200		800 to 1200	

[1] Quarterly Review, 1993
[2] Ozark Aquifer extent and base, and Kansas elevation maps from the Kansas Data Access and Support Center (DASC) 2001b above
[3] Missouri's Geological Survey, Water Resources Branch, claims no water supplies in these strata
[4] Not Available

8.2.3 Forest City Basin USDW Identification

The Forest City Basin includes parts of the States of Iowa, Kansas, Missouri, and Nebraska. In Iowa, the Forest City Basin lies within the Southern Iowa Drift Plain physiographic province (Figure A8-11). The most productive aquifer in this area is the dolomite and sandstone Jordan Aquifer (Figure A8-11). Wells in this aquifer commonly range in depth from 300 to 2,000 feet (some are as deep as 3,000 feet) and usually produce 100 to 1,000 gallons of water per minute (National Water Summary, 1984). This aquifer usually contains in excess of 1,500 mg/L TDS in the southern portion of the state (National Water Summary, 1984). Other aquifers used at various locations in the basin are found in the Silurian-Devonian age and in the Mississippian-age strata (Figure A8-11). Water wells in these aquifers range from 150 to 750 feet deep with variable production (Howes, 2002). Also contained within this basin in Iowa is a portion of the confined, poorly cemented sandstone Dakota aquifer (National Water Summary, 1984) (Figure A8-11). Water wells in this aquifer are typically 100 to 600 feet in depth, and commonly produce 100 to 250 gallons of water per minute (National Water Summary, 1984). An Iowa Division of Natural Resources Geological Survey Bureau geologist, Mary Howes, said that few towns in Iowa use Pennsylvanian strata for water, as they typically contain high concentrations of sulfate and TDSs (Howes, 2001). Most community water supplies in the southern portion of Iowa use surface water and shallow alluvial aquifers as drinking water sources, and there are a few wells in fractured bedrock. Private water supplies typically are derived from seepage wells, shallow bedrock wells, or purchased from a public supply (Howes, 2002).

In Kansas, the basin is located in the Lowlands physiographic province (Figure A8-7), and only the northeastern corner of the state falls within the Forest City Basin boundary. In addition to the Douglas Aquifers described above in the Cherokee Basin Aquifer descriptions, this portion of the Forest City Basin in Kansas also contains a glacial drift aquifer (is this and some alluvial aquifers adjacent to the Kansas River (National Water Summary, 1984) (Figure A8-7). In the glacial drift, wells are typically 10 to 300 feet in depth and usually produce 10 to 100 gallons of water per minute (National Water Summary, 1984). Wells in the alluvium are usually 10 to 150 feet deep and typically produce 10 to 500 gallons of water per minute (National Water Summary, 1984). The glacial drift aquifer's base varies from about 850 to 1,300 feet above sea level (DASC, Glacial Drift Base Map, 2001a). Since the elevation of the land surface in this portion of Kansas is roughly between 1,000 and 1,400 feet above sea level (DASC, Kansas Elevation Map, 2001b), the aquifer appears to extend only to an approximate maximum depth of 150 feet below the ground surface.

In Missouri, the basin lies within the Central Lowland physiographic province (Figure A8-8). The principal aquifer in this area is a glacial-drift aquifer (Figure A8-8). In this aquifer, water wells are typically 100 to 250 feet in depth and produce 5 to 200 gallons of water per minute. In addition to this aquifer, alluvial deposits along the Missouri River are also developed for water (National Water Summary, 1984)(Figure A8-8). Well depths in the alluvium usually range from 80 to 100 feet in depth, and the wells typically produce 100 to 1,000 gallons of water per minute (National Water Summary, 1984). Two public supply wells in Cass County, Missouri, extract water from Pennsylvanian strata for the town of East Lynn. A map of groundwater quality within Paleozoic aquifers of Missouri (Figure A8-12) shows that within the Forest City Basin, water quality ranges from about 500 mg/L TDS to 40,000 mg/L TDS in deeper portions of the basin (Missouri Division of Geological Survey and Water Resources, 1967). A 10,000 mg/L TDS boundary line delineated in the Mississippian aquifers of Missouri (located directly below Pennsylvanian-age strata) includes portions of Cass, Jackson, Lafayette, Carroll, Saline, Ray, Clay, Caldwell, Clinton, and Platte Counties (Netzler, 1982) (Figure A8-8).

Only the southeastern tip of Nebraska (primarily Richardson County) falls within the limits of the Forest City Basin. The principal aquifers in this area are undifferentiated aquifers in Paleozoic-age rocks (National Water Summary, 1984) (Figure A8-13). Locally overlain by saturated Quaternary-age sand and gravel deposits, wells within this area are commonly 30 to 2,200 feet in depth, and produce about 10 to 200 gallons of water per minute. TDS levels in the water can be as high as 6,000 mg/L, but are usually less than 1,500 mg/L (National Water Summary, 1984). The Ground Water Atlas of Nebraska (Flowerday et al., 1998) indicates that Richardson County is within the Southeastern Nebraska Glacial Drift rock unit. The thickness of the aquifer in Richardson County is less than 100 feet and the depth to water is 30 to 200 feet. The information in the Ground Water Atlas of Nebraska (Flowerday et al., 1998) appears to be in conflict with the data presented by the U.S. Geological Survey in the National Water Summary (1984). Matt Jokel of the Nebraska Conservation and Survey Division said it is very difficult to obtain water in this portion of the state, and most people use valley fill materials and paleochannels as water supply sources. He also believes that the coal resources, which could possibly be used for methane extraction, are probably too deep to be located coincident with the shallow water supplies in the area (Jokel, 2001).

Table A8-3 contains information concerning the relative location of potential USDWs and potential methane-bearing coalbeds in the Forest City Basin.

Table A8-3 Relative Locations of USDWs and Potential Methane-Bearing Coalbeds, Forest City Basin

	Iowa		Kansas		Missouri		Nebraska	
Coal Group	Depth to top of Coal [1] (ft)	Depth to base of fresh water[3] (ft)	Depth to top of Coal [1] (ft)	Depth to base of fresh water [4] (ft)	Depth to top of Coal [1] (ft)	Depth to base of fresh water [5] (ft)	Depth to top of Coal[1,6] (ft)	Depth to base of fresh water [2,7] (ft)
Cherokee Group	0 to >230	N/A[8]	720 to 1220	~ 150	300 to 1100	N/A[8]	1220 to 1396	129 to 299

[1] Bostic et al., 1993
[2] Note: The base of "fresh water" is not the base of the USDW. Fresh water is within the USDW and the base of fresh water is above the base of the USDW.
[3] Howes, Iowa Geological Survey Bureau (2001) believes water quality data may be available to define this depth
[4] Glacial Drift base and Kansas elevation maps from the Kansas Data Access and Support Center (DASC), 2001b
[5] Maps (Netzler, 1982) sent by Missouri show the extent of aquifers containing less than 10,000 mg/L of TDS, but not depths
[6] Condra and Reed, 1959
[7] The Groundwater Atlas of Nebraska, (Flowerday et al., 1998)
[8] Not Available

Presently, there does not appear to be a USDW located at the same depth as the coals of the Cherokee Group in the Forest City Basin. However, very little is known about the coal resources of this basin (Quarterly Review, 1993). Further research is required to delineate the possible link between coalbed methane resources and USDWs in the Forest City Basin.

8.3 Coalbed Methane Production Activity

GTI places total coalbed gas production in the Western Interior Coal region at 6.5 Bcf for the year 2000 (GTI, 2002).

8.3.1 Arkoma Basin Production Activity

In 1989, Bear Production Company became the first company to target coalbed methane production from the Hartshorne Coals of the Arkoma Basin in Haskell County, Oklahoma (Quarterly Review, 1993). As of 1993, Bear Production had 38 wells in operation, Aztec Energy Corporation had 19 wells, and Redwine Resources, Inc. had 40 wells in the Arkoma Basin (Quarterly Review, 1993).

As of 1993, Bear Production was not fracturing its wells, but rather completing them as open holes without perforated casings (Quarterly Review, 1993). However, other production companies were fracturing their wells for methane production. Before 1992, water, linear gel, acid, and nitrogen foam fracturing fluids were used, with most operators using foam with small sand volumes (35,000 to 60,000 lbs) (Quarterly Review, 1993). In 1993, slick water fracturing fluids containing no proppant were becoming more common (Quarterly Review, 1993). Well fracturing data from 36 wells in the Spiro Southeast Field of LeFlore County, Oklahoma show that either water or nitrogen foam was the base fracturing fluid used to carry sand proppant into coal cleats (Andrews et al., 1998). Fracturing continues in the Arkoma Basin today, at least in Oklahoma, where undisclosed amounts of initial water production are "frac" waters introduced during fracture stimulation (Cardott, 2001). Both Wendell (2001) and Marshall (2001) outline current hydraulic fracturing practices within the Arkoma Basin, and Wendell (2001) includes acids, benzene, xylene, toluene, gasoline, diesel, solvents, bleach, and surfactants as detrimental hydraulic fracturing substances in his "lessons learned" category.

A search of the Oklahoma Coal Database, updated on January 17, 2001, indicated that over 360 coalbed methane wells had been completed in Haskell, Le Flore, and Pittsburg counties alone, targeting the Hartshorne, McAlester, and Savanna coals. Additional operators in the Arkoma Basin today include Continental Resources, SJM Inc., Brower O & G, Mannix Oil, and OGP Operating (Oklahoma Coal Database, 2001).

Apparently there is little to no coalbed methane activity in the Arkoma Basin in Arkansas, based on the Arkansas Geological Commission's Web site, which states,

"…there exists the potential for coalbed methane production in this area of the state" (Arkansas Geological Commission, 2001). The low coalbed methane activity in this Basin is further confirmed by Andrews et al. (1998), which outlines Arkansas' restrictive field-spacing policy from the 1930s of only one well per 640-acre section for each producing zone in the Hartshorne. This policy effectively made exploration uneconomical. A change in field-spacing rules in 1995 has stimulated new interest among independent producers in Arkansas to develop methane from the Hartshorne coals (Andrews et al., 1998).

8.3.2 Cherokee Basin Production Activity

In the Cherokee Basin, unknown amounts of coalbed methane gas have been produced with conventional natural gas for over 50 years (Quarterly Review, 1993). Targeted coalbed methane production increased in the late 1980s, and at least 232 coalbed methane wells had been completed as of January 1993 (Quarterly Review, 1993). During this timeframe, development was centered on Montgomery County, Kansas, with the most active operators being Great Eastern Energy and Development Corporation with 81 wells, Kan Map Inc. with 47 wells, and Stroud Oil Properties Inc. with 35 wells, (Quarterly Review, 1993). In addition to these operators, Bonanza Energy Corporation, Conquest Oil Company, Foster Oil & Gas, Hunter, Quantum Energy, Uranus, and U.S. Exploration had active development programs, and Derrick Industries was planning a program (Quarterly Review, 1993).

The coalbed methane wells were typically fractured with water or nitrogen-based fluids and sand, although the shallower Mulky coal received fracturing treatments of 40-pound linear gel and sand (Quarterly Review, 1993). On average, 5,000 pounds of sand were used per foot of coal (Quarterly Review, 1993). Another technique used in Kansas consists of injecting 4 barrels of 15 percent hydrochloric acid mixed with 16 barrels of potassium chloride and 15,000 standard cubic feet of nitrogen (Stoeckinger, 1990). In the Sycamore Valley field in Kansas, Stroud Oil Properties used 426 barrels of cross-linked fluid with 52 percent pad and 3 percent flush, and 30,000 pounds of 12/20 sand mixed at one to nine pounds per gallon injected at 20 barrels per minute. Operators were avoiding large-volume treatments due to a fear that fractures could be induced in thick water-bearing sands above and below the coals, which would have created excess water production (Quarterly Review, 1993). Stoeckinger (2000) reports that current hydraulic fracturing practices in the Cherokee Basin in Kansas are water only, no gel, with nitrogen being popular and "slick-water down tubing."

Pam Hudson, of the Oklahoma Corporation Commission, indicated that coalbed methane extraction was beginning to grow in the Cherokee Basin in the northeastern section of Oklahoma, and more development was now centered on that region as opposed to the Arkoma Basin to the south. Ms. Hudson expected that much of the development would be focused on Washington, Nowata, and Craig Counties (Hudson, 2001).

In Missouri, there appears to be little to no coalbed methane extraction within the Cherokee Basin. David Smith, a geologist with the Missouri Geological Survey, stated that coalbed methane extraction in Missouri is essentially non-existent (Smith, 2001).

8.3.3 Forest City Basin Production Activity

The Forest City Basin was relatively unexplored in 1993, with about ten coalbed wells concentrated in Kansas' Atchison, Jefferson, Miami, Leavenworth, and Franklin Counties (Quarterly Review, 1993). The wells were hydraulically fractured with 500 to 30,000 pounds (with an average of 5000 pounds) of sand proppant. The types of fluids used during the fracturing process were not mentioned (Quarterly Review, 1993).

David Smith, believes that at one time there were some coalbed methane wells just south of Kansas City in Cass County (Smith, 2001). Sherri Stoner, of the Missouri Geological Survey, confirmed this in February 2001, and remarked that they were no longer in operation (Stoner, 2001). An Iowa Division of Natural Resources Geological Survey Bureau geologist, Mary Howes, stated that presently there was no coalbed methane production in Iowa (Howes, 2001).

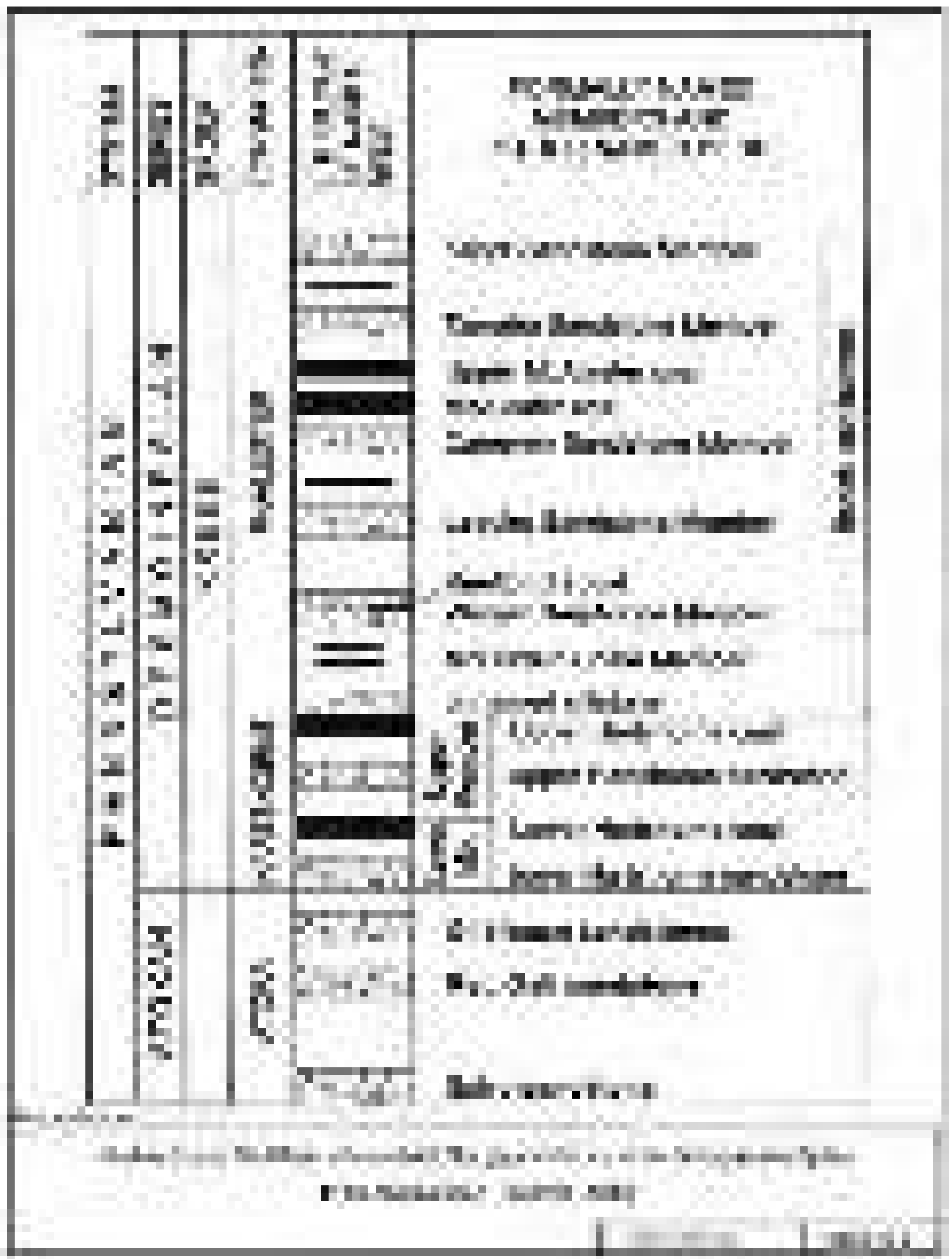
Information concerning coalbed methane production activity in Nebraska could not be found.

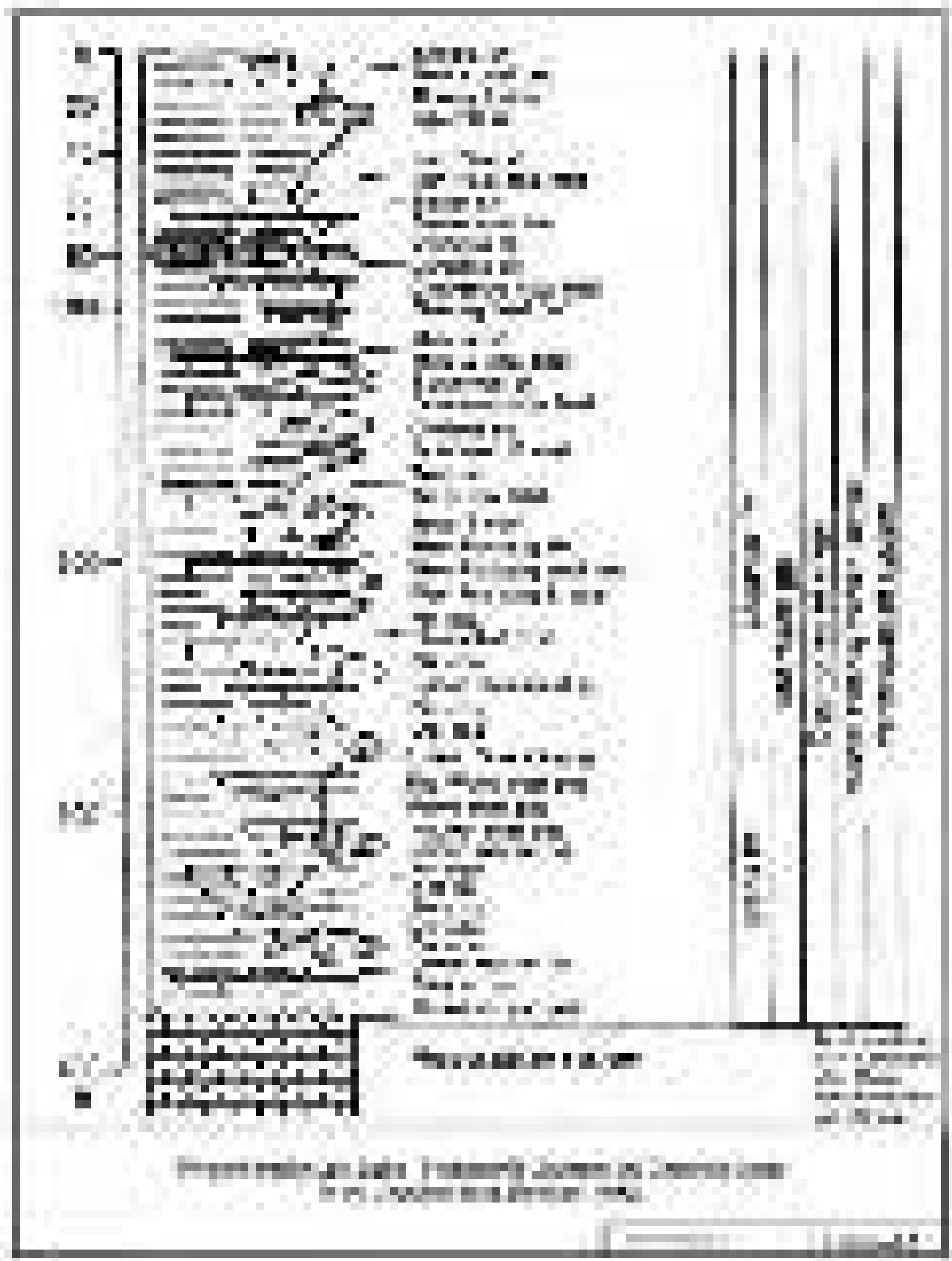
8.4 Summary

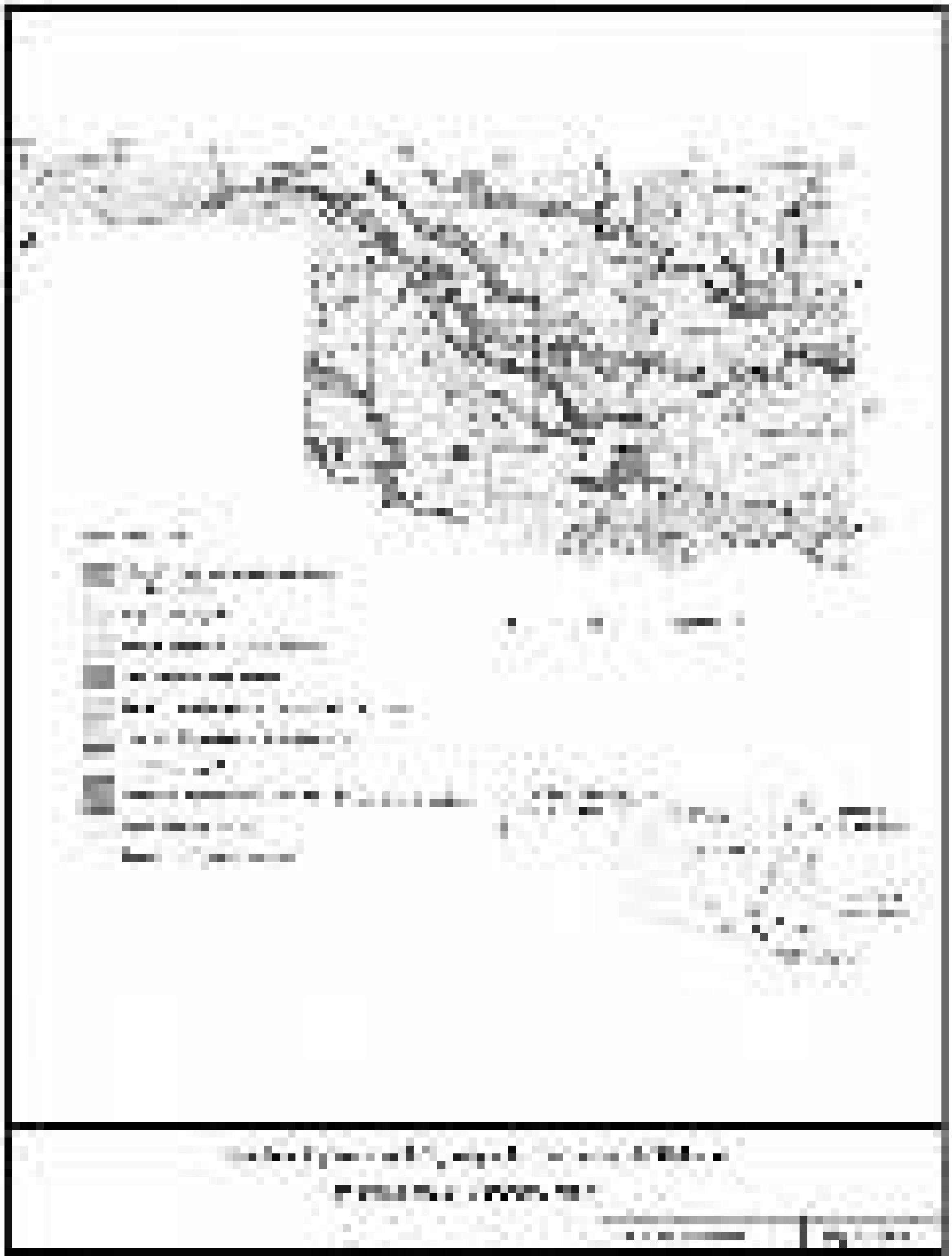
Based on depths to the Hartshorne Coal and the base of fresh water presented in Table A8-1, it appears that coalbed methane extraction wells in the Arkoma Basin could be coincident with potential USDWs in Arkansas, potentially allowing for impacts. Based on maps provided by the Oklahoma Corporation Commission (2001), which depicts the depths to the10,000 mg/L of TDS groundwater quality boundary in Oklahoma, the location of coalbed methane wells and USDWs would most likely not coincide in Oklahoma. This is based on depths to coals typically greater than 1,000 feet (Andrews et al., 1998) and depths to the base of the USDW typically shallower than 900 feet (Oklahoma Corporation Commission, 2001).

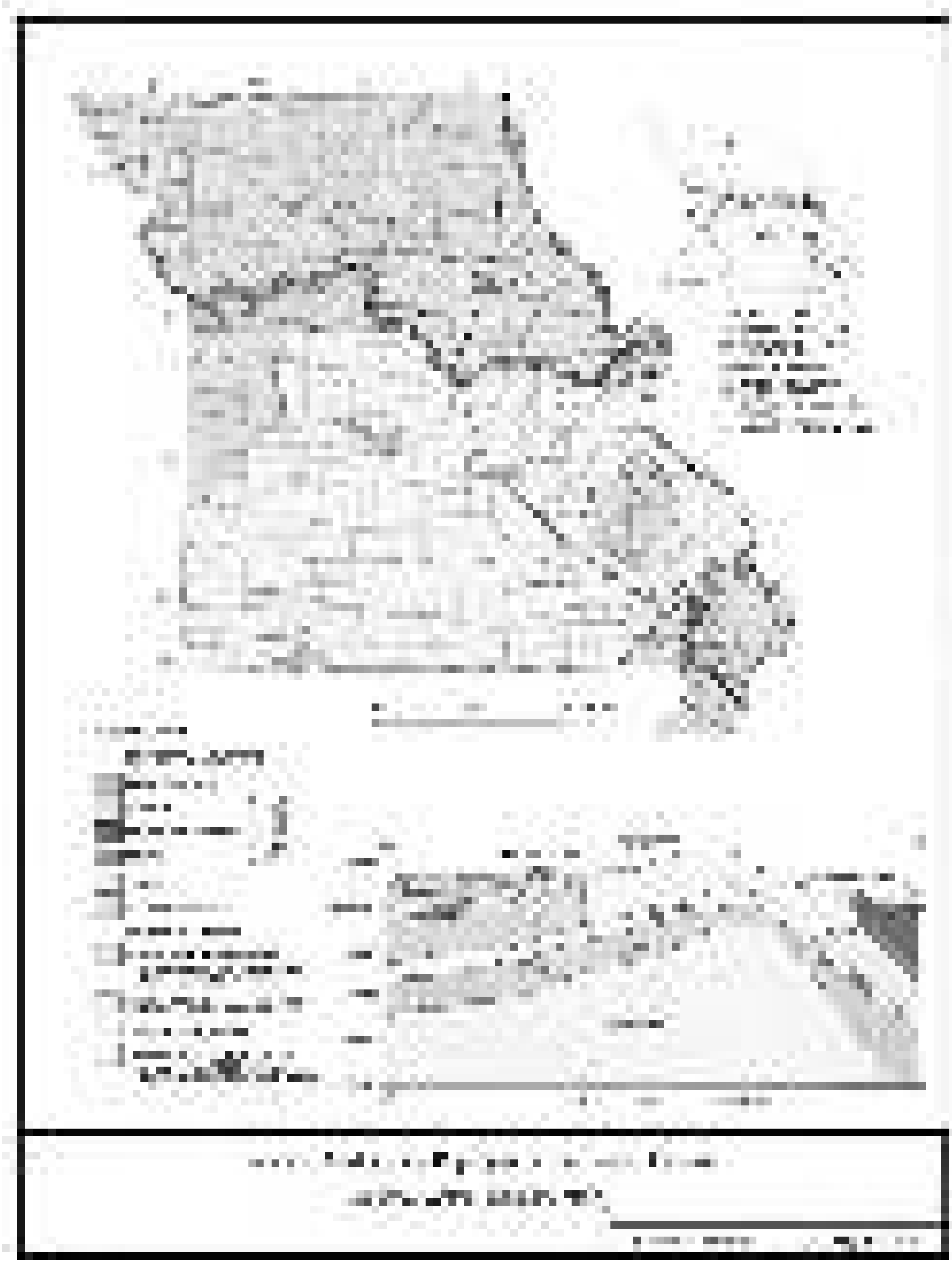
Table A8-2 supports the possibility that coalbed methane wells in the Cherokee Basin targeting the Cherokee Group coals in Kansas may coincide with USDWs, indicating the potential for impacts to drinking water. In Missouri, more water quality data is required prior to any determination of coalbed methane well/USDW conflict. In addition, since only a very small portion of the Cherokee Basin falls within the state, this portion of the basin needs to be delineated more precisely to see which USDWs lay within this small part of the basin. However, current levels of coalbed methane activity in Missouri are minimal.

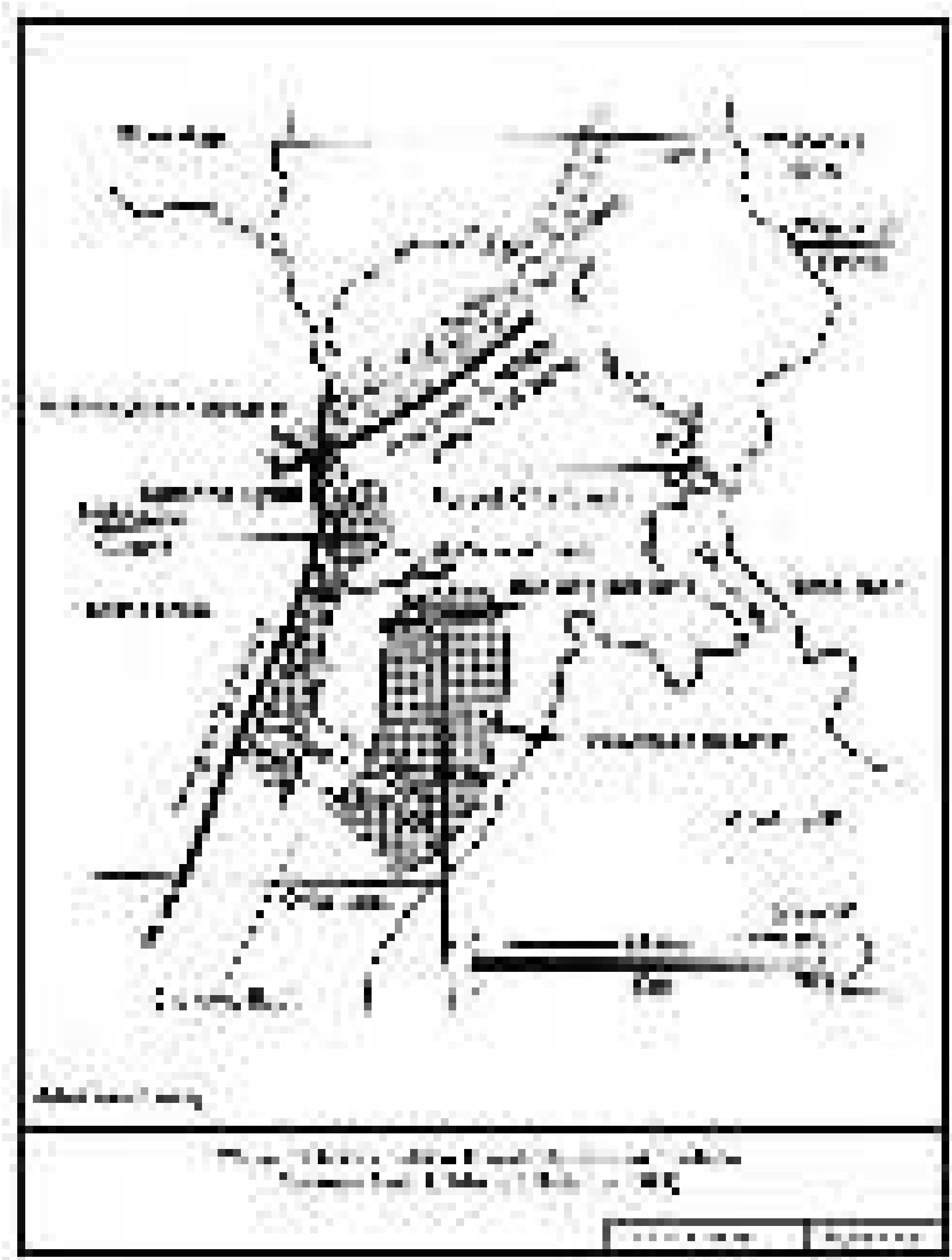
Last, in the Forest City Basin, there appears to be little physical relationship between coalbeds that may be used for coalbed methane extraction and water supplies. However, aquifer and well information from the National Water Summary (1984) indicate that a co-location of the two could exist in Nebraska. More information would be needed to fully investigate the relationship between coalbeds and USDWs in the Forest City Basin.

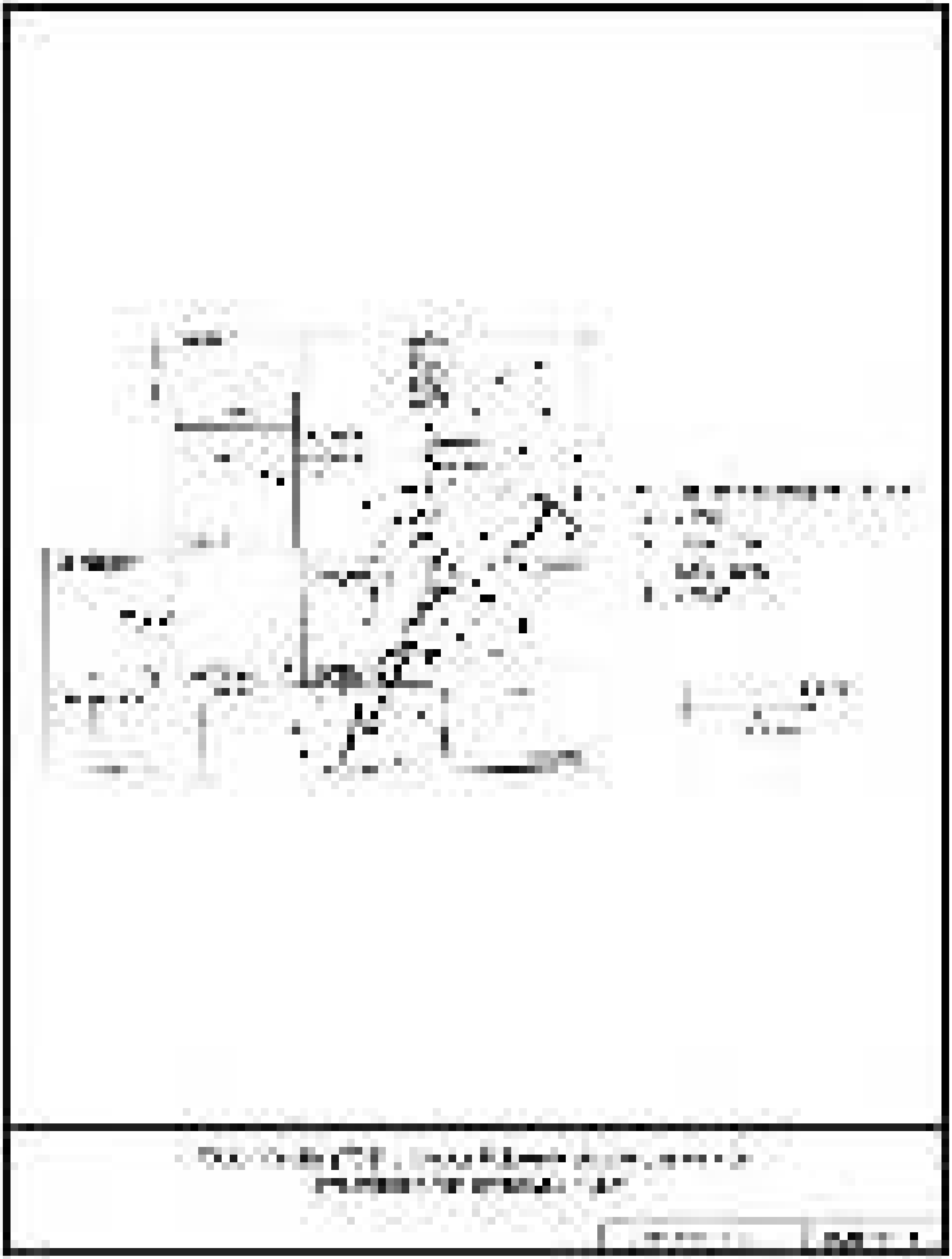


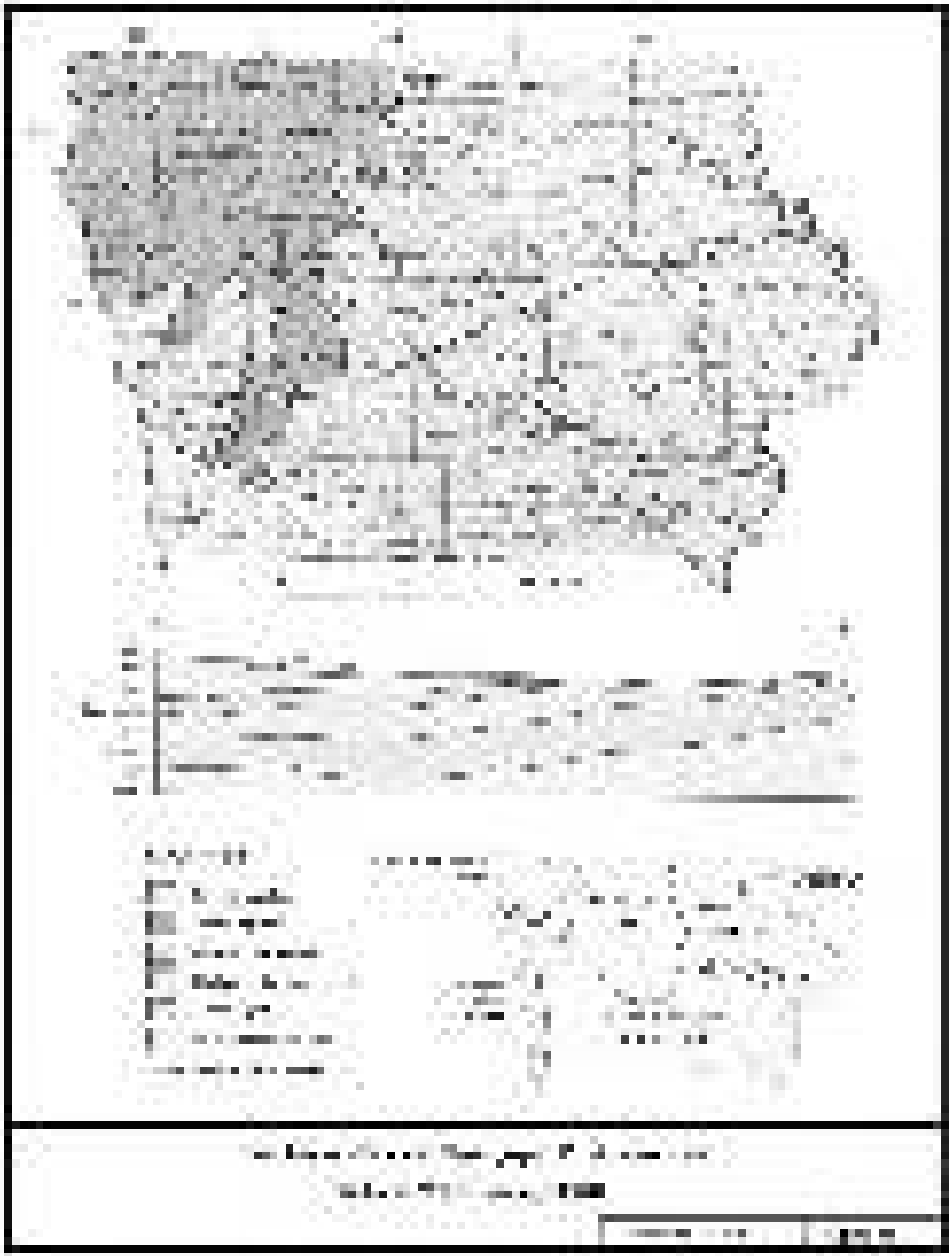


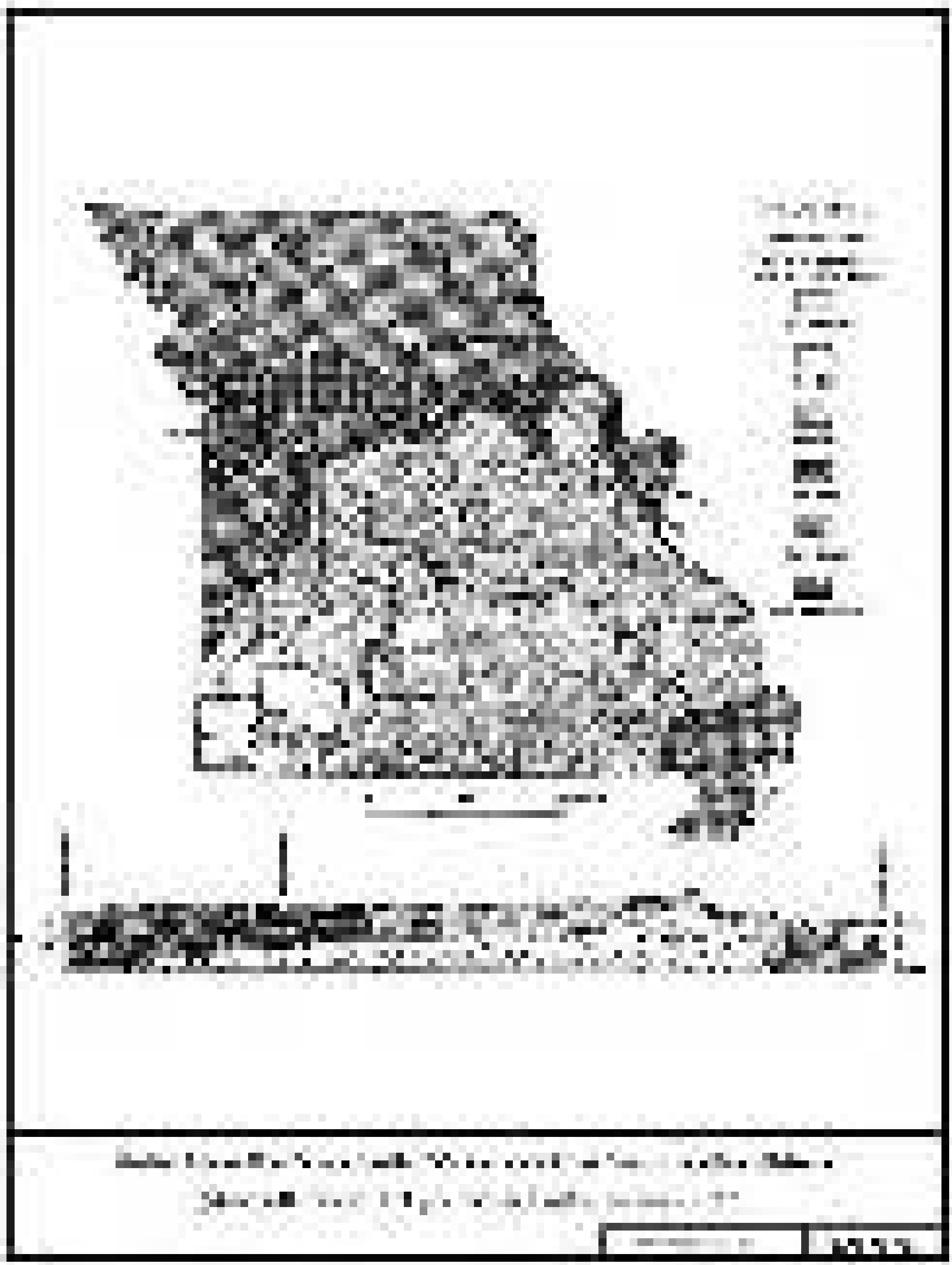


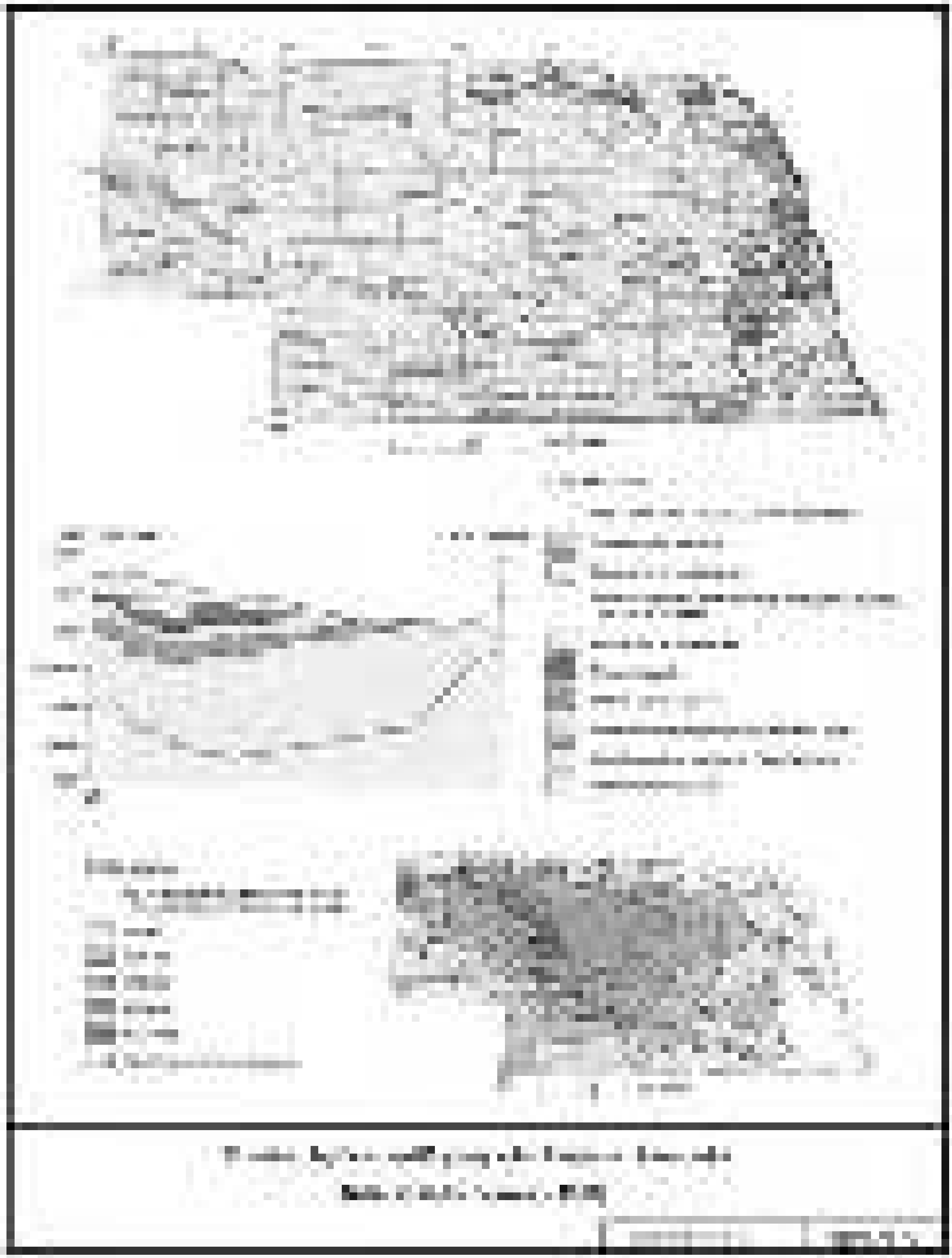












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Attachment 9
The Raton Basin

The Raton Basin covers an area of about 2,200 square miles in southeastern Colorado and northeastern New Mexico (Figure A9-1). It is the southernmost of several major coal-bearing basins along the eastern margin of the Rocky Mountains. The basin extends 80 miles north to south and as much as 50 miles east and west (Stevens et al., 1992). It is an elongate asymmetric syncline, with 20,000 to 25,000 feet of sedimentary rock in the deepest part. Coalbed methane resources in the basin, which have been estimated at approximately 10.2 trillion cubic feet (Tcf), are contained in the upper Cretaceous Vermejo Formation and upper Cretaceous and Paleocene Raton Formation (Stevens et al., 1992). In 2000, the average gas production rate per well in the Raton Basin was close to 300,000 cubic feet per day, and annual production was 30.8 billion cubic feet (Bcf) (GTI, 2002).

9.1 Basin Geology

The Raton structural basin is an asymmetric synclinal sedimentary basin containing sedimentary rocks as old as Devonian overlying basement Precambrian rocks, with Holocene sediments at the surface. The coal occurs in the Vermejo and the Raton Formations, which overlie the Trinidad Sandstone, a basin-wide regressive marine sandstone (Figure A9-2). The Vermejo and Raton Formations consist of deltaic lower coastal plain and fluvial deposits (Flores and Pillmore, 1987). Numerous discontinuous and thin coalbeds are located in the Vermejo Formation and the Raton Formation, which overlie the Trinidad Sandstone (Figure A9-3). The top of the Trinidad Sandstone forms the lower boundary of the Raton coal basin as shown in Figure A9-1. Development of coalbed methane wells has focused on development of the Vermejo coals rather than the Raton coals because the former are thicker and more abundant. The coalbeds are of limited extent and cannot be correlated over more than a few miles.

Individual coalbeds in the Vermejo Formation range from a few inches to about 14 feet thick, and total coal thickness typically ranges from 5 to 35 feet. An isopach map of total coal thickness in the Vermejo Formation, based on 92 well logs and measured sections, was published by Stevens et al. (1992) (Figure A9-4). Total coal thickness in the Raton Formation ranges from 10 feet to greater than 140 feet, with individual seams ranging from several inches to greater than 10 feet thick. Although the Raton Formation is much thicker and contains more total coal than the Vermejo Formation, individual coal seams in the Raton are less continuous and generally thinner. Additionally, because of extensive erosion of the Raton Formation, particularly in the eastern part of the basin, much of the original coal is no longer present (Stevens et al., 1992). Between 5 and 15 individual coalbeds produce coalbed methane for wells in the basin (Hemborg, 1996).

Middle Tertiary igneous intrusions are present in the central part of the basin (Steven, 1975). Sills and dikes have invaded sediments of the basin including both the Vermejo and Raton Formations. Sills have intruded along the coal seams destroying tremendous quantities of coal (Carter, 1956).

Coal seam depth is an important variable used to estimate gas production potential. Figure A9-5 is a thickness of overburden map from Stevens et al. (1992). The map shows the depth below land surface to the midpoint depth of the coal-bearing interval, using coal thickness as a weighting factor. Overburden thickness ranges from less than 500 feet near the basin perimeter to greater than 4,100 feet in the deep northwestern part of the basin. Many of the differences in thickness of overburden can be attributed to variations in topography and are thus a consequence of erosion and not necessarily subsurface geologic structure.

Stratigraphic cross-sections constructed to illustrate the regional subsurface geologic structure and the distribution of coal seams and igneous intrusions, as well as the areal locations of these cross-sections, are shown in Figures A9-6 through A9-8. The cross-sections use the top of the Trinidad Sandstone as the horizontal datum. The Vermejo Formation has a relatively uniform thickness of about 350 feet throughout the basin. The Raton Formation varies from about 0 to 2,100 feet thick. It grades westward into and is overlain by the conglomeratic Poison Canyon Formation (Flores, 1987; Flores and Pillmore, 1987).

A study of the relationship between coal cleat orientation and the compression stresses due to tectonic forces can indicate areas likely to have increased coal seam permeability and provide increased coalbed methane yield (Stevens et al., 1992). Cleats, or small-scale fractures in the coal, are commonly oriented perpendicularly to the maximum stress. These fractures tend to expand, thereby providing greater permeability and coalbed methane yields on the axes of the anticlines, such as the Vermejo Park anticline. Wells drilled near the axis of the La Veta syncline, in contrast, did not encounter adequate permeability (Stevens et al., 1992). Initially it was thought that sills that intrude along the bedding plane of the coal seams would reduce methane production, but several operators have noted that elevated methane contents have sometimes been measured in coal seams that have been intruded by igneous rocks (Stevens et al., 1992).

9.2 Basin Hydrology and USDW Identification

Regional groundwater flow in the Raton Basin is dependent on geologic structure and topography. Regional flow is generally down-slope from west to east or southeast (Figure A9-9). In the northern part of the basin, however, flow is radial away from Spanish Peaks (Howard, 1982; Geldon, 1990). Additionally, along the eastern margin of the basin, sediments dip to the west and groundwater flow is locally down-dip to the west. While recharge occurs primarily at elevations greater than 7,500 feet, discharge is mainly through streams and by evapotranspiration in the central and eastern parts of the basin.

Principle bedrock aquifers in the basin are the Cuchara-Poison Canyon, the Raton-Vermejo-Trinidad, the Fort Hayes-Codell, the Dakota-Purgatoire, and the Entrada (Geldon, 1990) (Figure A9-3). The pressure regime in the basin is poorly understood. Under-pressured conditions, or hydraulic heads in deep bedrock aquifers that are lower than those in shallow formations, appear to exist throughout much of the basin (Howard, 1982; Geldon, 1990; Tyler et al., 1995). This hydraulic head difference suggests that the deep bedrock aquifers are not in communication with shallow formations. Meteoric circulation, however, is indicated by the regional freshness of the produced waters (Stevens et al., 1992; Tyler et al., 1995).

All of the water produced along with coalbed methane in the Raton Basin has a total dissolved solids (TDS) content of less than 10,000 milligrams per liter (mg/L) (the water quality criterion for an underground source of drinking water (USDW)), and the aquifers from which the gas is produced meet the water quality criterion for a USDW (National Water Summary, 1984). A scatter diagram of potentiometric head versus TDS from coalbed methane wells in the Raton Basin (Figure A9-10) shows little correlation between potentiometric head and water quality. More importantly, this figure shows that all of the water had less than 10,000 mg/L of TDS, nearly all had a TDS of less than 2,500 mg/L, and more than half had a TDS of less than 1,000 mg/L. Two producers used injection wells for disposal, but operating permits issued to one gas producer (Evergreen Resources, Inc.) by the Colorado Department of Public Health and Environment allowed discharge of produced water into streambeds and stock ponds, indicating that the water was not too saline for surface discharge. Hemborg (1998) suggests that the wells yielding larger quantities of groundwater might be connected to the underlying water-bearing Trinidad Sandstone.

9.3 Coalbed Methane Production Activity

Hydraulic fracturing employed for enhancement of coalbed methane production is designed to enable gas within the rock to flow more readily to an extraction well. Coalbed methane well stimulation using hydraulic fracturing techniques is a common practice in the Raton Basin. Records show that fluids used are typically gels and water with sand proppants.

Hemborg (1996) reported that the average water production from coalbed methane wells in the Raton Basin was 700 barrels per million cubic feet (Mcf), and average daily production for 42 wells in the Spanish Peak Field was 0.309 Mcf (Hemborg, 1998). Conversion of these rates from coalbed methane industry units to those commonly used for water supplies gives an average water production rate for those wells of only 6.3

gallons per minute. These rates are generally not considered sufficient for public water supply or irrigation; however, they meet the water supply volume criterion for a USDW.

Hemborg (1998) showed that in most cases water yield decreased dramatically as coalbed methane production continued over time (Figure A9-11). However, some wells exhibited increased water production as coalbed methane production continued or increased over time (Figure A9-12). Two causal factors were suggested (Hemborg, 1998) for the rise in water production:

1. Well stimulation had increased the well's zone of capture to include adjacent water-bearing sills or sandstones that were hydraulically connected to recharge areas; or

2. Well stimulation had created a connection between the coal seams and the underlying water-bearing Trinidad Sandstone.

The Trinidad Sandstone is a bedrock aquifer confined by the Pierre Shale below and the shales and siltstones of the Vermejo Formation above (Figure A9-2). The Trinidad Sandstone exhibits low vertical and horizontal permeabilities of 0.186 and 0.109 meters per day, respectively, as reported by Howard (1982) in Stevens et al. (1992). One gas company reported that lower water production and improved gas production were achieved by avoiding known water-bearing horizons and by selectively completing the coal zones (Quarterly Review, 1993).

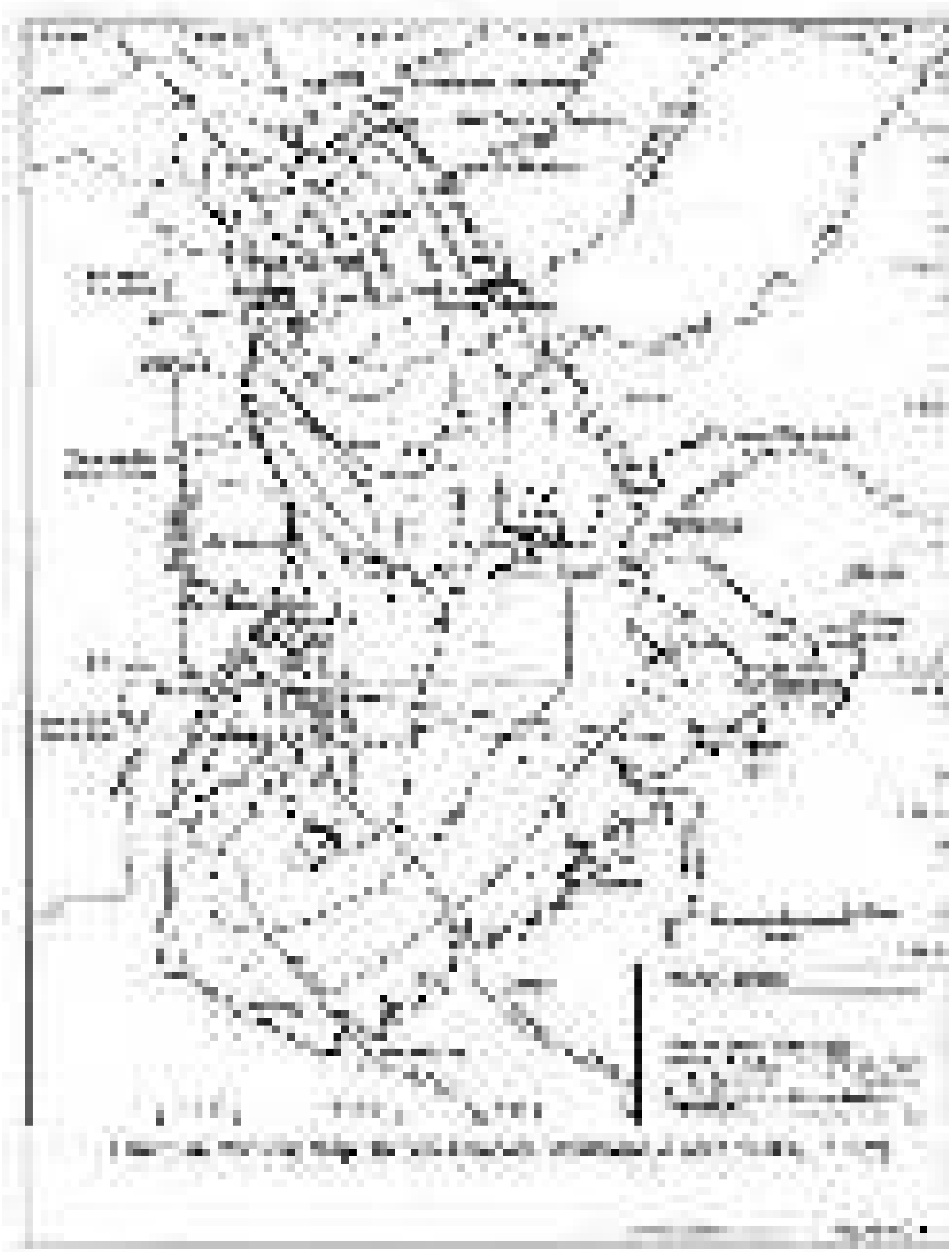
In-place coalbed methane resources in the Vermejo and Raton Formations were estimated by Stevens (1992) to be between 8.4 and 12.1 Tcf with a mean estimate of 10.2 Tcf. As of 1992, 114 coalbed methane exploration wells had been drilled in the basin (Quarterly Review, 1993). Soon after the Picketwire Lateral was constructed to convey gas from the fields to Trinidad and then to markets, gas well development in the basin increased significantly. The Purgatoire River Valley (Figure A9-1), which had been identified as having the highest coalbed methane potential in the basin, up to 8 Bcf per square mile (Stevens et al., 1992), became the focus of development. The Purgatoire Valley area was considered favorable for development because total coal thickness ranges from 5 to over 15 feet, drilling depths are shallow and coalbed methane content is high. The New Mexico portion of the basin was estimated to have methane resources ranging from 4 Bcf per square mile in the southern and eastern margins of the basin to more than 8 Bcf per square mile in the area south of the Vermejo Park anticline. Coal seams in the Vermejo Park area (Figure A9-1) are relatively thick, but shallow and of low rank, making estimates of coalbed methane content relatively low (Stevens et al., 1992).

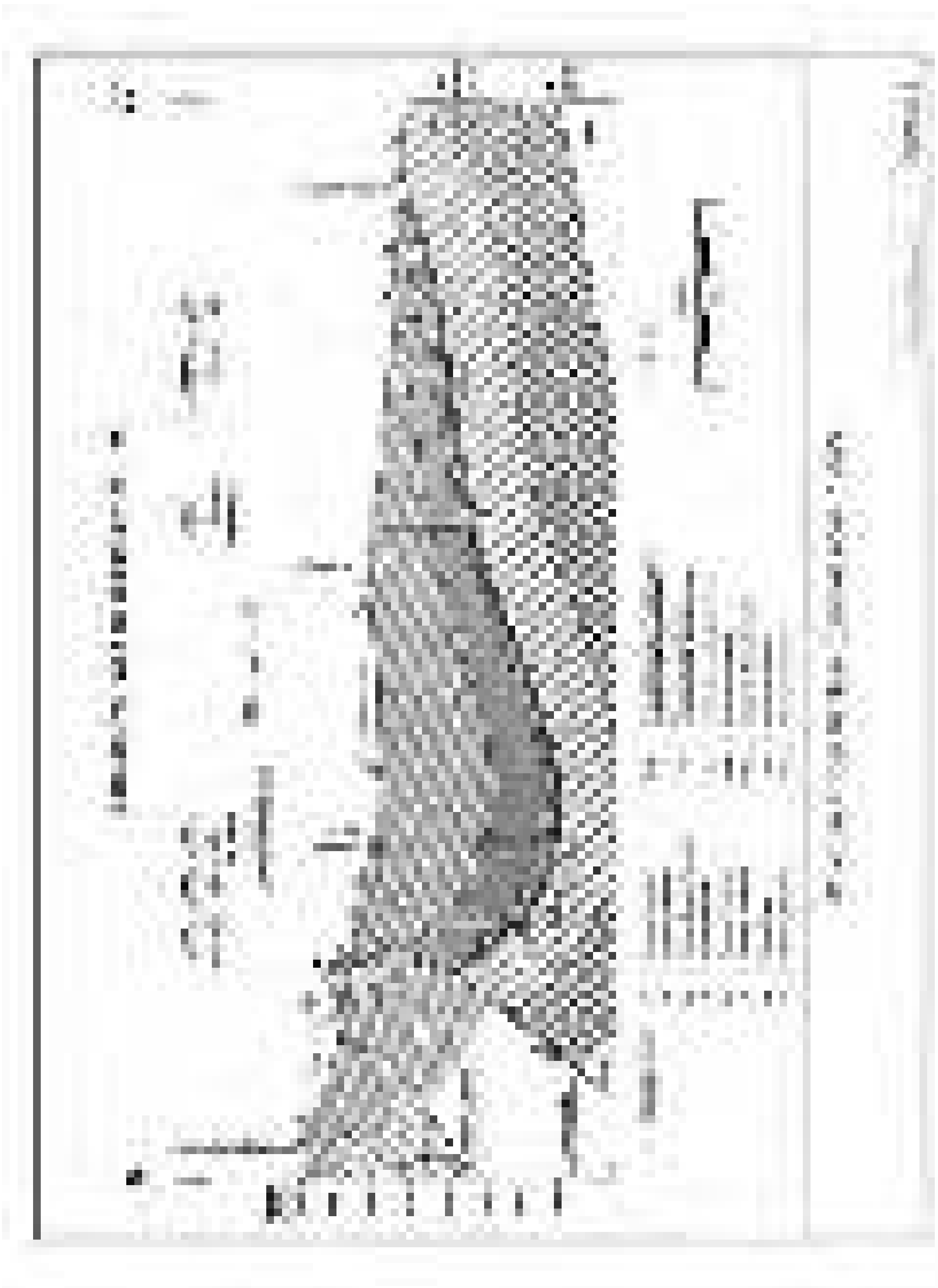
The Spanish Peak Field, in the Purgatoire River development area in Las Animas County, Colorado (Figure A9-1), had 53 active wells in December 1996. Plans had been announced by Evergreen Resources, Inc. to drill and complete an additional 40 wells in 1997 (Hemborg, 1998). In 1996, the Purgatoire development area was projected to be capable of producing 122-137 Mcf per day in 3 to 4 years (Figure A9-1) (Hemborg, 1996). Total coalbed methane production within the Raton Basin was 30.8 Bcf per year in 2000 (GTI, 2002).

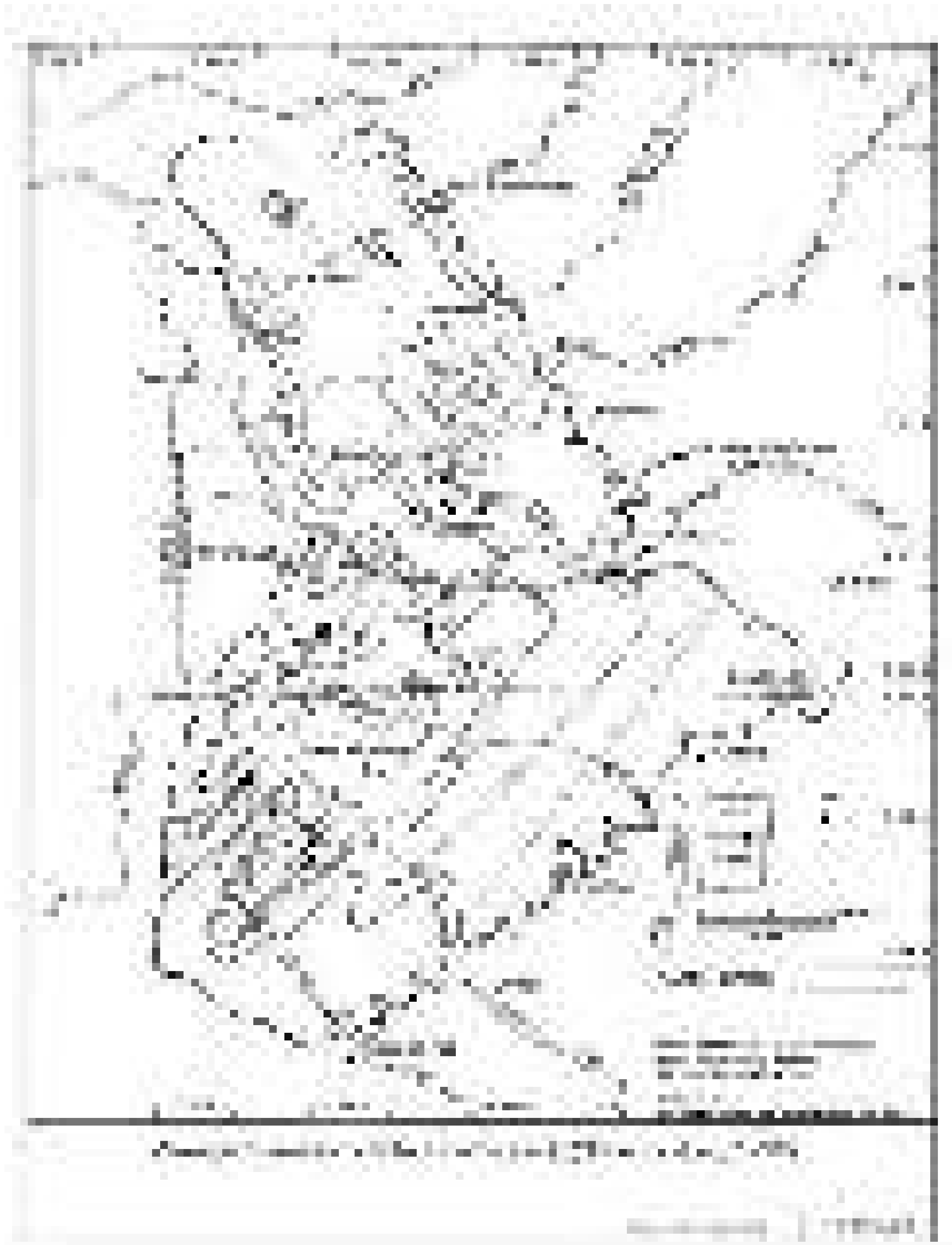
Methane production wells have generally been completed with 5.5-inch (outer diameter) casing with two to eight perforations per foot through the casing at the depths of the coal seams. The coal seams are stimulated with hydraulic fracturing treatments of sand and gelled-water, but detailed information on the nature, volumes, and use of hydraulic fracturing fluids in gas well development in this basin are not readily available. Water and gels with 10/40-mesh sand proppant seem to be the fluids of choice for fracturing practices in the Raton Basin. Stevens et al. (1992) report that multiple zones in one well are typically developed with 200,000 pounds of 10/20 or 20/40-mesh sand with 100,000 gallons of cross-linked gel per well. In one series of tests, wells were hydraulically fractured with 283,000 to 532,000 pounds of 12/20 and 20/40-mesh sand as proppant and 110,000 to 769,000 barrels of water or gel. The wells were fractured in two stages, one for a 25-foot thick upper zone and another for a 75-foot thick lower zone (Quarterly Review, 1993). Relatively high rates of water flow in these wells may be the result of fractures penetrating sandstones as well as coal seams. Another set of tests led a different methane producer to conclude that high water production was the consequence of induced fractures that intercept water-bearing sandstone and intrusive rocks. While operators initially assumed that large hydraulic fracture stimulations were necessary to link the thin and widely-spaced coal seams, it was found that such fracturing increased unwanted water production from associated sandstones, sills and water-bearing faults (Quarterly Review, 1993).

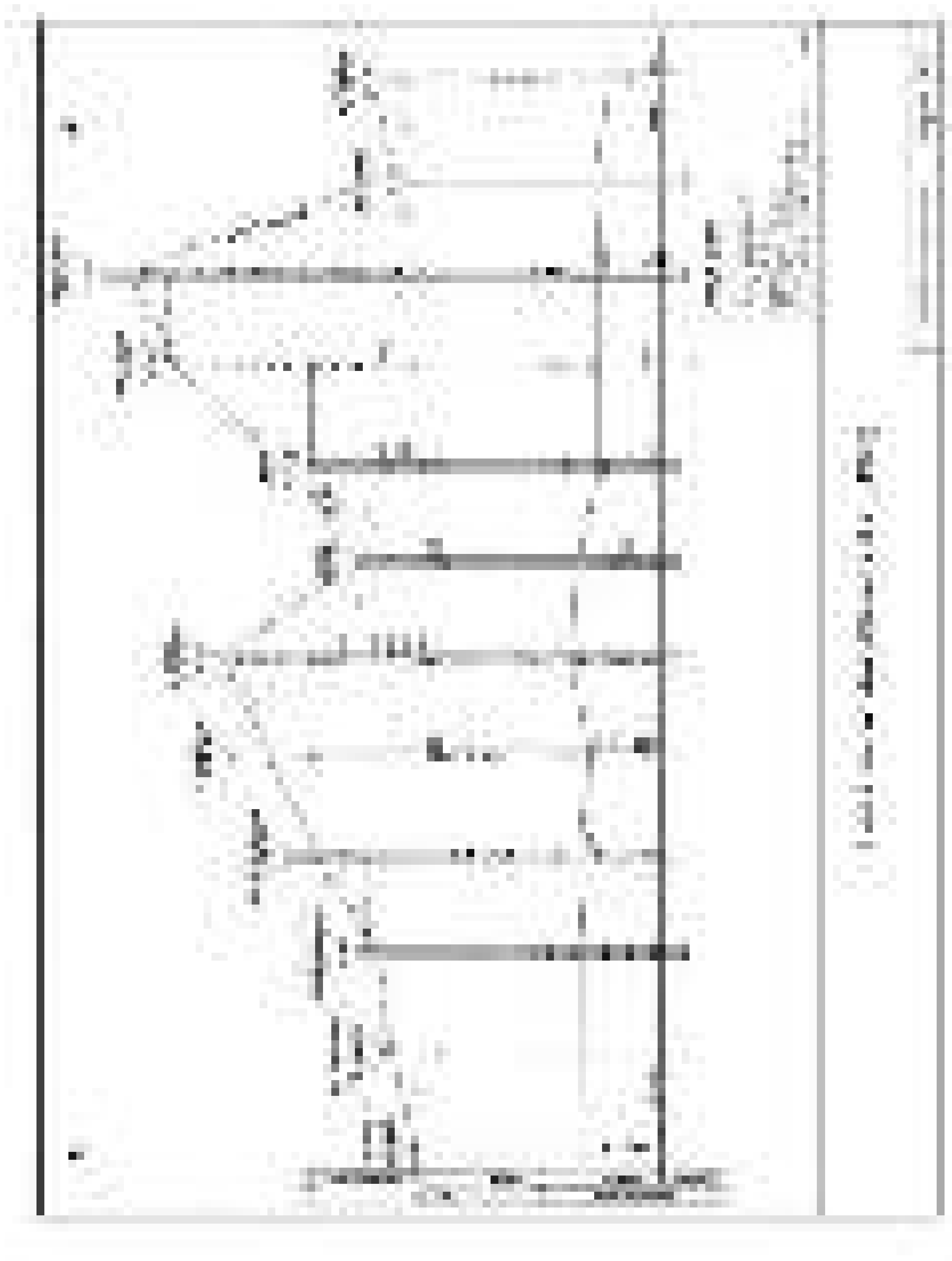
9.4 Summary

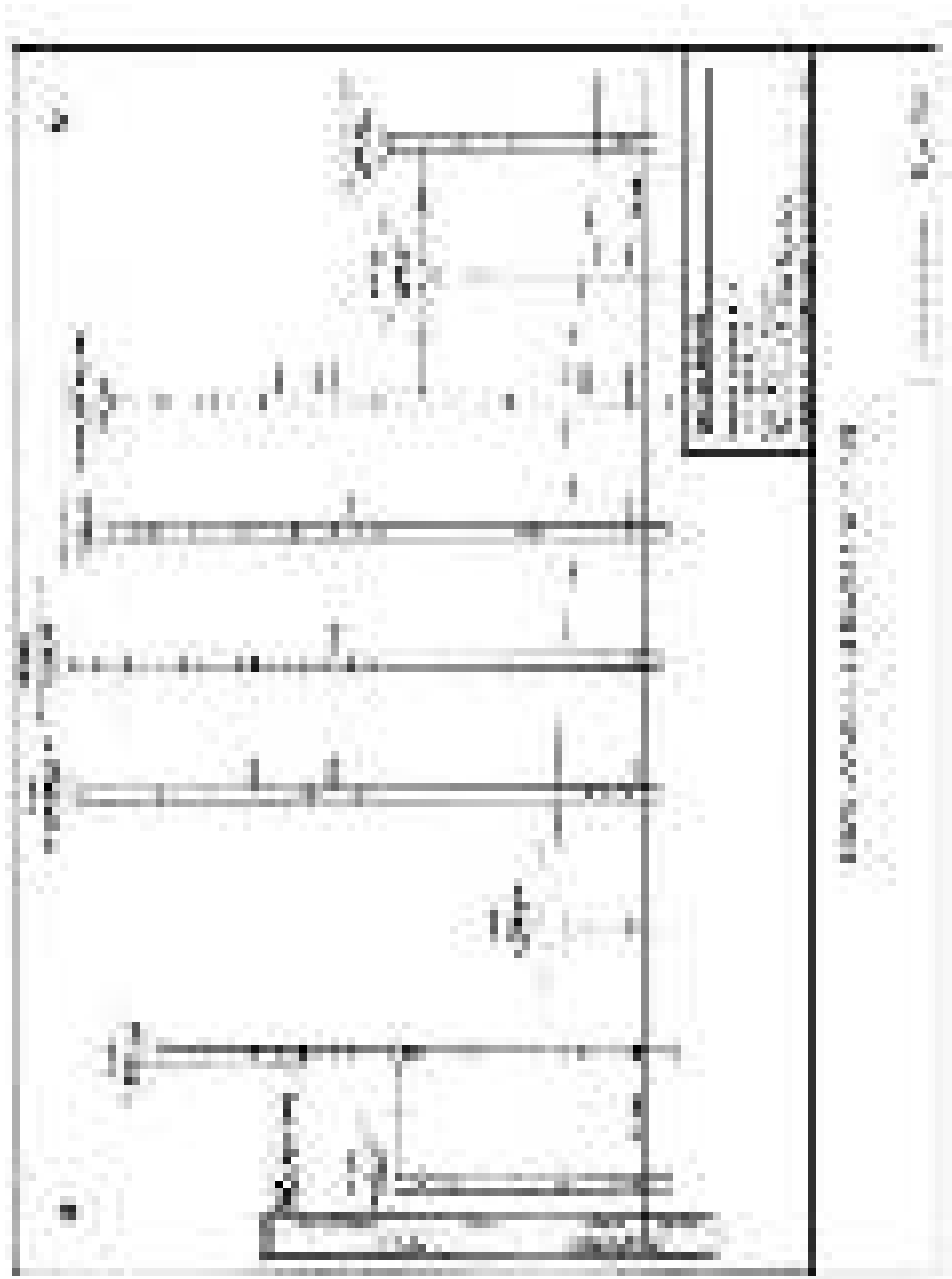
There are two major coal formations in the Raton Basin, the Vermejo Formation and the Raton Formation. The Vermejo coals range in thickness from 5 to 35 feet while the Raton coal layers range from 10 to over 140 feet thick. The coal seams of the Vermejo and Raton Formations, developed for methane production, also contain water that meets the water quality criteria for a USDW; therefore, it can be assumed that the Raton Basin coals are located within a USDW. The Cuchara-Poison Canyon, Fort Hayes-Codell, Dakota-Purgatoire, Entrada and Trinidad Sandstone and other sandstone beds within the Vermejo and Raton Formations, as well as intrusive dikes and sills, also contain water of sufficient quality to meet the USDW water quality criteria. Hydraulic fracturing may create connections to the Trinidad Sandstone, as shown by increases in water withdrawal from production wells over time. On the other hand, hydraulic connections to other adjacent water-bearing formations may also account for the increase in water production.

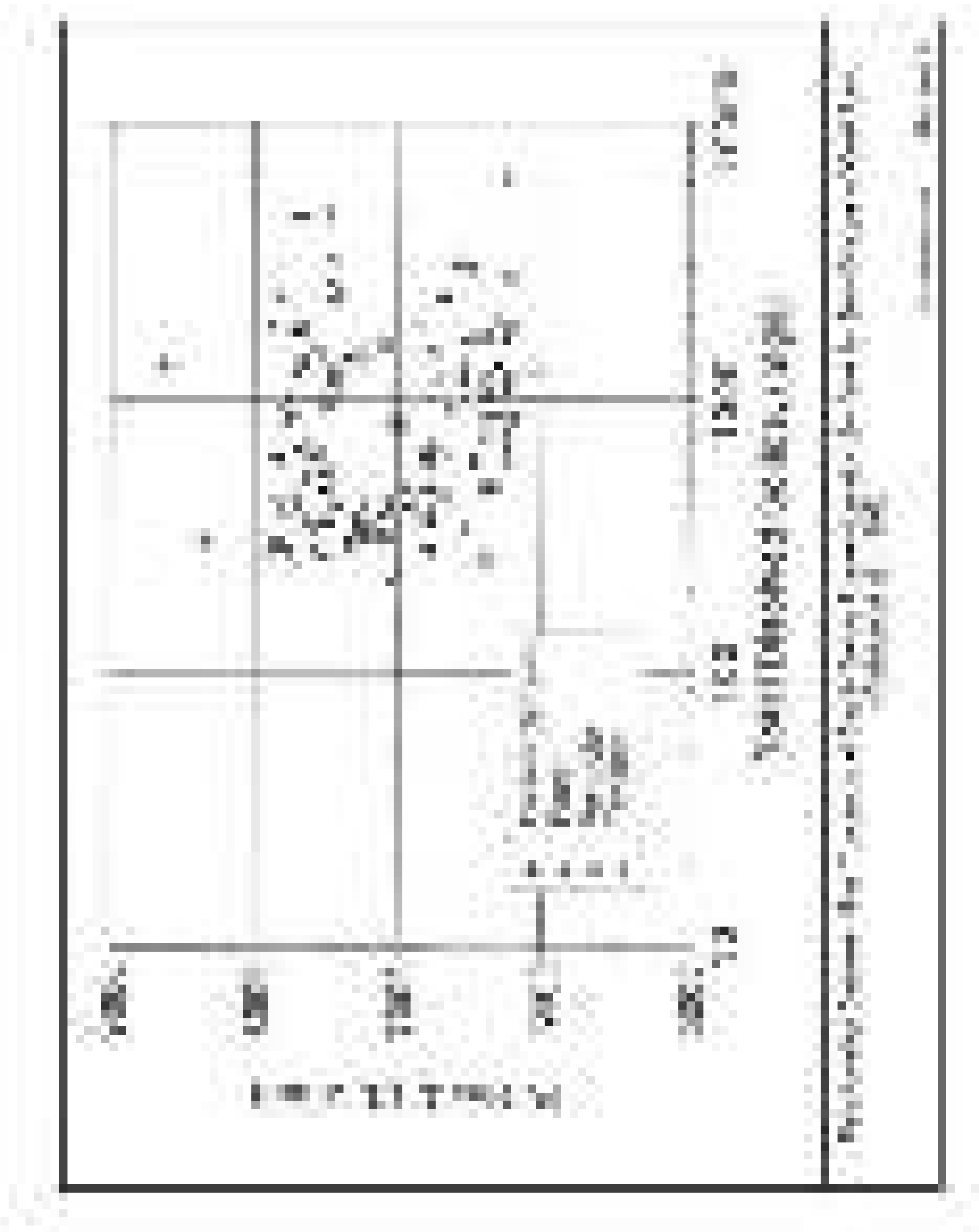


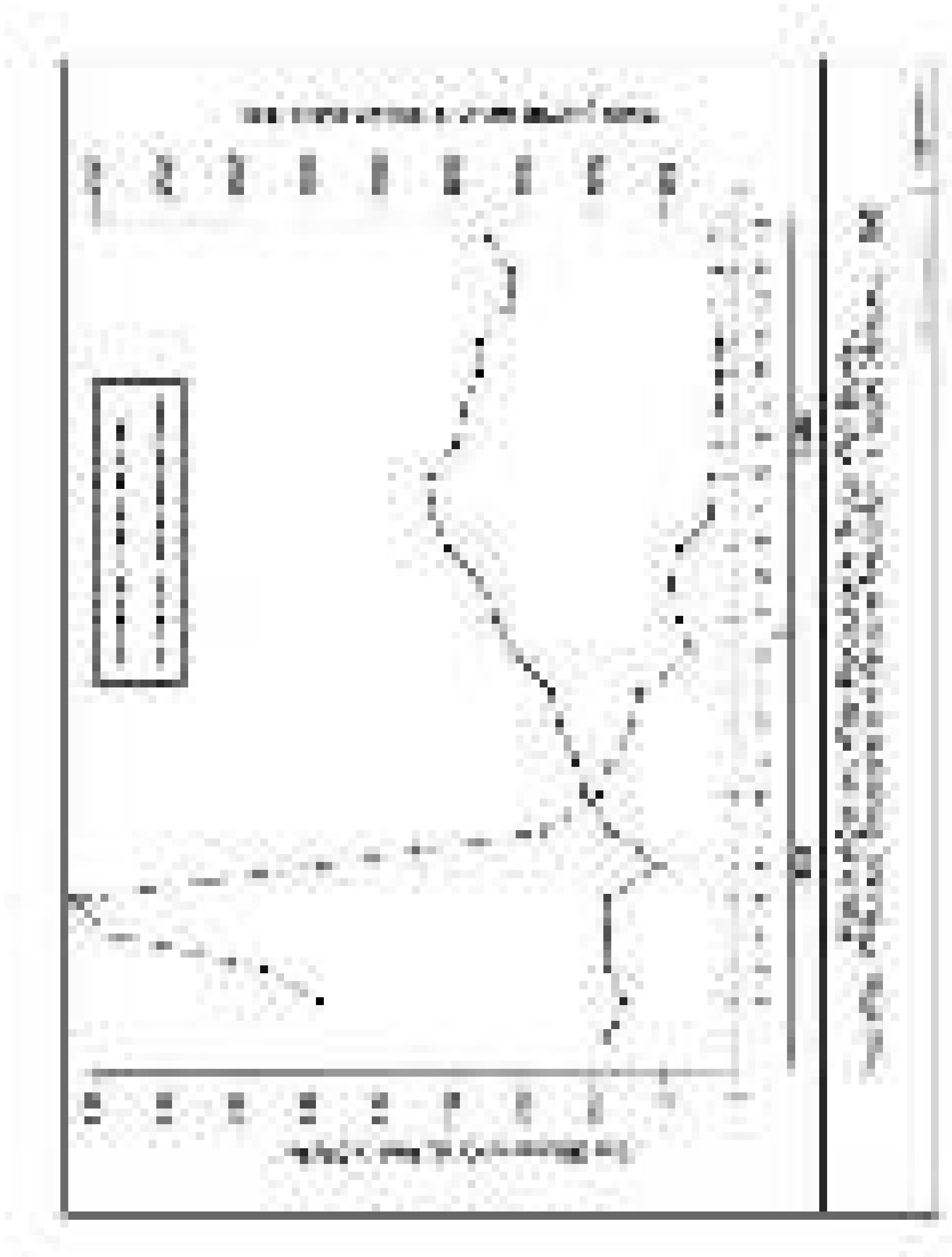


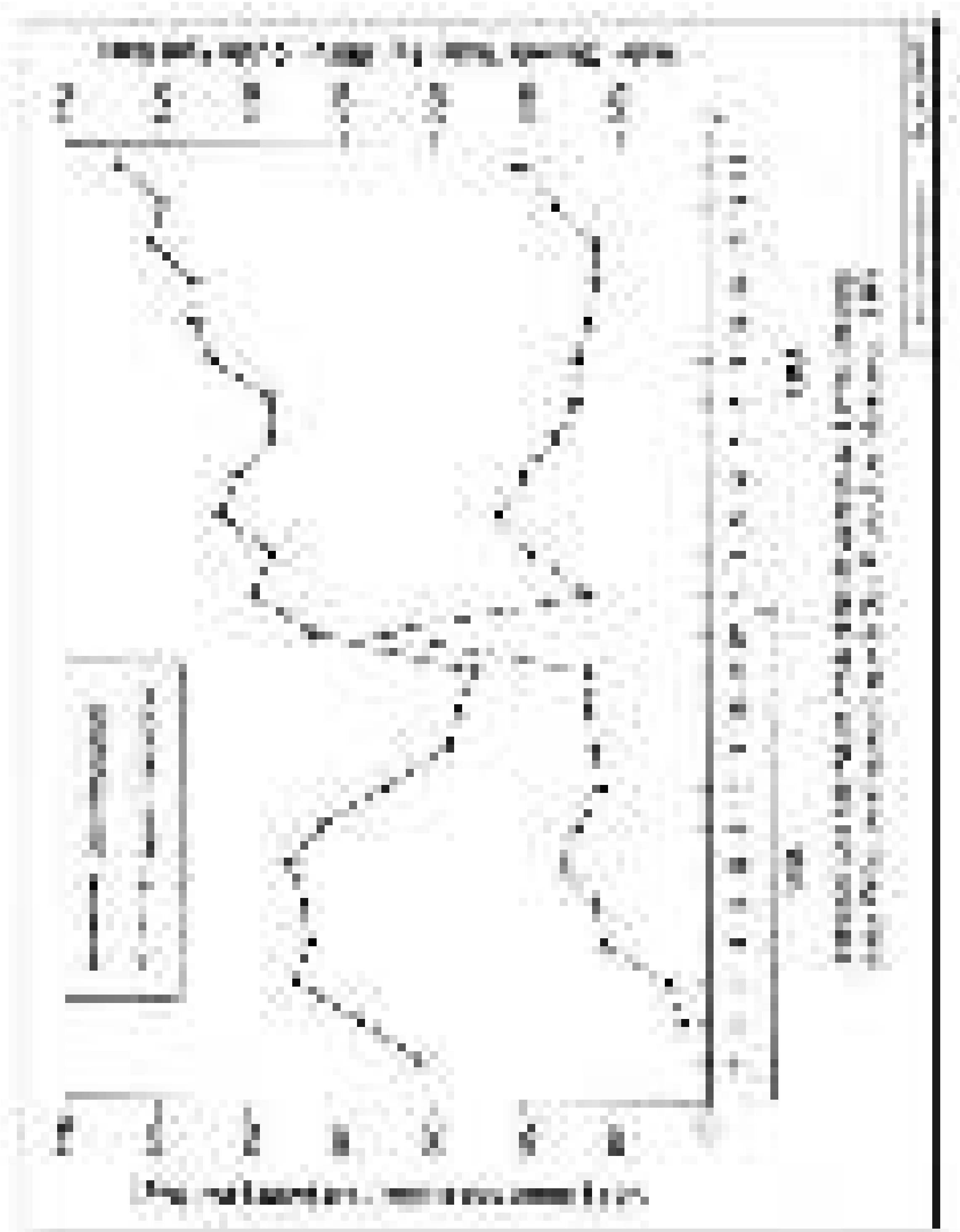












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Shaping the Future

Hydraulic Fracturing Study

PXP Inglewood Oil Field

October 10, 2012

Prepared For

Plains Exploration & Production Company

Hydraulic Fracturing Study

PXP Inglewood Oil Field

October 10, 2012

Prepared for

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Acronyms and Abbreviations

µg/L	micrograms per liter
µS/cm	microSiemens per centimeter
AML	acute myelogenous leukemia
AOR	area of review
API	American Petroleum Institute
bgs	below ground surface
BOD	biological oxygen demand
BOD_5	nitrate, nitrite, metals and biological oxygen demand
BOGM	Bureau of Oil and Gas Management
BTEX	benzene, toluene, ethylbenzenes and xylenes
BWPM	barrels of water per month
Cal-Tech	California Institute of Technology
CARB	California Air Resources Board
CAP	Community Advisory Panel
CBM	coalbed methane
CCAR	California Climate Action Registry
CCR	California Code of Regulations
CDMG	California Department of Conservation, Division of Mines and Geology
CEQA	California Environmental Quality Act
CFR	Code of Federal Regulations
CH_4	methane
CML	chronic myelogenous leukemia
CO	carbon monoxide
CO_2	carbon dioxide
CO_{2e}	carbon dioxide equivalents
COGA	Colorado Oil & Gas Association
C-PM_{10}	dust particles 10 microns or less in size
C-$PM_{2.5}$	dust particles 2.5 microns or less in size
CSD	Community Standards District
dB	decibels
dBA	A-weighted decibels
DEP	Department of Environmental Protection

DMRM	Divisions of Mineral Resource Management
DOGGR	California Department of Conservation, Division of Oil, Gas and Geothermal Resources
DTSC	California Department of Toxic Substance Control
DWR	California Department of Water Resources
EIR	Environmental Impact Report
EQAP	Environmental Quality Assurance Program
FRAC Act	Fracturing Responsibility and Awareness of Chemicals Act
GEIS	Generic Environmental Impact Statement
GHG	greenhouse gasses
GPS	global positioning system
GWP	Global Warming Potential
GWPC	Ground Water Protection Council
H_2S	hydrogen sulfide
HBO®	Home Box Office
Hz	hertz
IEA	International Energy Association
in/sec	inches per second
IOGCC	Interstate Oil and Gas Compact Commission
IPCC	Intergovernmental Panel on Climate Change
LAC DPH	Los Angeles County Department of Public Health
LARWQCB	Los Angeles Regional Water Quality Control Board
lbs	pounds
LNG	liquefied natural gas
MCL	California maximum contaminant level
Mg/L	milligrams per liter
MIT	mechanical integrity test
MRRP	monitoring, recordkeeping and reporting protocol
MTBE	methyl tert-butyl ether
MUN	municipal supply
N_2O	nitrogen dioxide
NESHAP	National Emissions Standards for Hazardous Air Pollutants
NGO	non-governmental organization

NO_x	nitrogen oxides
NPDES	National Pollutant Discharge Elimination System
NRC	National Research Council
NSPS	new source pollutant standards
NYCDEP	New York City Department of Environmental Protection
ODNR	Ohio Department of Natural Resources
OGCD	Oil and Gas Conservation Division
PAW	Petroleum Association of Wyoming
ppb	parts per billion
ppm	parts per million
ppmv	parts per million by volume
psi	pounds per square inch
PXP	Plains Exploration and Production Company
RWQCB	Regional Water Quality Control Board
SCAQMD	South Coast Air Quality Management District
SDWA	Safe Drinking Water Act
SGEIS	Supplemental Generic Environmental Impact Statement
SO_X	sulfur oxides
SPCC	Spill Prevention, Control, and Countermeasures
STRONGER	State Review of Oil and Natural Gas Environmental Regulations
TDS	Total Dissolved Solids
TOC	toxic organic compound
TPH	Total Petroleum Hydrocarbons
TPH-DRO	Total Petroleum Hydrocarbons – Diesel Range Organics
TRPH	Total Recoverable Petroleum Hydrocarbons
UIC	underground injection control
USDOE	U.S. Department of Energy
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
USDW	underground sources of drinking water
VOC	volatile organic compounds

Executive Summary

ES.1 Hydraulic Fracturing Study Objectives

The Inglewood Oil Field was discovered in 1924 by Standard Oil, and encompasses an approximate 1,000-acre area of the Baldwin Hills of Los Angeles County (Figure ES-1). Plains Exploration & Production Company (PXP) has operated the oil field since December 2002.

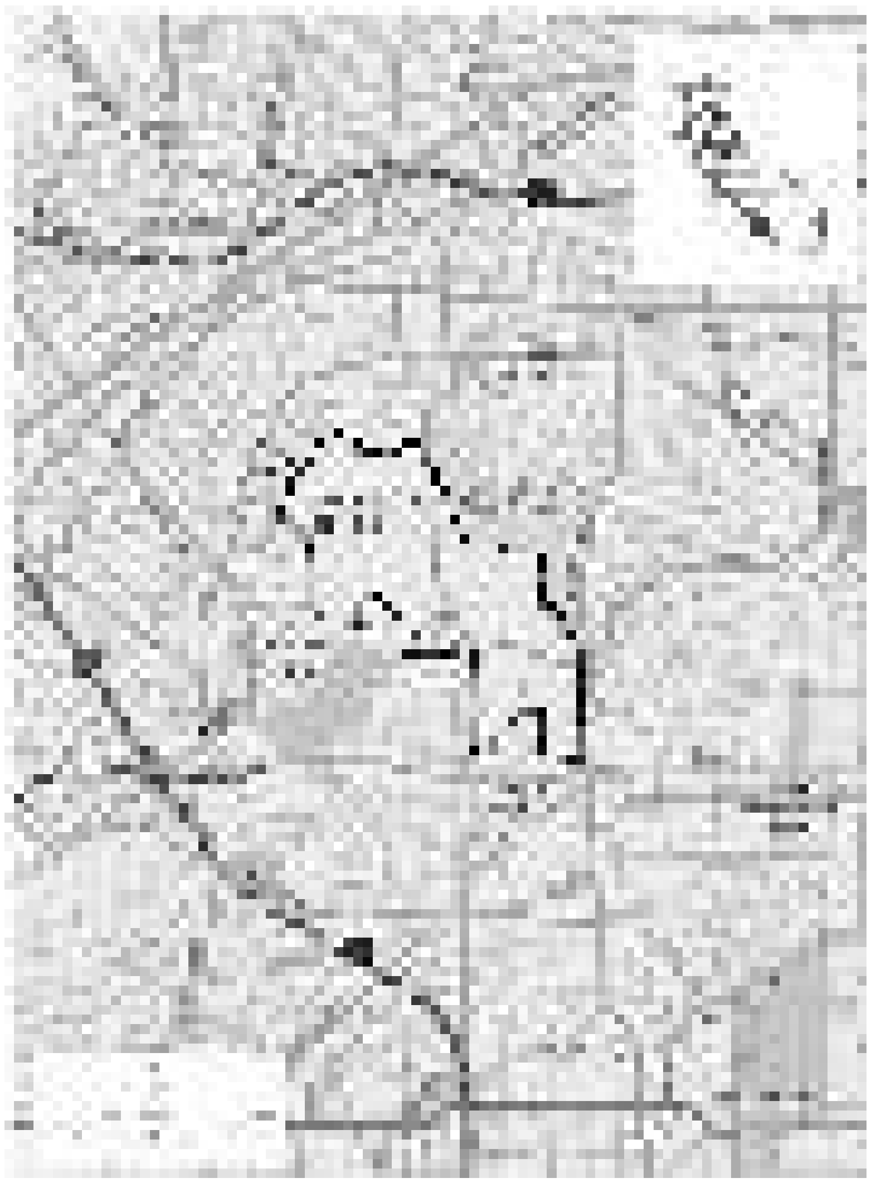

Figure ES-1 Regional Location Map

In October 2008, the County of Los Angeles (County) approved the Baldwin Hills Community Standards District (CSD), creating a supplemental district to improve the compatibility of oil production with adjacent urban land use. A lawsuit was filed in late 2008 against the County and PXP challenging the validity of the CSD. The lawsuit was settled July 15, 2011. This Hydraulic Fracturing Study is the direct result of Term 13 of the Settlement, which states:

> *PXP shall pay for an independent consultant to conduct a study of the feasibility and potential impacts (including impacts to groundwater and subsidence) of the types of fracturing operations PXP may conduct in the Oil Field. The study will also consider PXP's historic and current use of gravel packing. Such study will be completed within twelve (12) months of the date of this Agreement. Such study and all the back-up information for such study shall be provided to a qualified peer reviewer selected by the County and PXP, who shall review the study, back-up materials, and conclusions for completeness and accuracy. PXP must provide the independent expert with all materials requested and reasonably necessary for an accurate and verifiable study. The peer reviewer will be provided with access to all the data and materials provided to the independent expert. The peer reviewer shall agree to keep all proprietary information confidential. If the peer reviewer determines that the study is materially inadequate, incomplete or inaccurate, it shall so advise PXP's consultant who will complete the study as reasonably recommended by the peer reviewer and provide the revised study to the peer reviewer within 90 days. Upon acceptance by the peer reviewer, the study and all supporting material, including comments by the peer reviewer, shall be forwarded to the County, DOGGR, the Regional Water Quality Control Board ("RWQCB"), CAP and Petitioners and be available to the public, with any proprietary information redacted.*

This study draws on several sources, including sources in the peer-reviewed literature, the Inglewood Oil Field CSD, the 2008 Environmental Impact Report (EIR) conducted for the CSD, data and analyses provided by the contractor who conducted the recent hydraulic fracturing and high-rate gravel packing operations at the field, and from numerous contractors performing monitoring studies before, during, and after the recent hydraulic fracturing and high-rate gravel packing operations at the field.

In accordance with the Settlement Agreement, this study was reviewed by peer reviewers, jointly selected by the County and PXP. The peer reviewers, John Martin, Ph.D. and Peter Muller, Ph.D., C.P.G., were provided with the draft study and all reference materials. The peer reviewer's comments on the study, and their statement indicating that the revised study addressed all comments adequately and completely, thereby determining the study compete, is provided in Appendix A.

ES.2 Summary of Findings

The following are the primary findings of the Hydraulic Fracturing Study:

1. **Microseismic Monitoring:** The microseismic monitoring of high-volume hydraulic fracturing indicated that proppant-filled fractures were confined within the deep shale formations beneath the Inglewood Oil Field. Microseismic monitoring showed all fractures

were separated from the designated base of fresh water by 7,700 feet (1.5 miles) or more. Monitoring also showed all high-rate gravel packs stayed within their target zones.

2. **Groundwater:** Groundwater beneath the Inglewood Oil Field is not a source of drinking water, although the water quality must meet the standards for such a source. Groundwater beneath the Baldwin Hills is geologically isolated from the surrounding Los Angeles Basin and any water supply wells. Routine tests by the water purveyor show the community's water supply meets drinking water standards, including the period of high-rate gravel packs and conventional hydraulic fracturing, as well as the first high-volume hydraulic fracture in September 2011. In addition, the Inglewood Oil Field has an array of groundwater monitoring wells to measure water quality. Apart from arsenic, which is naturally high in groundwater of the Los Angeles Basin, the analyzed constituents meet drinking water standards. Before-and-after monitoring of groundwater quality in monitor wells did not show impacts from high-volume hydraulic fracturing and high-rate gravel packing.

3. **Well Integrity:** Tests conducted before, during and after the use of hydraulic fracturing and high-rate gravel packing showed no effects on the integrity of the steel and cement casings that enclose oil wells. There is also an ongoing program of well integrity tests at the Inglewood Oil Field.

4. **Methane:** Methane analyzed in soil gas and groundwater, as well as carbon and hydrogen isotopic rations in methane, at the Inglewood Oil Field did not show levels of concern. There was no indication of impacts from high-volume hydraulic fracturing or high-rate gravel packing.

5. **Ground Movement and Subsidence:** Before-and-after studies of high-volume hydraulic fracturing and high-rate gravel packing at the Inglewood Oil Field showed no detectable effect on ground movement or subsidence.

6. **Induced Earthquakes:** Before-during-and-after measurements of vibration and seismicity, including analysis of data from the permanently installed California Institute of Technology accelerometer at the Baldwin Hills, indicates that the high-volume hydraulic fracturing and high-rate gravel packs had no detectable effects on vibration, and did not induce seismicity (earthquakes).

7. **Noise and Vibration:** Noise and vibration associated with high-volume hydraulic fracturing and high-rate gravel packing operations at the Inglewood Oil Field were within the limits set forth in the CSD.

8. **Air Emissions:** Emissions associated with high-volume hydraulic fracturing were within standards set by the regional air quality regulations of the South Coast Air Quality Management District.

9. **Community Health:** The Los Angeles County Department of Public Health conducted a community health assessment that found no statistical difference of the health of the local community compared to Los Angeles County as a whole. Conventional hydraulic fracturing and high-rate gravel packs operations took place at the oil field, within the period addressed by the health assessment. Given the fact that public health trends in the area surrounding the

field were consistent with public health trends throughout the L.A. Basin it is reasonable to conclude that the conduct of hydraulic fracturing during the analyzed period did not contribute or create abnormal health risks

The Baldwin Hills CSD, and the associated Environmental Impact Report (EIR), together addresses the issues that are part of a hydraulic fracturing operation, such as truck traffic, water use, community compatibility (noise, light and glare, etc.), air emissions from vehicles and equipment used during the well development process, and other environmental resource categories. In addition, the EIR evaluates cumulative impacts, and environmental justice. These two documents support this Hydraulic Fracturing Study, which evaluates the effects measured and monitored during the high-volume hydraulic hydraulic fracturing and high rate gravel packing operations conducted in 2011 and 2012, as well as past activities of this type. The Hydraulic Fracturing Study did not identify a new impact not analyzed in the EIR, nor did it identify impacts greater in significance than those analyzed in the EIR.

Exacting protective measures and close monitoring are required by the Baldwin Hills CSD and by county, regional and federal agencies. These field-specific reviews and public and agency interactions compel PXP to enforce real-time compliance with all environmental standards in the Inglewood Oil Field. The long history of oil production in the area provides operators with an excellent understanding of the local subsurface conditions and reduces standard risks and uncertainties that would be present in new operations.

ES.3 Oil Production in the Los Angeles Basin and the Inglewood Oil Field

California is the fourth largest oil producing state in the U.S. (U.S. Energy Information Agency 2012), and the Los Angeles Basin is the richest oil basin in the world based on the volume of hydrocarbons per volume of sedimentary fill (Biddle 1991). Oil was first discovered in the area at the Brea-Olinda Oil Field in 1880, followed by numerous fields, including the Inglewood Oil Field in 1924 (Figure ES-2). As of this writing, there are 42 active fields in the Los Angeles Basin.

The Los Angeles Basin represents, from a global perspective, the ideal conditions for the generation and accumulation of hydrocarbons (Barbat 1958, Gardett 1971, Wright 1987a). The relatively recent geologic, tectonic, and structural history of the region has provided a thermal history that brings the organic-rich material into the “oil window”; the thermal regime that is optimum for development of oil and gas from organic precursors.

Discovery and development of the Los Angeles Basin oil fields accompanied rapid urbanization. Many oil fields were later covered by residential or commercial development, sometimes with continuing oil production. Chilingar and Endres (2005) evaluated urban encroachment on active and inactive oil fields, primarily in the Southern California area. They conclude that “a clear case is made for the urgent need for closer coordination and education by the petroleum industry of the local government planning departments…and in establishing mitigation measures for dealing with long-term environmental hazards.” The Inglewood Oil Field CSD, the associated EIR, and this Hydraulic Fracturing Study are coordinated processes that are meant to address such concerns.

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Figure ES-2
Location of Los Angeles Basin Oil Fields
09 | 21 | 12

The approximately 1,000-acre Inglewood Oil Field is one of the largest contiguous urban oil fields in the United States, with an estimated recovery of 400 million barrels of oil. Oil and natural gas produced from the field is sold and used entirely in California. The oil field is adjacent to the County of Los Angeles communities of Baldwin Hills, View Park, Windsor Hills, Blair Hills and Ladera Heights, as well as the City of Culver City.

The Baldwin Hills consist of rolling hills up to 511 feet above sea level, cut by canyons and gullies, and form part of a chain of low hills along the Newport-Inglewood Fault Zone. The Baldwin Hills have been uplifted above the Los Angeles basin by folding and faulting of the underlying geological formations.

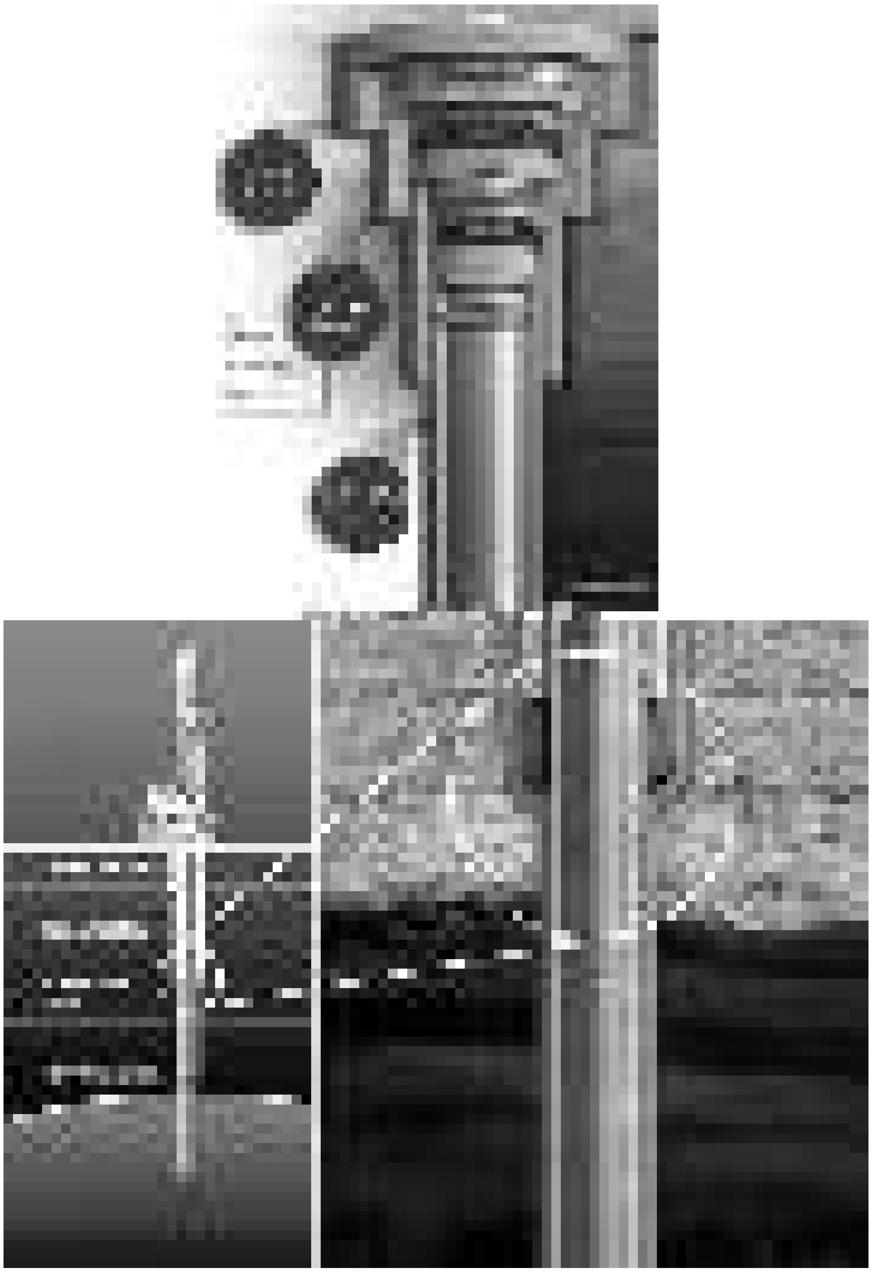
The petroleum producing zones in the Inglewood Oil Field comprise nine strata that range in depth from approximately 900 to 10,000 feet below the ground surface. In order of increasing depth and geologic age, the producing formations are: Investment, Vickers, Rindge, Rubel, Upper and Lower Moynier, Bradna, City of Inglewood, Nodular Shale, and Sentous. Water that is recovered along with the oil and gas, known as produced water, can make up to 90 percent or more of the total fluids pumped. The produced water has been reinjected in to the shallow depressurized Vickers and Rindge zones (known as a waterflood for enhanced oil recovery) since 1954, with much lesser amounts injected into the deeper Rubel and Moynier zones.

A total of 1,475 wells have been drilled over the life of the oil field; currently these are active, idle, or plugged. Many have been directionally drilled and are non-vertical. As of this writing, there are approximately 469 active production wells and 168 active waterflood injection wells operating at the Inglewood Oil Field.

ES.4 Well Drilling and Completion

Well drilling is the process of drilling a hole in the ground for the purposes of extracting a natural substance (e.g., water, oil, or gas). Oil wells are drilled using a drill string which consists of a drill bit, drill collars (heavy weight pipes that put weight on the bit), and a drill pipe. As the well is drilled and drilling fluid (i.e. mud, water and soil) is removed by the cementing process, a series of steel pipes known as casings are inserted and cemented to prevent the boring from closing in on itself (Figure ES-3). Cemented casing also serves to isolate the well from the surrounding formation. Each length of casing along the well is commonly referred to as a casing string. Cemented steel casing strings are a key part of a well design and are essential to isolating the formation zones and ensuring integrity of the well. Cemented casing strings protect against migration of methane, fugitive gas, and any formation fluid and protect potential groundwater resources by isolating these shallow resources from the oil, gas, and produced water inside of the well.

When initial drilling extends just below the base of fresh water, which is typically isolated from deeper saline formation water with an impermeable confining layer below it, the casing is placed into the drilled hole. The casing used in wells at the Inglewood Oil Field meets the State of California Department of Conservation, Division of Oil, Gas and Geothermal Resources (DOGGR) regulations and American Petroleum Institute (API) standards, which include requirements for compression, tension, collapse and burst resistance, quality, and consistency so that it is able to withstand the anticipated pressure from completing and producing the well (API 2009).



Source: Halliburton 2012

Figure ES-3 Depiction of Casing Strings

The space between this casing and the drilled hole (wellbore) is called the annulus. The annulus is filled with cement, permanently holding the casing in place and further sealing off the interior of the well from the surrounding formation. Cement serves two purposes: (1) it protects and structurally supports the well; and (2) it provides zonal isolation between different formations, including full isolation of the groundwater. Cement is fundamental in maintaining integrity throughout the life of the well, and after the well is idled and abandoned. It also protects the casing from corrosion. This bonding and the absence of voids stops the development of migration paths and isolates the production zone (Halliburton 2012, API 2009).

The final steps to install an oil-producing well are collectively known as well completion. Well completion includes the application of techniques such as sand control and well stimulation, including hydraulic fracturing, and installation of the production tubing and other downhole tools.

Well completions are not a part of the drilling process, but are applied after the well is drilled, sealed, and the drilling equipment has been removed. The first step to complete a well is to perforate the casing to allow the fluid from the producing formation to enter the well. Perforations are simply holes that are made through the casing. Once the casing is perforated, the well stimulation or sand control process is then initiated, depending on which technique is required. There are four types of well completion techniques described in this study that have occurred or may occur at the Inglewood Oil Field: conventional hydraulic fracturing and high-volume hydraulic fracturing, to stimulate and enhance production; and high-rate gravel packing and gravel packing, for sand control.

ES.4.1 Hydraulic Fracturing

In general, the process of hydraulic fracturing consists of injecting water, sand, and chemical additives into the well over a short period of time (typically less than one hour) at pressures sufficient to fracture the rocks to enhance fluid movement through the perforations and into the wellbore. Water and small granular solids such as sands and ceramic beads, called proppants, make up approximately 99.5 percent of the fluid used in hydraulic fracturing (Halliburton 2012). The flow of water acts as a delivery mechanism for the sand, which enters the newly-created fractures and props them open. These proppant-filled fractures allow oil and gas to be produced from reservoir formations that are otherwise too tight to allow flow. If proppant does not enter a new fracture, then the pressure of the overlying rocks forces the fracture closed once the overpressure is stopped, typically in less than one hour.

The chemical additives consist of a blend of common chemicals that increase water viscosity and help the sand and water mixture be carried further out into the fracture network. Additives include gels, foams, and other compounds. Additives have two primary functions: (1) to open and extend the fracture; and (2) to transport the proppant down the length of the fracture to maintain the permeability. Additives also perform critical safety functions such as controlling bacterial growth and inhibiting corrosion to help maintain the integrity of the well, which in turn protects groundwater. Most of the additives are recovered in the water that flows back after the hydraulic fracture (15 to 80 percent depending on the completion), and the remainder is recovered once the oil well is brought on to production and begins pumping fluids from the zone that was fractured (Halliburton 2012, USEPA 2010).

Hydraulic fracturing applied in oil and gas completions typically takes one of two forms, although some hybrid approaches are also in use. As indicated in the descriptions below, the process of fracturing in both forms are the same; the difference generally lies in the type of reservoir where the fracturing is occurring.

ES.4.1.1 Conventional Hydraulic Fracturing

This completion approach uses water, sand, and additives to fracture and stimulate the producing formation itself to a distance of up to several hundred feet from the well. This method is intended to affect the formation surrounding the perforated zone of the well, and enhance the permeability of the target producing zone itself. It is typically applied in sandstone, limestone, or dolomite formations.

ES.4.1.2 High-Volume Hydraulic Fracturing

This higher energy completion approach is generally applied to shales rather than sandstones that typically require a greater pressure to fracture. Sand and additives are used in the process, similar to conventional hydraulic fracturing; however, the primary distinguishing factor is the amount of fluid used in the process.

ES.4.2 High-Rate Gravel Packing

This completion approach uses water, gravel, and additives to place sand and gravel near the well itself with the objective of limiting entry of formation sands and fine-grained material into the wellbore, i.e., sand control. In this process, the space between the formation and the outer casing of the well is packed, at a high-rate, with gravel that is small enough to prevent formation grains (sand) and fine particles from mixing and entering the wellbore with the produced fluids, but large enough to be held in place by the well perforations. This relatively low-energy completion approach creates a fracture using water, sand, and additives that improve the proper placement of the gravel filter. This process is not intended to increase the permeability of the producing formation, and it only affects the area near the well itself.

Gravel packing, in contrast to high-rate gravel packing, does not exceed the local geological fracture pressure. In gravel packing operations, a steel screen is placed in the wellbore and the surrounding annulus packed with prepared gravel of a specific size designed to prevent the passage of formation sand. The primary objective is to stabilize the formation while causing minimal impairment to well productivity (Schlumberger 2012a). The gravel is circulated into place rather than pumped in under high pressure.

ES.5 Summary of Past and Future Hydraulic Fracturing and High-Rate Gravel Packing at the Inglewood Oil Field

Both conventional and high-volume hydraulic fracturing have been used at the Inglewood Oil Field. Figure ES-4 shows the location of Inglewood Oil Field wells that have either been completed by high-volume hydraulic fracturing or conventional hydraulic fracturing since 2003 when PXP began operating the field. All of the hydraulic fracturing has been completed on producing wells, that is, on pumping wells rather than injection wells. After the completion, flowback water brings back most of the additives used during the hydraulic fracturing operation to the surface. After the stimulation operation is completed, the well is brought on line and begins pumping, and any residual hydraulic fracturing fluids are drawn towards the well during pumping.

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LEGEND

Conventional Hydraulic Fracture

High Volume Hydraulic Fracture

Inglewood Oil Field Boundary

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Figure ES-4

Locations of Hydraulic Fracturing Operations at Inglewood Oil Field

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In conjunction with this Hydraulic Fracturing Study, PXP conducted high-volume hydraulic fracturing tests at two wells at the Inglewood Oil Field (VIC1-330 and VIC1-635). Only one stage was conducted as part of each of these tests. These are the only two high-volume hydraulic fracture jobs known to have been performed on the Inglewood Oil Field. The stages are representative of anticipated future hydraulic fracturing in terms of pressure, water use, and other factors.

Conventional hydraulic fracturing has been conducted on 21 wells in the deep Sentous, Rubel, Moynier, Bradna, City of Inglewood, and/or the Nodular Shale formations. Combined, a total of approximately 65 stages of conventional hydraulic fracturing have occurred at the Inglewood Oil Field since 2003.

PXP expects that, in the future, high-volume hydraulic fracturing and conventional hydraulic fracturing may be conducted in the deeper Rubel, Bradna, Moynier, City of Inglewood, Nodular, and Sentous zones (all located greater than 6,000 feet below ground surface).

PXP expects that, in the future, high-volume hydraulic fracturing and conventional hydraulic fracturing may be conducted in the deeper Bradna, City of Inglewood, Nodular, and Sentous zones (all located greater than 6,000 feet below ground surface).

PXP has operated the Inglewood Oil Filed since December 2002, and since that time, has conducted high-rate gravel pack completions on approximately 166 wells, in the Vickers and the Rindge formations, and one completion in the Investment Zone. Each high-rate gravel pack included an average of 5 stages per well. Approximately 830 stages of high-rate gravel packs have been completed at the Inglewood Oil Field since PXP began operating the field.

It is anticipated that high-rate gravel packing operations may be conducted on as many as 90 percent of all future production wells drilled within sandstones on the Inglewood Oil Field. This procedure results in less sand being drawn into the well during pumping, and reduces the amount of formation sand that must be managed at the surface. High-rate gravel pack operations use less water (~1,000 barrels vs. ~3,000 barrels) and lower pressures (~1,900 psi vs. ~9,000 psi) than hydraulic fracturing operations.

ES.5.1 Recent Hydraulic Fracturing Completions

PXP conducted two high-volume hydraulic fracture jobs at separate wells on the Inglewood Oil Field for the purposes of this study. The first hydraulic fracture completion was conducted on September 15 and 16, 2011, at the VIC1-330 well. The second completion was conducted on January 5 and 6, 2012, at the VIC1-635 well. Only one stage was completed during each operation.

Both of these operations were conducted in the Nodular Shale, a subunit of the Monterey Shale, approximately 8,000 to 9,000 feet below ground surface. The hydraulic fracture completions were conducted by Halliburton Energy Services with PXP oversight. Microseismic monitoring and fracture mapping was conducted by Schlumberger on the VIC1-330 and by Pinnacle (a Halliburton Company) on the VIC1-635. Halliburton (2012) contains a full report of both operations.

The applied pressure, water use, and monitored effects are expected to be similar between these two high-volume hydraulic fracture jobs and any future high-volume hydraulic fracture jobs to be conducted at the field. However, future high-volume hydraulic fracturing completions would likely utilize more than one fracturing stage each. In hydraulic fracture jobs that consist of more than one stage, each stage is conducted one after the other, never simultaneously. Therefore any one stage will be similar to those described in this section. The amount of water and chemicals used would be proportional to the number of stages.

Although both VIC1-330 and VIC1-635 are vertical wells, in the future, hydraulic fracturing may be conducted using horizontal wells, and with more stages. The high-volume hydraulic hydraulic fracturing job itself and the monitored effects would be the same in each stage as those measured during this study. The intent of the two high-volume hydraulic fracture jobs was to bound the potential effects of this process on the field. In the future, the only difference between these two jobs could be the construction of the well, including the number of stages applied. Each stage would be an isolated event, and each stage would be similar to the two analyzed in this Study. Although a horizontal well can be much longer than a vertical well in the same formation, the hydraulic fracture completion targets an individual zone, and so the amount of water, sand, and additives used would be the same, stage for stage. Horizontal wells, by drilling along the producing zone itself at depth, significantly reduce the number of wells needed to produce the same formation. As such, horizontal wells minimize the surface footprint of the oil production operation.

Water for the hydraulic fracturing operations at the Inglewood Oil Field is provided either from produced water at the field or, if a potassium-chloride gel is used, fresh water provided by California American Water Company, the provider of all fresh water used at the Inglewood Oil Field. For both of the high-volume hydraulic fracturing operations on the field, PXP used fresh water. Water produced from the target reservoirs during hydraulic fracturing operations, known as flowback water or flush water, is transported by pipeline to the field water treatment plant where it is mixed with other produced water generated on the field and processed. The treated water is then reinjected into the oil and gas producing formations as part of the waterflood process. This operation is in accordance with CSD Condition E.2(i), which requires that all produced water and oil associated with production, processing, and storage be contained within closed systems at all times. This process substantially reduced air emissions from the fluids. The total volume of additives is small and is diluted in the fluids of the producing zone.

ES.5.2 Recent High-Rate Gravel Pack Completions

PXP also conducted high-rate gravel pack jobs at two wells on the Inglewood Oil Field to collect data for this study. The first high-rate gravel pack was a five-stage completion performed on January 9, 2012, at the TVIC-221 well. The second high-rate gravel pack was a six-stage completion performed on the same day at a different well, TVIC-3254. Both of these operations were conducted in the Vickers and Rindge formations. The high-rate gravel pack operations were conducted by Halliburton with PXP oversight. The conditions of the high-rate gravel packs are representative of other high-rate gravel packs previously conducted across the field, and are also representative of future high-rate gravel pack jobs that could be expected to be conducted at the oil field.

The maximum applied pressure during both high-rate gravel packs was 1,900 pounds per square inch (psi). In comparison the high-volume hydraulic fracturing projects described in ES.6.1, had an average treatment pressure of 2,971 psi (VIC1-330) and 6,914 psi (VIC1- 635). The high-rate gravel pack influenced the zone within 125 feet of the well within the target oil-producing zone;

whereas, the high-volume hydraulic hydraulic fractures affected areas up to 1,100 horizontal feet from the subject wells (2,200 feet in length tip to tip, Halliburton 2012).

ES.6 Monitoring Conducted During Hydraulic Fracturing and High-Rate Gravel Packing at Inglewood Oil Field

ES.6.1 Hydrogeology, Water Quantity and Quality

In all parts of the world, fresh (not salty) groundwater lies at relatively shallow depths. At greater depths the water is saline, not drinkable, and is sometimes called formation water. The United States Environmental Protection Agency (USEPA) recognizes this distinction in the Safe Drinking Water Act which requires that the shallow, fresh water is protected from contamination by deeper, saline formation water. In most of the Los Angeles Basin, the base of the fresh water zone, below which saline formation water is found, is defined by the top of a marine geological unit called the Pico Formation. The zone at the Baldwin Hills considered to potentially contain fresh groundwater is from the ground surface to a depth of approximately 500 feet. Below approximately 500 feet, a "hydrocarbon seal" (or a nearly impermeable geologic formation) separates the fresh water zone from the oil producing zones and saline water containing formations below.

Nineteen groundwater borings have been drilled on the Inglewood Oil Field since 1992, only eleven of which encountered any water. Where water is encountered, it can range from 30 to 500 feet below ground surface, in zones less than 10 feet thick. The four deepest wells were installed to reach the "base of the fresh water zone," that is, the top of the Pico Formation. As such, current understanding of groundwater hydrogeology and water quality at the Inglewood Oil Field is based on a well-documented investigation of the entire zone beneath the surface that has any potential to contain fresh water. Although many borings for wells did not encounter any water, those that did were found to pump dry rapidly at low flow rates and recharge slowly. These data indicate that the water bearing zone from which they draw is limited in extent and not suitable for a water supply that could serve the oil field or the surrounding community.

None of these thin, discontinuous water-bearing zones within the Inglewood Oil Field connect to the aquifers of the Los Angeles Basin (USGS 2003, DWR 1961, this Study). The observed zones are perched within the folded and faulted confines of the field. In groundwater models of freshwater flow in the Los Angeles Basin aquifer systems prepared by the U.S. Geological Survey (USGS 2003), the Baldwin Hills are modeled as a "no flow" zone since the sediments beneath the Baldwin Hills are disconnected from the regional aquifers and groundwater flow is discontinuous across the Baldwin Hills. The California Department of Water Resources (DWR 1961) states "the Baldwin Hills form a complete barrier to groundwater movement, where the essentially non-water bearing Pico Formation crops out" (DWR 1961). The findings of the studies and ongoing groundwater monitoring of the Baldwin Hills commissioned by PXP and summarized in this study are in complete agreement with the findings of the USGS and DWR. Due to this lack of water in the geological formations beneath the Baldwin Hills, groundwater in the area is not suitable as a water supply (DWR 1961, USGS 2003, County of Los Angeles 2008).

The local community does not receive water from any formations beneath the Baldwin Hills, or from any well within 1.5 miles of the Baldwin Hills. Rather, approximately two-thirds of the community's water is delivered from sources in northern California (the Sacramento - San Joaquin River Delta) or sources such as the Colorado River. The nearest groundwater supplies outside the

Baldwin Hills are all very limited in supply and are geologically separated from the subsurface geologic formations of the Inglewood Oil Field. Therefore, activities associated with oil and gas development in the Baldwin Hills do not affect the community's drinking water supply.

All of the water service providers to the communities surrounding the Baldwin Hills must test their water from local wells at least four times a year and report the results to the water users. These reports indicate that the community receives water that meets USEPA's drinking water standards. Ongoing (four times per year) monitoring corroborates that this portion of the water supply meets these standards. The most recent data posted by the water purveyor covers the high-volume hydraulic fracturing that occurred in September 2011, as well as earlier conventional hydraulic fracturing and high-rate gravel packs. All public water supplies in California must also meet these requirements.

ES.6.1.1 Groundwater Monitoring Before and After Hydraulic Fracturing and High-Rate Gravel Packing

The Los Angeles Regional Water Quality Control Board (LARWQCB) Water Quality Control Plan, or Basin Plan, establishes beneficial uses of surface and groundwater in the Los Angeles Basin. Based on the State Board Resolution No. 88-63, "Sources of Drinking Water Policy", all groundwater in the state must be considered a potential source of drinking water, and carry a beneficial use designation of Municipal Supply (or MUN). This designation does not imply that the groundwater has sufficient capacity to support a municipal supply, presently or in the future. The designation addresses requirements to maintain groundwater quality in the sense of meeting drinking water standards.

As such, any water that may be encountered beneath the Inglewood Oil Field, regardless of its ability to actually supply water, must carry the beneficial use designation of MUN. Groundwater is collected from monitoring wells within the oil field, and is analyzed on a quarterly basis. A review of quarterly groundwater monitoring reports for 2010 and 2011 indicates that the perched, isolated groundwater meets the water quality requirements for MUN waters with the exception of arsenic, the concentrations of which are likely due to the high background level that naturally occurs in Southern California (Chernoff et al. 2008, Welch et al. 2000). As documented by USEPA, when "compared to the rest of the United States, western states have more systems with arsenic levels greater than USEPA's standard of 10 parts per billion (ppb)" (USEPA 2012a). Arsenic delineation maps produced by the USGS in 2011 have documented increased levels of arsenic in both the County of Los Angeles and Southern California as a whole (Gronberg 2011).

These data are also consistent with soils data from the 2008 California Department of Toxic Substance Control (DTSC) memo "Determination of a Southern California Regional Background Arsenic Concentration in Soil" (Chernoff et al. 2008). Areas in Southern California have been shown to have higher than average levels of arsenic present in soil and thus, through the release of naturally occurring arsenic in sediments, levels can be inferred to also be higher than average in groundwater resources throughout Southern California.

Monitoring was also conducted in April and August, three months and seven months after high-volume hydraulic fracturing and high-rate gravel packs conducted in January. The water was analyzed for the following constituents: pH, total petroleum hydrocarbons (TPH), benzene, toluene, ethylbenzene, total xylenes, methyl tertiary butyl ether (MTBE), total recoverable petroleum hydrocarbons (TRPH), total dissolved solids (TDS), nitrate, nitrite, metals, and biological oxygen demand (BOD5). These compounds include those used in hydraulic fracturing. The results of this monitoring were consistent with past groundwater monitoring and results. Groundwater will continue to be collected, analyzed, and reported consistent with the CSD and irrespective of when hydraulic fracturing and high rate gravel pack operations are conducted in the future.

Based on comparison of two sampling rounds after the high-volume hydraulic fracturing operations and high-rate gravel pack operations with the quarterly sampling rounds conducted prior to the operations, none of the analytical results indicated constituents above the state drinking water standard, with the exception of arsenic, which occurs naturally in soil and rock formations in Southern California. For the compounds detected, the concentrations after hydraulic fracturing were within the range of concentrations detected during the baseline period before hydraulic fracturing. The only exception was a minor increase in chromium from one well, MW-7 (2.7 to 3.0 µg/L, both results were well below the 50 µg/L state standard). Chromium is not associated with hydraulic fracturing additives.

Several new groundwater monitoring wells were installed after high-volume hydraulic hydraulic fracturing and high-rate gravel packing operations were conducted. Accordingly, the pre and post-hydraulic fracturing and high-rate gravel packing data in these specific wells cannot be compared. However, we can compare the results in the new wells with the pre- and post-hydraulic fracturing and high-rate gravel packing results from the pre-existing wells. In comparing the results of groundwater collected from new wells installed after hydraulic fracturing and high-rate gravel packing with the existing wells, the results were also consistent. No compounds violated the drinking water standard except for arsenic, as was the case with the pre-existing wells. The new wells were within the ranges of values detected in the pre-existing wells, with the new wells ranging to slightly higher total dissolved solids, zinc, and biological oxygen demand. The total dissolved solids and zinc may be due to conditions at depth, closer to the saline formation water. The biological oxygen demand is not associated with hydraulic fracturing additives.

Groundwater monitoring shows similar groundwater quality results before and after high-volume hydraulic fracturing and high-rate gravel packing. The Inglewood Oil Field's groundwater is not a source of drinking water. The groundwater bearing water bodies of the Baldwin Hills are geologically isolated from the nearest groundwater wells used for the municipal supply; and, two-thirds of the community water supply is from Northern California (the Sacramento-San Joaquin Delta) or the Colorado River. The local community does not receive water from closer than 1.5 miles to the Baldwin Hills. Community water supply is tested on a quarterly basis by the water purveyor, meets drinking water standards, and the results are publicly available.

ES.6.2 Well Integrity

During each stage of the hydraulic fracturing and high-rate gravel pack operations, the well casing of the subject well is tested in order to ensure integrity prior to injection of fracturing fluids (Halliburton 2012). Information about the well integrity tests is described in the post job reports. Well integrity testing is done by pressure testing the well up to 70 percent of the strength of the casing, in conformance with field rules established by the DOGGR. Offset wells, production wells, and injection wells are also tested for proper zonal isolation (i.e., annular cement) prior to any hydraulic fracturing operations. All measurements of well integrity during the hydraulic fracture and high-rate gravel pack operations conducted for this study indicated that there were no losses in pressure. The offset wells easily withstood the pressures of high-volume hydraulic fracturing; and no evidence of damage to the offset well was demonstrated by the pressure testing. The applied energy of the high-volume hydraulic fracturing rapidly decreases away from the completed well, and as such surrounding wells would not be adversely affected by the operation.

In addition to the well-integrity tests conducted for the high-volume hydraulic fracturing and high-rate gravel pack operations, active injection wells at the Inglewood Oil Field are surveyed annually (and pressure tested after each well work) per DOGGR requirements pursuant to CCR, Chapter 4, Article 3, §1724.10(j)3. PXP also monitors active injection wells weekly for injection rates and pressures (what also indicates the integrity of the wellbore and confinement of fluids to the injection zone) and reports to DOGGR on a monthly basis, pursuant to CCR Chapter 4, Article 3, §1724.10(c).

Tests conducted before, during and after the use of high-volume hydraulic hydraulic fracturing and high-rate gravel packing showed no impacts on the integrity of the steel and cement casings that enclose oil and gas wells.

ES.6.3 Containment of High-Rate Gravel Packs and High-Volume Hydraulic Fractures to the Target Zones

The measured distribution of fractures caused by the high-rate gravel pack completions were all less than 250 feet from the well, and were confined to the perforated zone within the Vickers and Rindge formations. The measured distribution of fractures from the high-volume hydraulic fracture completions were less than 1,100 feet in length from the well, and, with minor exceptions, were contained within the target zone (Halliburton 2012). For the few fractures that were outside the Nodular Shale target zone, they were deeper (with the oil-bearing Sentous Shale) and not filled with proppant. They therefore would reseal after the cessation of the increased pressure of hydraulic fracturing. Fractures grew either horizontally from the well or at angles less than 20 degrees depending on the local angle of the geological formations. Vertical fracture growth was very limited. The high-volume hydraulic fracture completions were conducted between 8,000 and 9,000 feet below the ground surface, and fractures did not form at shallower depths than approximately 8,000 feet below the ground surface. By comparison, the deepest groundwater encountered that had relatively low salinity was at a depth of 500 feet below the ground surface, corresponding to the base of fresh water beneath the Inglewood Oil Field, 1.5 miles above the hydraulic fracturing.

The results of microseismic monitoring indicate that fractures created during the high-volume hydraulic hydraulic fracturing operations were contained to the deep Nodular Shale with the exception of a minor few that were not filled with proppant. The fractures were all greater than 7,500 feet below the designated base of fresh water. The fractures created during all high-rate gravel packs were confined to the target zones.

ES.6.4 Subsurface Occurrence of Methane

Most of the oil and natural gas in the Los Angeles Basin lies trapped beneath both shales and faults, allowing it to accumulate at depth. However, some surface seeps do occur, as at the La Brea Tar Pits, and were the initial targets in the development of the Los Angeles Basin fields. In accordance with the CSD, field-wide methane monitoring is conducted at the Inglewood Oil Field on an annual basis to gauge for shallow occurrences of methane, and detections are investigated to determine the cause and remediate it.

Due to the potential of methane gas migration from the naturally occurring, prolific oil and gas province underlying the entire Los Angeles Basin, the City of Los Angeles has established a zoning ordinance identifying two zones, a Methane Zone and a Methane Buffer Zone, with special requirements for new construction, existing construction, and methane monitoring. The Baldwin Hills are outside the City of Los Angeles, and therefore are not classified on the methane map; however, they are adjacent to such zones. Although past methane detections have either been low or associated with a well to be re-abandoned, methane concentrations beneath portions of the field would reflect the relatively high background levels of methane in the Los Angeles Basin. All shallow detections of methane associated with the monitoring have been biogenic, based either on the composition (almost pure methane) or isotopic composition. Monitoring of shallow methane after high-volume hydraulic fracturing and high-rate gravel packing did not detect increases in soil gas methane concentrations.

Groundwater was not measured for methane prior to high-volume hydraulic fracturing or high-rate gravel packing. Samples collected after high-volume hydraulic fracturing detected dissolved methane in all but one well (MW-7), with concentrations up to 9.7 mg/L methane; all but two of the detections were less than 0.2 mg/L. Methane is not toxic and so there is not a drinking water standard established for it in water. There are few standards that have been promulgated for the nuisance effects of methane; the most widely applied are those of the U.S. Office of Surface Mining and the U.S. Bureau of Land Management. The highest value measured in groundwater at the Inglewood Oil Field is within the levels considered safe (10 mg/L), and well within levels that would actually trigger contingency actions (28 mg/L). The City of Los Angeles methane zoning ordinance does not address methane in groundwater; the ordinance only addresses levels in soil gas and applies construction standards as contingencies. Based on isotopic analysis of the dissolved methane in groundwater, it is thermogenic (from the oil-bearing formations) in origin, whereas detections in shallow soil gas are biogenic in origin. There are shallow occurrences of oil in the Investment Zone, within the Pico Formation. Since these zones are in closest proximity to the water bearing zones, and the occurrence of methane is pervasive in the monitoring results, it does not appear to be related to oil and gas production activity but to the natural occurrence of the underlying oil and gas. The occurrence is also not correlated to the locations of high-volume hydraulic fracturing or high-rate gravel packing.

The results of methane testing in soil and groundwater showed no influence from high-volume hydraulic fracturing or high-rate gravel packing.

ES.6.5 Slope Stability, Subsidence, Vibration, and Induced Seismicity

Slope stability is a primary geologic concern in the Baldwin Hills, and is addressed by conditions in the CSD that require ongoing monitoring. The California Department of Conservation, Division of Mines and Geology (CDMG), has studied the occurrence of slope instabilities and related geological issues of the Baldwin Hills (CDMG 1982). The study notes widespread damage from slope failures caused by rains in 1969, 1978, and 1980, and less widespread damage in other years. The study concludes that slope stability is a substantial problem in the Baldwin Hills because the terrain that has been developed for residential use consists mostly of steep natural slopes underlain by soft sedimentary rocks that are prone to land sliding and erosion. In addition, many of the communities in the Baldwin Hills were developed prior to the enactment of strict grading codes by local government, and therefore lack adequate protections against these natural geological conditions. The CDMG study notes that the Inglewood Formation is particularly susceptible to slope instability because the surficial soils developed on the formation are clay-rich. The study also notes that the Culver Sands are particularly susceptible to erosion. Monitoring for vibration and subsidence did not detect a change due to hydraulic fracturing or high-rate gravel packing. As such, hydraulic fracturing and high-rate gravel packing would not affect surface slope stability.

Subsidence is another geological concern in the Baldwin Hills. As described in the Baldwin Hills CSD EIR, prior to 1971, the maximum cumulative subsidence of any of the areas along the Newport-Inglewood fault zone was centered over the Inglewood Oil Field. Injection of produced water into the active producing zones began in 1957 to counteract this subsidence, and since 1971, water injection into the shallow production horizons has effectively eliminated subsidence associated with oil and gas production. The oil field has an ongoing program of annual subsidence monitoring that is reported in the framework of the CSD. To date, no changes in ground surface are attributed to oil and gas production activities. In evaluating pre- and post-hydraulic fracturing and high-rate gravel packing subsidence, none were attributed to the hydraulic fracturing or high-rate gravel packing.

ES.6.5.1 Subsidence and Ground Movement Monitoring during Hydraulic Fracturing

The CSD requires an annual ground movement survey at the Inglewood Oil Field. Surveying for both vertical and horizontal ground movement is accomplished using satellite-based GPS technology. Accumulated subsidence or uplift is measured using repeat pass Differentially Interferometric Synthetic Aperture Radar technology. The data are then evaluated to determine whether oil field operations (oil production and/or produced water injection volumes) are related to any detected ground motions or subsidence. Baseline survey points were collected in 2010 and then resurveyed in January 2011 and February 2012 (following the hydraulic fracturing operations of VIC1-330 and VIC1-635 and the high-rate gravel packing operations of TVIC 221 and TVIC 3254) to calculate annual subsidence or uplift at each point (Fugro NPA 2011, Psomas 2012). Based on a comparison of the ground movement survey results in 2011 and 2012 to operations production and injection records over the same time periods, there is no correlation between measured elevation changes and field activities.

The high-volume hydraulic fracturing and high-rate gravel packing had no detectable effect on ground movement, vibration, seismicity or subsidence, based on the results of studies conducted before and after the activities. As such, there would also be no detectable effect on slope stability.

ES.6.5.2 Vibration and Induced Seismicity

PXP retained Matheson Mining Consultants, Inc. to conduct vibration and ground surface monitoring during the high-volume hydraulic fracturing operations at the VIC1-330 and VIC1-635 wells, and at TVIC-221 and TVIC-3254 for the high-rate gravel pack jobs.

Vibration records for the VIC1-330 and VIC1-635 wells were collected using four and eight seismographs, respectively, installed at different locations in relation to the high-volume hydraulic fracture operations. The TVIC-221 and TVIC-3254 wells are directly adjacent to one another; therefore, the same seismographs were used to monitor the high-rate gravel packs on these wells. Based on analysis of the seismograph data, Matheson Mining Consultants, Inc. concluded that no seismic activity was produced by any of the high-volume hydraulic fracturing or high-rate gravel pack operations. In addition to the seismic monitoring conducted by Matheson Mining Consultants, Inc., seismic data collected by the permanently installed California Institute of Technology (Cal-Tech) accelerometer (seismometer) at the Baldwin Hills was reviewed for the time periods before and during the high-volume hydraulic fracturing and high-rate gravel pack operations. Background levels range from 0.0003 to 0.0006 inch per second (ips); however, random spikes occur in the record approximately every two to three hours. These spikes are likely related to local traffic or some other passing noise source, and are common in urban areas. The data collected from the seismograph during the VIC1-635 operation showed two minor spikes during the time period reviewed (the largest measuring 0.0012 ips). Analysis of the data by Dr. Hauksson, a Senior Research Associate in Geophysics with the Cal-Tech Seismological Laboratory, concludes that these spikes are not indicative of any seismic events above background levels were recorded (Matheson Mining Consultants, Inc. 2012a). The data collected from the seismograph during the TVIC high-rate gravel pack operations showed some spikes during the time period reviewed but no significant signals above the background levels. No data above background levels were recorded on the Cal-Tech seismograph during the VIC1-330 operation.

Petersen and Wesnousky (1994) evaluated all seismic events greater than Magnitude 2 on the Newport-Inglewood Fault zone, and determined that most epicenters are located at depths between 3.5 miles and 12 miles deep (Petersen and Wesnousky 1994, Hauksson 1987). In comparison, the waterflood operation at the Inglewood Oil Field extends to depths of up to 3,000 feet (0.57 mile) and the deepest hydraulic fracturing occurs at less than 10,000 feet depth (1.9 miles). Therefore, oil field operations are much shallower than the zones typically associated with earthquake epicenters along the Newport-Inglewood Fault zone.

Results of studies conducted before and after high-volume hydraulic fracturing and high-rate gravel packing operations indicate that the operations had no detectable effect on vibration, and did not induce seismicity at the surface.

ES.6.6 Noise and Vibration

To address concerns regarding perceptible vibration and noise during high-volume hydraulic fracturing operations, PXP commissioned Behrens and Associates, Inc., a firm specializing in noise and vibration studies, to measure produced vibration during the VIC1-330 and VIC1-635 high-volume hydraulic fractures and the TVIC-221 and TVIC-3254 high-rate gravel pack events. The ground-borne vibration survey for each event was completed while all equipment was operated under normal loads and conditions.

The high-volume hydraulic fracturing treatment on September 16, 2011, was completed on the VIC1-330 well, located in the northwestern portion of the field. Measured levels indicate that the maximum ground-borne vibration produced during the operation was 0.006 inch per second, as measured 40 feet from the operation. At 160 feet from the operation, measured vibration was 0.001 inch per second. Both of these levels are imperceptible to humans (Behrens and Associates, Inc. 2011).

In addition to ground-borne vibration measurements, Behrens and Associates, Inc. also took sound level measurements during the high-volume hydraulic fracturing operation at VIC1-635 and the high-rate gravel pack operations at TVIC-221 and TVIC-3254 using a calibrated sound level meter. The microphone was set at 5 feet above ground surface. The measured noise level at 100 and 200 feet from the operation at VIC1-635 was 68.9 and 68.4 decibels (dBA), respectively (Behrens and Associates, Inc. 2012a), The measured noise level at 100 and 200 feet from the TVIC-221 and TVIC-3254 operations was 68.1 dBA and 63.5 dBA, respectively (Behrens and Associates, Inc. 2012b). These measured noise levels are all in compliance with CSD limits.

The noise and vibration associated with the high-volume hydraulic fracturing and high-rate gravel pack operations did not exceed CSD limits.

ES.6.7 Air Emissions

Air emissions on the Inglewood Oil Field are monitored as described in an Air Monitoring Plan in accordance with Section E.2(d) of the Baldwin Hills CSD. This plan requires monitoring for hydrogen sulfide and total hydrocarbon vapors. It also requires that drilling or completions operations shut down if monitoring detects concentrations of hydrogen sulfide greater than 10 ppm or hydrocarbon concentration of 1,000 ppm or greater. Vehicle use for on-road and off-road vehicles and construction equipment is also regulated by the CSD under Sections E.2(j) through E.2(n).

Mass emissions of criteria pollutants and greenhouse gases (GHG) for off-road equipment and on-road vehicles were calculated using emission factors published by the South Coast Air Quality Management District (SCAQMD 2008) and USEPA (2011a, 2011b). The project schedule and equipment/vehicle list provided by PXP and Halliburton served as the basis for the analysis. The results of the analysis are presented in the emissions summary tables contained in section 4.7 of this study. These levels are consistent with those considered in the CSD.

Air emissions associated with high-volume hydraulic fracturing and high-rate gravel packing were compliant with the regulations of the South Coast Air Quality Management District and the CSD.

ES.6.8 County of Los Angeles Department of Public Health Study

The County of Los Angeles Department of Public Health (LAC DPH) conducted a community health assessment on the population living in communities surrounding the Inglewood Oil Field in 2011. The assessment was designed to determine if health concerns in the communities surrounding the Inglewood Oil Field reflect a higher than expected rate or an unusual pattern of disease. The report was sent to three external peer reviewers who found it to be technically sound.

The conclusions of the health assessment indicate that the health of the community adjacent to the Baldwin Hills is not statistically different from that of Los Angeles County as a whole, including cancer rates in the community. The report acknowledges that the data cannot determine adverse health effect below its detection limit, nor can the data address the contribution of other, non-quantifiable health-related issues such as smoking, lack of exercise, and social determinants of health.

Conventional hydraulic fracturing and high-rate gravel packing have occurred at the field since 2003, along with other oil and gas development activity. Based on the results of the health assessment, these activities had no detectable adverse effect on the health of the local community.

The health assessment recommends careful monitoring of the oil field operations to ensure compliance with regulations and standards to protect community health and safety. In compliance with the CSD, such monitoring occurs via Environmental Compliance Coordinator weekly inspections and an annual Environmental Quality Assurance Program (EQAP) audit.

The Los Angeles County Health Study found no detectable health consequences to the local community from oil and gas development (including hydraulic fracturing and high-rate gravel packing) at the Inglewood Oil Field. The study recommends careful monitoring of the oil field operations to ensure compliance with regulations and standards to protect community health and safety.

ES.6.9 Issues Associated with Hydraulic Fracturing in Shale Gas and Relevance of Inglewood Oil Field Hydraulic Fracturing Study Results

Since high-volume hydraulic fracturing has been used for shale gas development in the northeastern United States, there has been extensive media coverage of controversies surrounding its use. Although most of the news has been about the development of shale gas, tight sands and coalbed methane deposits rather than the type of oil and natural gas development that occurs at the Inglewood Oil Field, community outreach conducted as part of this study has indicated that many of the concerns surrounding shale gas development are shared by the local community and applied to oil development. The primary environmental and health issues of concern associated with hydraulic fracturing operations include:

- Potential for contamination of groundwater, including drinking water supplies, and gas migration;
- Environmental hazards associated with the chemical additives used during hydraulic fracturing operations;
- Potential for hydraulic fracturing operations to cause earthquakes;
- Issues related to well integrity; and,
- Air emissions and greenhouse gas emissions of hydraulic fracturing operations in comparison to regular oil field operations.

A description of each of these issues as they relate to hydraulic fracturing operations is provided in the study, along with the direct measurements taken at the Inglewood Oil Field to determine their relevance.

ES.7 Regulatory Perspective on the Inglewood Oil Field

The federal, state, and local laws, ordinances, regulations, and standards that govern oil field development throughout the United States require protections against the potential environmental impacts of the entire development process. These protections range from provisions in the Clean Air Act, Clean Water Act, Safe Drinking Water Act, Endangered Species Act, and through extensive California regulation addressing air quality, water resources, biological resources, and cultural resources, and at the local level. The Inglewood Oil Field is unusual in that it has much greater regulation and oversight of its operations than most other onshore oil fields as a result of the County of Los Angeles CSD.

The current national regulatory framework and government-sponsored studies of hydraulic fracturing are summarized in Section 5 of this Study to provide a national perspective to this Hydraulic Fracturing Study. Most of these studies address hydraulic fracturing associated with the development of shale gas, which is different than oil and gas development. Although the Inglewood Oil Field is not a shale gas field, and many of the concerns associated with the development of shale gas do not apply to the Inglewood Oil Field, the findings presented in Section 5 are intended to place concerns commonly seen in the news media in the local context of the Inglewood Oil Field.

The Baldwin Hills CSD, and the associated EIR, together address most of the issues that are part of a hydraulic fracturing operation, such as truck traffic, water use, community compatibility (noise, light and glare, etc.), air quality, and other environmental resource categories. In addition, the EIR evaluates cumulative impacts, and environmental justice. These two documents support this Hydraulic Fracturing Study, which evaluates the effects measured and monitored during the high-volume hydraulic hydraulic fracturing and high rate gravel packing operations conducted in 2011 and 2012, as well as past activities of this type. The Hydraulic Fracturing Study did not identify a new impact not analyzed in the EIR, nor did it identify impacts greater in significance than those analyzed in the EIR.

Exacting protective measures and close monitoring are required by the Baldwin Hills CSD and by county, regional and federal agencies. These field-specific reviews and public and agency interactions compel PXP to enforce real-time compliance with all environmental standards in the Inglewood Oil Field. The long history of oil production in the area provides operators with an excellent understanding of the local subsurface conditions and reduces standard risks and uncertainties that would be present in new operations.

Chapter 1

Introduction

Plains Exploration & Production Company (PXP) operates the Inglewood Oil Field, an approximately 1,000 acre area of the Baldwin Hills in Los Angeles County (Figure 1-1). Oil was discovered in the Baldwin Hills in 1924 by Standard Oil, and the oil field was operated by Chevron (successor company to Standard Oil), followed in 1990 by Stocker Resources, Inc., which was then acquired by Plains Resources, Inc. in 1992. PXP was incorporated in September 2002, and acquired all of Plains Resources, Inc.'s California operations, including the Inglewood Oil Field, in December 2002. PXP has operated the oil field since late 2002.

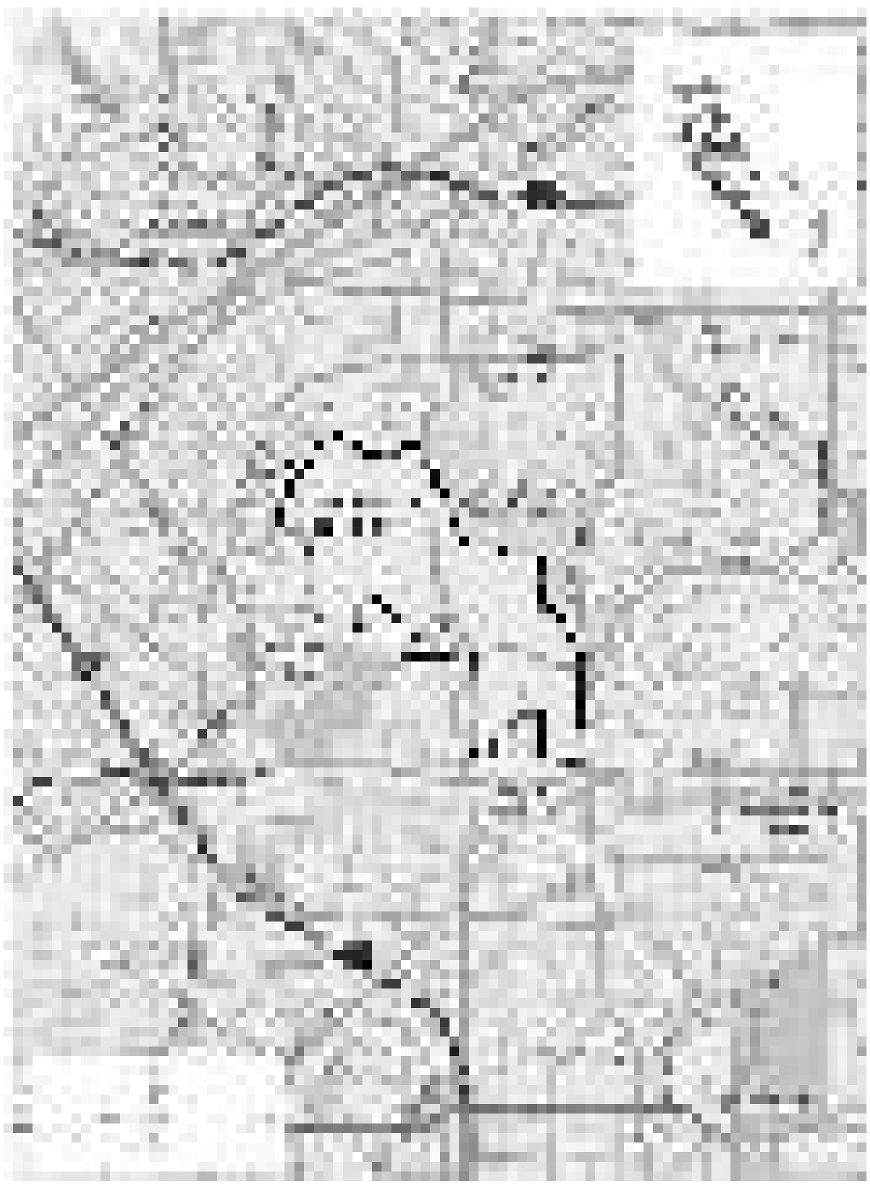

Figure 1-1 Regional Location Map

In October 2008, the County of Los Angeles (County) approved the Baldwin Hills Community Standards District (CSD), which created a supplemental district within the County to address the compatibility of oil production with adjacent urban land use. The CSD established permanent development standards, operating requirements, and procedures for the Los Angeles County portion of the Inglewood Oil Field. The northernmost areas of the field are within the city limits of Culver City, and PXP has voluntarily complied with the provisions of the CSD in that portion of the oil field as well.

Following adoption of the CSD, a lawsuit was filed against the County and PXP in late 2008, challenging the validity of the ordinance. The lawsuit was resolved through a Settlement Agreement that was signed on July 15, 2011 by the City of Culver City, Natural Resources Defense Council, Concerned Citizens of South Los Angeles, Citizens Coalition for a Safe Community, Community Health Council, the California Attorney General's Office, PXP, and the County of Los Angeles. The Settlement Agreement augments the protections contained in the CSD with 15 additional terms. This Hydraulic Fracturing Study is the direct result of Term 13, which states:

> *PXP shall pay for an independent consultant to conduct a study of the feasibility and potential impacts (including impacts to groundwater and subsidence) of the types of fracturing operations PXP may conduct in the Oil Field. The study will also consider PXP's historic and current use of gravel packing. Such study will be completed within twelve (12) months of the date of this Agreement. Such study and all the back-up information for such study shall be provided to a qualified peer reviewer selected by the County and PXP, who shall review the study, back-up materials, and conclusions for completeness and accuracy. PXP must provide the independent expert with all materials requested and reasonably necessary for an accurate and verifiable study. The peer reviewer will be provided with access to all the data and materials provided to the independent expert. The peer reviewer shall agree to keep all proprietary information confidential. If the peer reviewer determines that the study is materially inadequate, incomplete or inaccurate, it shall so advise PXP's consultant who will complete the study as reasonably recommended by the peer reviewer and provide the revised study to the peer reviewer within 90 days. Upon acceptance by the peer reviewer, the study and all supporting material, including comments by the peer reviewer, shall be forwarded to the County, DOGGR, the Regional Water Quality Control Board ("RWQCB"), CAP and Petitioners and be available to the public, with any proprietary information redacted.*

The Settlement Agreement Term 13 requires that the practice of high-rate gravel packing be included in this Hydraulic Fracturing Study. The process of high-rate gravel packing does not serve the same purpose as hydraulic fracturing and is a different process. Nonetheless, the practice is fully discussed in this study, in compliance with the agreement.

This study draws on several sources, including peer-reviewed literature, the Inglewood Oil Field CSD, the 2008 Environmental Impact Report (EIR) conducted for the CSD, data and analyses provided by Halliburton, who conducted the recent hydraulic fracturing operations at the field, and from numerous contractors performing monitoring studies before, during, and after the recent hydraulic fracturing and high-rate gravel pack test operations at the field.

In accordance with the Settlement Agreement, this study was reviewed by peer reviewers, jointly selected by the County and PXP. The peer reviewers, John Martin, Ph.D. and Peter Muller, Ph.D., C.P.G., were provided with the draft study and all reference materials. The peer reviewer's comments on the study, and their statement indicating that the revised study addressed all comments adequately and completely, thereby determining the study complete, is provided in Appendix A.

This Hydraulic Fracturing Study is organized as follows:

- **Chapter 1** presents the Study objectives.
- **Chapter 2** presents a brief summary of the distribution of oil production in the Los Angeles Basin providing regional perspective for the Hydraulic Fracturing Study. Chapter 2 also describes the geological setting at the Inglewood Oil Field, including the results of a 3-D depiction of the subsurface geology.
- **Chapter 3** describes oil and gas well drilling and completion methods. Hydraulic fracturing is a completion method and is described in the context of the overall well drilling and completion process. This Chapter describes hydraulic fracturing jobs performed at the Inglewood Oil Field by PXP, including a discussion of past, current, and potential future methods of hydraulic fracturing that have occurred, or may occur, at the field, and the two high-volume hydraulic fracture tests and two high-rate gravel pack tests conducted in 2011 and 2012.
- **Chapter 4** describes the setting, methods and results of extensive environmental monitoring conducted in conjunction with the hydraulic fracturing and high-rate gravel pack tests. Chapter 4 also includes a discussion of each environmental issue as raised in regulatory proceedings, agency studies, university studies, and in the media regarding high-volume hydraulic fracturing as applied in shale gas and tight sands reservoirs, principally in the northeastern United States, Texas, New Mexico and Colorado. Although exploration and development of shale gas differs from oil and gas production at the Inglewood Oil Field, the issues and concerns in states like Pennsylvania have helped shape public perceptions in the local community surrounding the oil field. The relevance of these issues to the Inglewood Oil Field is addressed in the context of the environmental monitoring conducted at the Inglewood Oil Field.
- **Chapter 5** describes the regulatory framework that governs hydraulic fracturing, drawing on information from across the country. This chapter also summarizes recent and ongoing studies by federal and state agencies on the environmental effects of hydraulic fracturing.
- **Chapter 6** provides the qualifications of the preparers of this document.
- **Chapter 7** provides supporting material and references, with complete citations and internet addresses for all sources used in this study.

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Chapter 2

Oil Production in the Los Angeles Basin and at the Inglewood Oil Field

2.1 Introduction

California is the fourth largest oil producing state in the U.S. (U.S. Energy Information Agency 2012), and the Los Angeles Basin is the richest oil basin in the world based on the volume of hydrocarbons per volume of sedimentary fill (Biddle 1991). Oil was first discovered in the area at the Brea-Olinda Oil Field in 1880, followed by the development of the Los Angeles City Oil Field in 1893, the Beverly Hills Oil Field in 1900, the Salt Lake Oil Field in 1902, the Long Beach Oil Field in 1921, the Inglewood Oil Field in 1924, the Wilmington Oil Field in 1932, and many others. Figure 2-1 shows the distribution of major oil fields in the Los Angeles Basin (refer to Table 2-1 for the names of each oil field corresponding to the numbers on the figure). The size of this province, and its continuing potential for new discoveries and technologies, ensure its continued development into the future.

The Los Angeles Basin represents, from a global perspective, the optimum conditions for the generation and entrapment of hydrocarbons (Barbat 1958, Gardett 1971, Wright 1987a). Barbat, in particular, considered eight major controls on the occurrence and amount of oil in different basins around the world. He concluded that, “no matter how the Los Angeles Basin may differ from other oil-producing areas, the differences favor the Los Angeles Basin” (Barbat 1958).

The unique abundance of oil in the Los Angeles Basin derives from a thick section of layered sediments and organic-rich materials. The relatively recent geologic, tectonic, and structural history of the region has provided an optimal thermal history to bring the organic-rich material into the “oil window,” the thermal regime that is ideal for oil production. This means that as the sediments and organic materials were buried, these source rocks reached high enough pressures and temperatures that they transformed to oil and natural gas.

The oil and natural gas migrated then from the source rocks, typically the Monterey shale formation, into overlying sandstones. The sandstones acted as reservoir rocks, accumulating and holding the oil and natural gas underground. The Los Angeles Basin is folded and faulted, and as a result, after migrating into the sandstone reservoir rocks, the oil and gas deposits become trapped by the folds and faults which are impermeable (do not allow for the passage of fluid), as well as relatively impermeable shale rocks which are also present. Therefore, the traps allow oil in the reservoir rocks to continue to accumulate at depth and not continue to migrate up to the surface. These traps are not ubiquitous, and in some locations oil continued to rise to the surface as seeps. The most famous local surface seep of oil is the La Brea Tar Pits.

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Legend

Oil Field

PLAINS EXPLORATION & PRODUCTION COMPANY

Figure 2-1

Location of Los Angeles Basin Oil Fields

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Table 2-1 Los Angeles Basin Oil and Gas Field

Number as shown on Figure 2-1	Field Name	Discovery Year	Estimated Ultimate Recovery		
			Oil[1] (kbbl mcf)	Gas[1] (mcf)	Oil-equivalent barrels (kbbl)
1	Alondra	1946	2,154	1,408	2,406
2	Anaheim (abd)	1951	4	--	4
3	Bandni	1953	5,969	15,469	8,738
4	Belmont Offshore	1948	68,500	41,931	76,006
5	Beta	1976	214,272	21,866	218,186
6	Beverly Hills	1900	164,131	215,163	202,645
7	Boyle Heights (abd)	1955	273	113	293
8	Brea-Olinda	1880	439,691	481,986	524,967
9	Buena Park, East (abd)	1942	197	20	201
10	Buena Park, West (abd)	1944	50	17	53
11	Cheviot Hills	1958	26,180	142,492	51,686
12	Chino-Soquel	1950	324	349	387
13	Coyote, East	1909	121,829	60,804	132,713
14	Coyote, West	1909	257,522	271,005	306,032
15	Dominguez	1923	276,846	387,394	346,190
16	El Segundo	1935	14,744	34,725	20,960
17	Esperanza	1956	1,331	699	1,456
18	Gaffey (abd)	1955	10	--	10
19	Howard Townsite	1947	6,162	27,810	11,140
20	Huntington Beach	1920	1,138,034	861,117	1,291,805
21	Hyperion	1944	798	209	835
22	Inglewood	1924	400,048	285,002	451,063
23	Kraemer	1918	3,925	1,078	4,118
24	Kraemer, Northeast (abd)	1953	unknown	--	--
25	Kraemer, West (abd)	1956	10	--	10
26	La Mirada (abd)	1946	25	10	27
27	Lapworth	1935	55	--	55
28	Las Cienegas	1960	36,349	55,550	75,293
29	Lawndale	1928	3,747	6,729	4,958
30	Leffingwell (abd)	1946	763	2,460	1,203
31	Long Beach	1921	927,428	1,087,440	1,121,773
32	Long Beach Airport	1954	11,572	35,003	17,838
33	Los Angeles City	1892	23,575	--	23,575
34	Los Angeles Downtown	1964	15,233	22,922	19,336
35	Los Angeles, East	1946	6,936	12,401	9,156
36	Mahala	1920	4,077	1,586	4,361

Table 2-1 Los Angeles Basin Oil and Gas Field

Number as shown on Figure 2-1	Field Name	Discovery Year	Estimated Ultimate Recovery: Oil[1] (kbbl mcf)	Gas[1] (mcf)	Oil-equivalent barrels (kbbl)
37	Montebello	1917	202,004	234,712	243,917
38	Newgate	1956	296	370	362
39	Newport	1922	187	259	233
40	Newport, west	1923	77,647	8,371	79,145
41	Olive	1953	3,020	1,209	3,236
42	Playa del Rey	1929	63,008	62,061	74,118
43	Portrero	1928	15,672	72,967	28,733
44	Prado-Corona	1966	1,632	5,192	2,561
45	Richfield	1919	217,340	173,067	248,319
46	Rosecrans	1924	83,339	166,330	113,112
47	Rosecrans, East	1959	202	234	243
48	Rosecrans, South	1940	8,835	20,661	12,533
49	Rowland (abd)	1931	2	--	2
50	Salt Lake	1902	53,683	211,894	91,612
51	Salt Lake, South	1970	10,091	4,503	10,897
52	Sansinena	1898	60,840	74,661	74,204
53	San Vicente	1968	21,043	19,433	24,522
54	Santa Fe Springs	1919	622,254	836,512	771,990
55	Sawtelle	1965	15,274	13,100	17,619
56	Seal Beach	1924	217,236	219,786	256,484
57	Sherman (abd)	1965	93	50	102
58	Sunset Beach	1954	6,910	9,591	8,627
59	Talbert (abd)	1947	126	4	127
60	Torrance	1922	247,562	162,573	276,593
61	Turnbell (abd)	1941	766	582	870
62	Union Station	1967	1,895	5,298	2,843
63	Venice Beach	1966	4,030	2,678	4,508
64	Walnut	1948	131	25	135
65	Whittier	1898	55,731	52,193	65,074
66	Whittier Heights, North (abd)	1944	85	84	235
67	Wilmington	1932	2,788,158	1,192,802	3,001,670
68	Yorba Linda	1930	94,781	2,174	95,170
	Totals		**9,074,637**	**7,628,134**	**10,439,275**

SOURCE: Biddle 1991
[1]EUR = estimated ultimate recovery
kbbl = thousand barrels
mcf = million cubic feet
abd = abandoned

2.2 Petroleum Geology of the Los Angeles Basin

The Los Angeles Basin is approximately 70 miles long and 10 miles wide. It is a coastal sediment-filled trough located between the Peninsular Mountain Ranges and the Transverse Mountain Ranges in southern California. The Los Angeles Basin contains the central part of the city of Los Angeles as well as its southern and southeastern suburbs (both in Los Angeles and Orange counties).

The Los Angeles Basin was formed in a strike-slip tectonic setting (crust generally sliding side-to-side along faults). Two different phases of motion were involved: early extension overlain on the strike-slip motion, followed by more recent compression overlain on a weakening strike-slip system. These phases of evolution are in part illustrated by the number of faults that cut other faults in the subsurface (Biddle 1991). The following paragraphs describe this history in greater detail.

The Los Angeles Basin originated as a depositional basin caused by crustal extension overlapping with the regional, right-lateral strike-slip movement. Prior to five million years ago the Los Angeles Basin was submerged approximately 5,000 feet under the waters of the Pacific Ocean. During this period the marine basin collected sand, silt, and clay sediment from the surrounding upland areas. As surrounding mountain ranges (including the San Gabriel and Santa Monica mountains) rotated clockwise, the crust cracked, extended, and released molten rock from below. Over time the crust thinned and formed a basin, or bowl, with boundaries formed by the San Gabriel Mountains, Santa Monica Mountains, Santa Ana Mountains, and the Palos Verdes Peninsula. Sand, silt and clay from the sea and ancient rivers poured into the bowl-shaped depression. The sedimentary formations resulting from this deposition extend more than 30,000 feet downward before reaching bedrock.

The more recent history of faulting represents shortening of the basin caused by compression (counteracting the earlier extension), and a reduction in the amount of strike-slip motion, beginning approximately five million years ago. Compression of the basin created thrust faults. A thrust fault is a type of a break in the earth’s crust in which older rock is uplifted over younger rock material. In the Los Angeles Basin, faults of this type uplifted the sediments and rock that had once lain at the ocean floor and brought them to the surface. This rock from the ocean floor consisted of alternating layers of sandstones and shales that had also previously folded and faulted. As it rose above sea level, this pile of sediment began forming the Los Angeles Basin.

Each of these phases of activity affected the oil producing characteristics of the basin. The extensional phases created a container into which sediments poured: both the Monterey shale, which is the source of the hydrocarbons, and the overlying sedimentary rocks that acted as the reservoir rocks once the oil formed and rose towards the surface. The more recent shortening of the basin has changed the overall shape of the basin, and modified the traps that allowed oil to accumulate in the reservoir sediments. The rapidly-subsiding, deep, Los Angeles Basin formed at the right time, and in the right place, with an appropriate geometry and thermal history, to form this uniquely rich oil province (Biddle 1991).

2.3 Petroleum Production in the Los Angeles Basin

The Los Angeles Basin is one of California's most prolific crude oil and natural gas regions. Figure 2-2 shows the amount of oil produced from Southern California oil fields since the discovery of the first field, Brea Olinda. Table 2-1, taken from Biddle (1991), summarizes the oilfields of the area, including the year of discovery and amount of oil and gas produced.

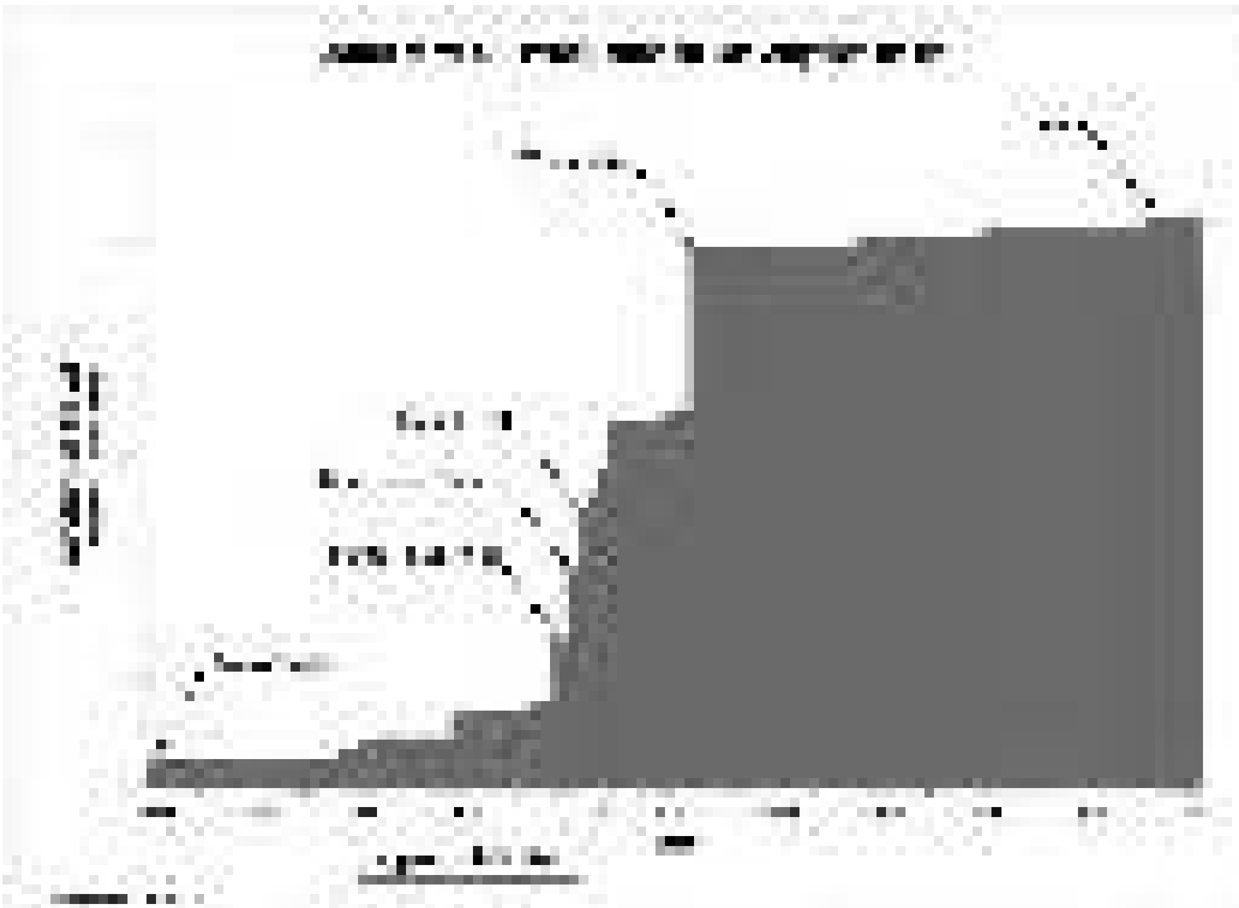


Figure 2-2 Cumulative Oil Production in the Los Angeles Basin

As of 2011, there are currently 42 active fields in the Los Angeles Basin. In 2011, the combined onshore and offshore oil production in California totaled approximately 197 million barrels, of which the Los Angeles Basin accounted for approximately 18 percent. Since 2007, an average of 2,700 wells has been drilled statewide annually (DOGGR 2011). Figure 2-3 shows the number of barrels of oil produced annually in California over the past decade (DOGGR 2007, 2011).

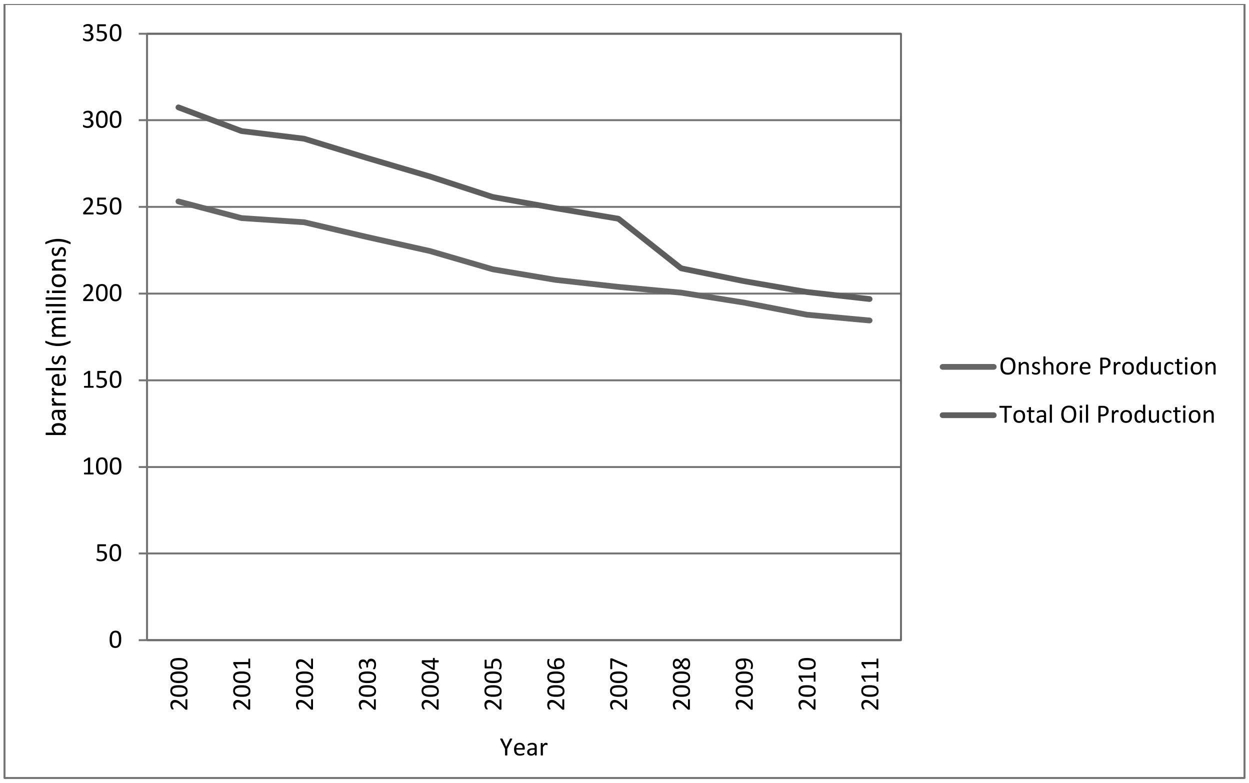


Figure 2-3 California Oil Production Since 2000

2.4 Petroleum Geology and Production at the Inglewood Oil Field

The approximately 1,000-acre Inglewood Oil Field is one of the largest contiguous urban oil fields in the United States. The Inglewood Field was discovered in 1924 and has produced an estimated cumulative production of 400 million barrels of oil. Oil and natural gas produced from the field is sold and used entirely in California. The oil field is adjacent to Culver City and the Los Angeles County communities of Baldwin Hills, View Park, Windsor Hills, Blair Hills and Ladera Heights. As of 2010, the U.S. Census Bureau estimated Los Angeles County's population to be 9.8 million. The area surrounding the field had a population of 65,892 in 2000. The population of this area has since remained relatively stable in comparison to the 2000 census data (U.S. Census Bureau 2010).

The Inglewood Oil Field was first commercially produced by Standard Oil in 1924, when livestock grazing (primarily by sheep) was the prevailing economic use of the land. The cultivated croplands had been reclaimed from the low-lying swampy terrain (cienegas) in the gently sloping portions of the Los Angeles Basin that surrounded the Baldwin Hills. Many of these lands were gradually converted to residential suburbs. With the incorporation of the City of Inglewood, residential development was spurred by transportation improvements, including the growth of highway network that transformed farmlands and displaced brick making industrial areas to the south of the Baldwin Hills.

In Culver City, both residential development and the foundation of movie studios and their associated supporting industries encroached upon the foothill slopes of the Baldwin Hills from the west and northwest. The northeastern and eastern sides of the Baldwin Hills were encroached upon by the westerly spread of the suburban growth of the City of Los Angeles (County of Los

Angeles 2008). Chilingar and Endres (2005) evaluate urban encroachment on active and inactive oil fields, primarily in the Southern California area. They conclude that "*a clear case is made for the urgent need for closer coordination and education by the petroleum industry of the local government planning departments…and in establishing mitigation measures for dealing with long-term environmental hazards*". The Baldwin Hills CSD, the associated EIR, and this Hydraulic Fracturing Study are examples of this advice put into practice.

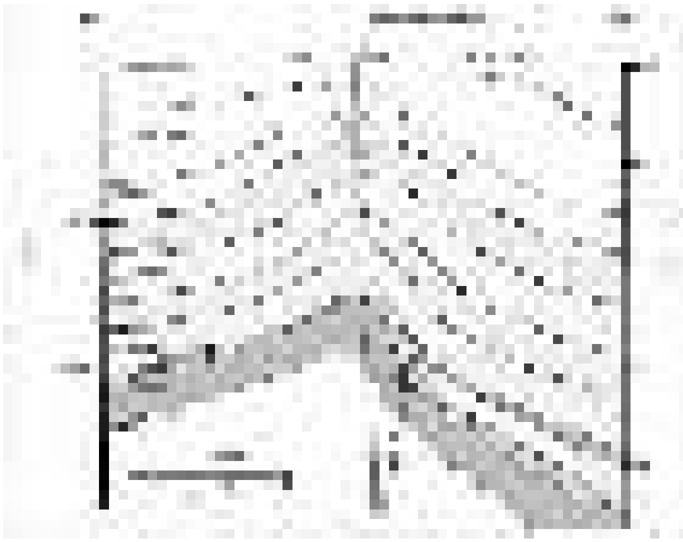
2.4.1 Inglewood Oil Field Geology

Overview

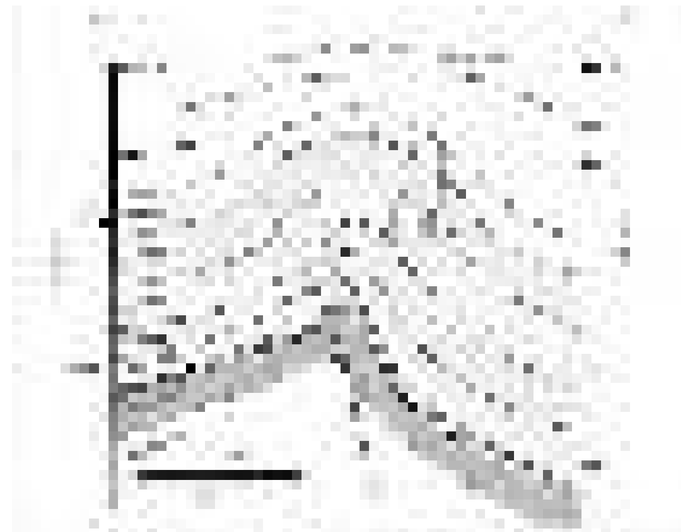
The Baldwin Hills form part of a chain of low hills along the Newport-Inglewood Fault Zone. The Baldwin Hills are the highest of these hills, reaching an elevation of 511 feet above mean sea level. The hills are in sharp relief against the relatively flat Los Angeles Basin, and include rolling hills cut by canyons and gullies. The northern flank of the Baldwin Hills has been deeply incised by erosion while the southern flank slopes gently to the Torrance Plain and Rosecrans Hills.

Figure 2-4 provides three geologic cross sections illustrating the sub-surface geology (cross section locations shown in Figure 2-6), while Table 2-2 provides details regarding the thickness of each formation. The southernmost cross section in Figure 2-4 shows the Newport-Inglewood Fault as cutting all the petroleum-producing units at the field. Moving north to the central part of the field, the cross section depicts the dissipation of the Newport-Inglewood Fault as it approaches the relatively east-west Santa Monica fault further to the north. At depth, the Newport-Inglewood Fault transitions to the series of folds and thrust faults at depth. Moving to the northernmost cross section, the Newport-Inglewood Fault is no longer present, and the movement here and further to the north is likely accommodated by a combination of folds and thrust faults. This depiction of the geology is based on the data collected by well drilling and by seismic surveys, and is described in Wright (1991).

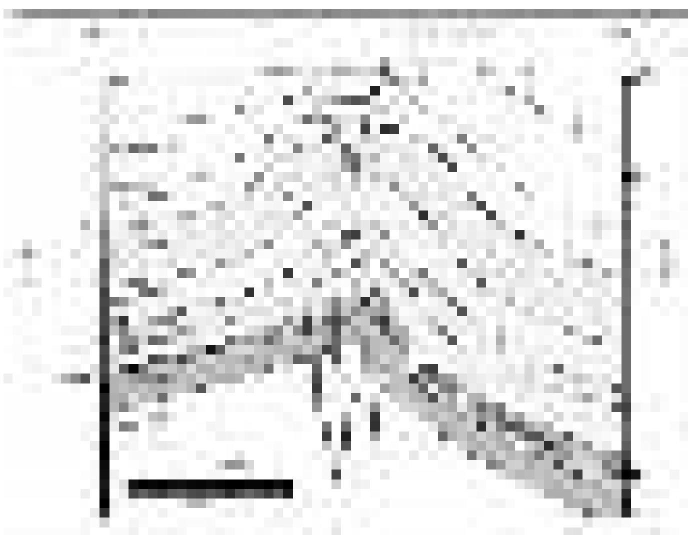
The Baldwin Hills have been uplifted by folding and faulting of the underlying geological formations. A northwest-trending anticline (upward-directed fold) is developed in sediments of Tertiary and Pleistocene age (23 million to 1.8 million years ago–see Table 2-3) beneath the Baldwin Hills. Two principal northwesterly trending, nearly parallel faults offset the central portion of the hills, developing a down-dropped trench, or graben, across the crest of the anticline. The more easterly of the two structures is the Newport-Inglewood Fault; the other fault is unnamed. Both faults are offset by secondary cross faults which trend northeast. The block east of the Newport-Inglewood Fault is composed of sediments of Pliocene age (approximately 5 million years ago) and older and is cut by several small unnamed faults. The modified geological timescale (Figure 2-5) summarizes the intensity of tectonic activity with time, as well as the major units that formed during each phase and the principal biological markers used to identify the units.



Schematic cross section of Inglewood Field, southern portion (Elliot 2009)



Schematic cross section of Inglewood Field, northern portion (Elliot 2009)



Schematic cross section of Inglewood Field, central portion (Elliott 2009)

Source: Wright, 1991

Figure 2-4

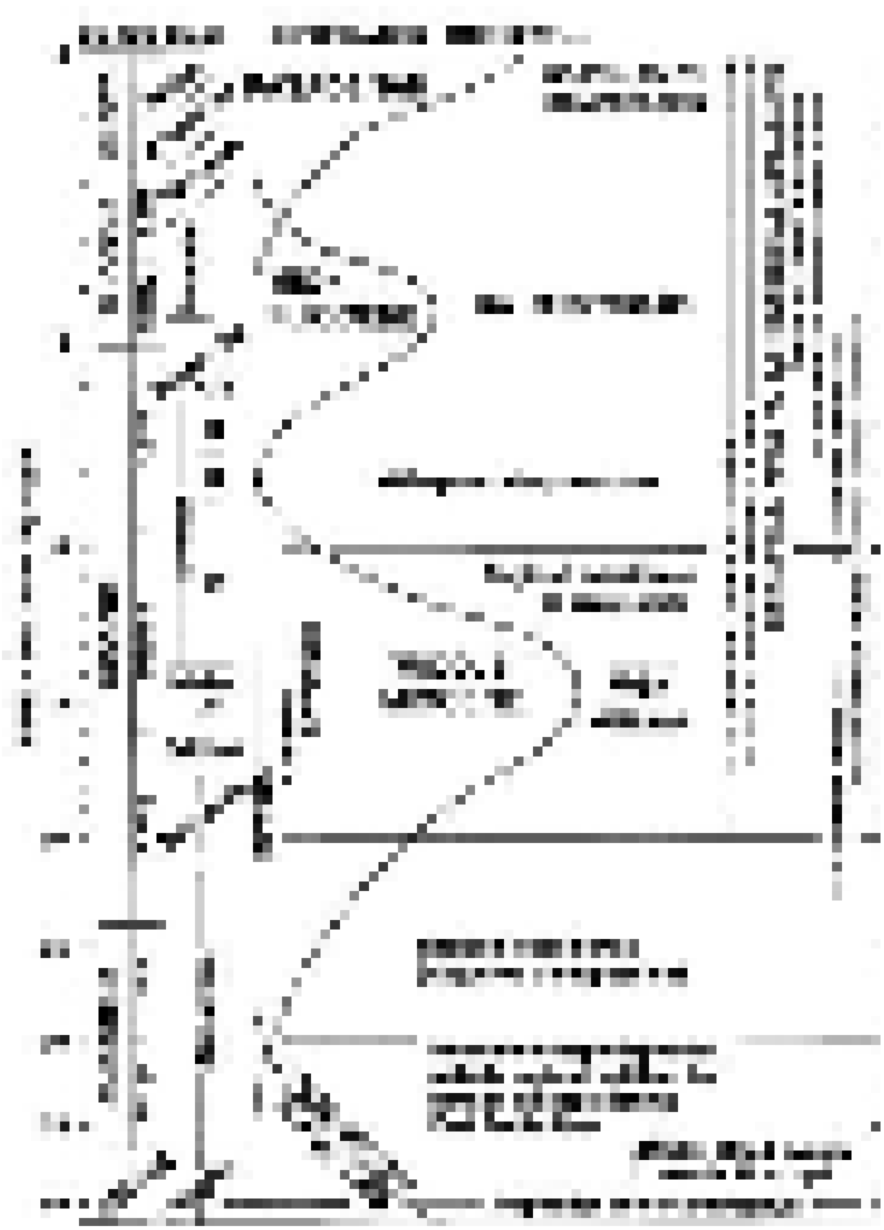
Cross Section of Structure and Geologic Formation

Table 2-2 Stratigraphy of the Inglewood Oil Field

Epoch	Formation		Reservoir	Thickness
Pleistocene	San Pedro			0' - 200'
Pleistocene	Inglewood			150' - 300'
Pleistocene	Pico	Upper		150' - 300'
Upper Pliocene	Pico	Middle	Investment	200' - 600'
Upper Pliocene	Pico	Middle, Lower	Vickers	1500' - 1700'
Lower Pliocene	Repetto	Upper	Vickers	1500' - 1700'
Lower Pliocene	Repetto	Upper	Rindge	900' - 1000'
Lower Pliocene	Repetto	Middle	Upper Rubel	250' - 300'
Lower Pliocene	Repetto	Middle	Lower Rubel	600' - 700'
Lower Pliocene	Repetto	Lower	Upper Moynier	300' - 400'
Lower Pliocene	Repetto	Lower	Lower Moynier	600' - 700'
Upper Miocene	Puente		Bradna	700' - 1800'
Middle Miocene	Puente		City of Inglewood	0' - 250'
Middle Miocene	Puente		Nodular Shale	150' - 175'
Middle Miocene	Puente		Sentous	200' - 1000'
Middle Miocene	Topanga		Topanga	1500'

Table 2-3 Geologic Time Scale

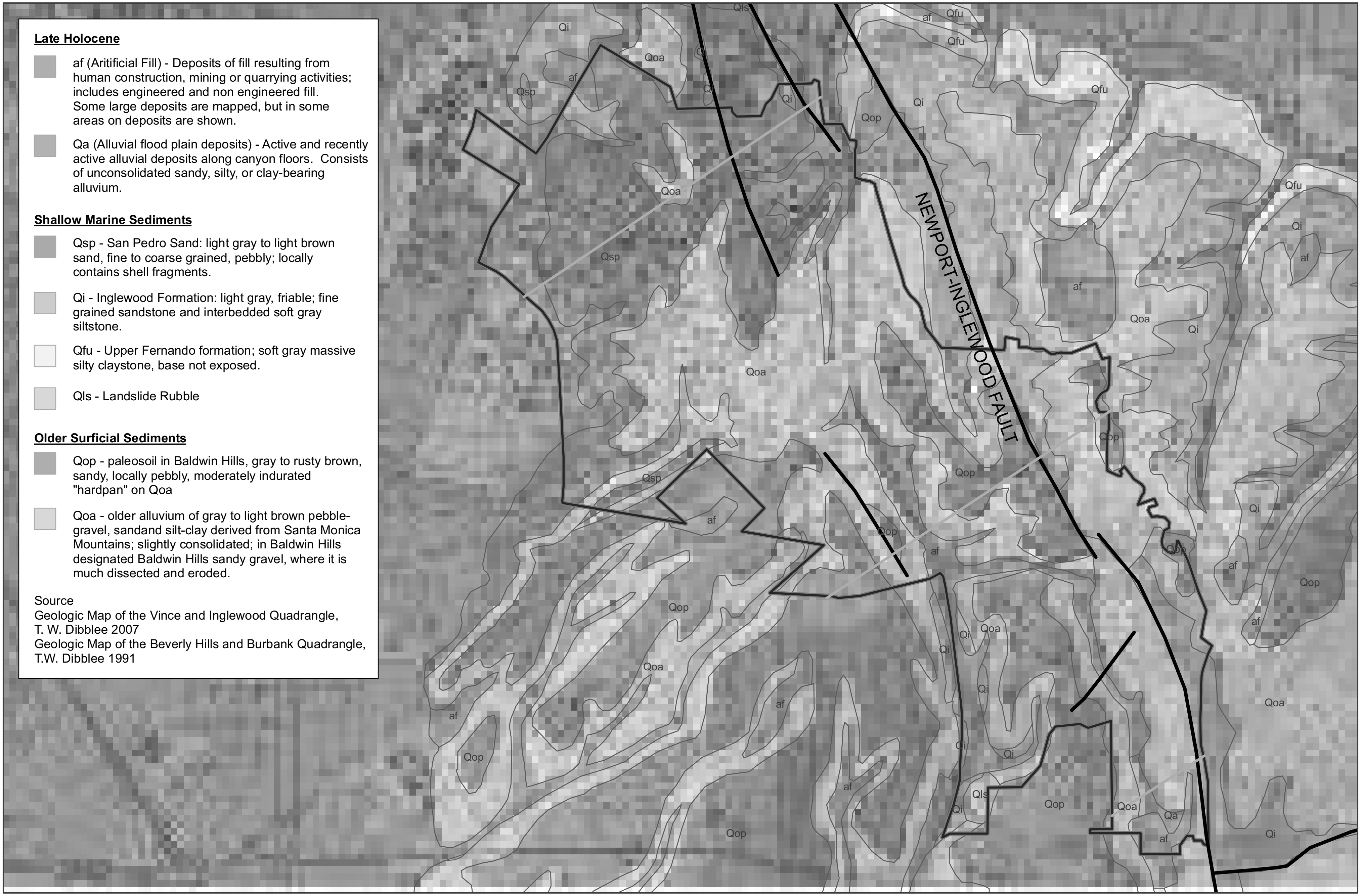
Era	Period	Epoch	Years Before Present
Cenozoic	Quaternary	Holocene	10 thousand
		Pleistocene	1.8 million
	Tertiary	Pliocene	5 million
		Miocene	23.5 million
		Oligocene	39 million
		Eocene	53.5 million
		Paleocene	65 million
Mesozoic	Cretaceous		144 million
	Jurassic		208 million
	Triassic		245 million
Paleozoic	Permian		286 million
	Pennsylvanian		320 million
	Mississippian		360 million
	Devonian		408 million
	Silurian		438 million
	Ordovician		505 million
	Cambrian		570 million
	Ediacarian		700 million



Source: Wright 1991

Figure 2-5 Chronology of Major Cenozoic Events in the Los Angeles Region

Figure 2-6 depicts the surface geology. Compared to the surrounding Los Angeles Basin, the geology of the Baldwin Hills exposes older and deeper geological formations. In addition, the Newport-Inglewood Fault and other related faults are shown as they are interpreted to occur near the surface. This structural discontinuity between the Baldwin Hills and the surrounding basin in part explains the occurrence of oil and gas at this location, and the discontinuity of shallow groundwater with deeper groundwater formations in the Los Angeles Basin (USGS 2003).



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- Fault Line
- Inglewood Oil Field Boundary
- Approximate location of cross sections displayed in Figure 2-4

PLAINS EXPLORATION & PRODUCTION COMPANY

Figure 2-6

Geologic Formations Present at the Inglewood Oil Field and Vicinity

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3-D Depiction of Inglewood Oil Field Geology

As part of the development of this study, Halliburton was retained to develop a three-dimensional (3-D) geological depiction of the subsurface of the Baldwin Hills, from the depth of the Sentous, the lowest known formation (10,000 feet below ground surface), up to the surface. The 3-D depiction is based on geological and structural data from drilling oil wells. The objective was to assist in the interpretation of the results of the high-volume hydraulic fracturing operation, and to better understand the relationship between the deep oil-producing formations and the shallow subsurface including the occurrence and distribution of shallow groundwater. The following discussion describes each formation, ending in the present-day land surface. The final 3-D model is used to depict the results of hydraulic fracturing, and of the discontinuous, fragmented water-bearing zones at shallow depths beneath the field, and the units that constitute the hydrocarbon seal that traps oil and gas in the deep subsurface (Figure 2-7).

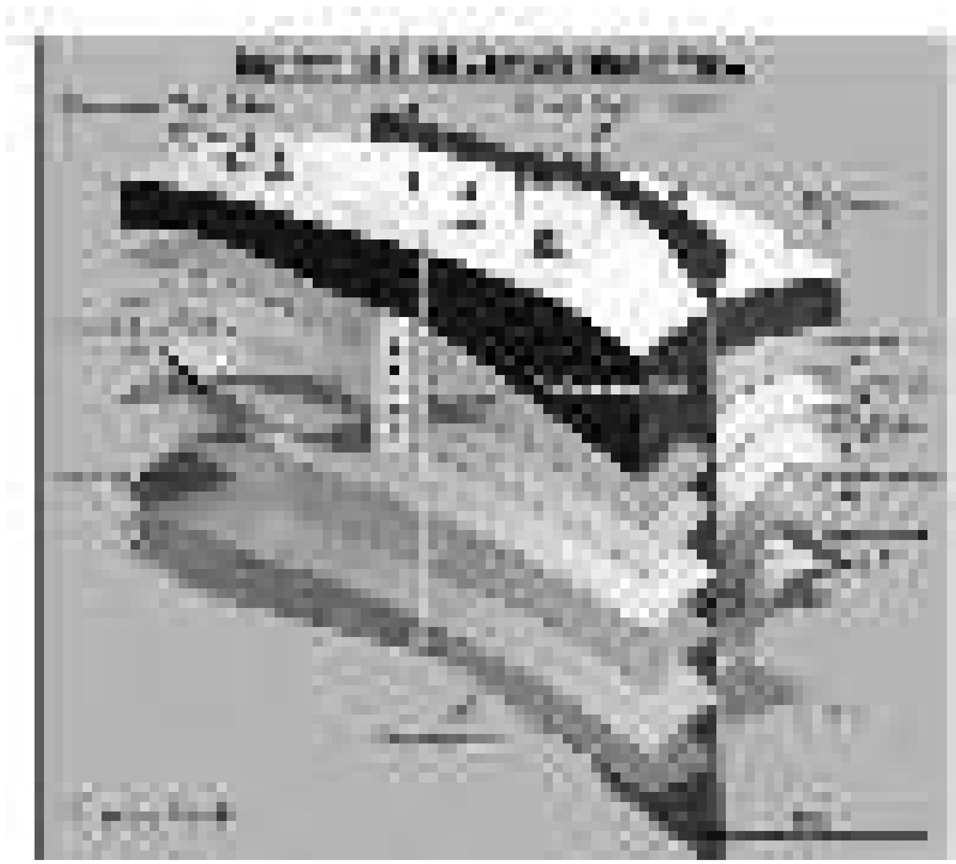


Figure 2-7 Cross Section of the Inglewood Oil Field Earth Model

In the following, each figure shows the progressive development of the field, starting with the deepest, oldest unit evaluated, the Sentous Sandstone. This presentation is used to show the growth of the formations and the folding and faulting specific to the area beneath the Inglewood Oil Field, as it is currently understood. Once constructed, this 3-D depiction is used to illustrate some of the study results later in the study.

Also shown for reference are the two wells that had high-volume hydraulic fracturing (VIC1-330 and VIC1-635), four wells that had conventional hydraulic fracturing in the past in the Sentous formation, and the two wells that had high-rate gravel packs (TVIC-221 and TVIC-3254). The yellow marker is the surface location of the well, and the red line is the length of the well. Each layer represents the top of one of the formations described above, and the space in between would be filled with that particular geologic formation (shale or sandstone). The fault planes are shown as colored layers cross-cutting the geologic formations. All the depictions are constrained by geological and structural data for the oil field.

Figure 2-8A Sentous Surface

Figure 2-8B Nodular Surface on Top

The top of the Sentous sandstone is shown in Figure 2-8A. The Sentous is also known as the Topanga Formation elsewhere in the Los Angeles Basin. This was a period of active volcanic activity; the basin was under an extensional regime and a strike-slip regime, forming a pull-apart basin that was actively subsiding. The volcanic intrusions into the sediments filled from the bottom, and at the same time erosion from distant land areas fed sandy sediments to form the Sentous sandstone. The microfauna indicate a depth of 3,000 to 4,000 feet below the ocean surface during this time. There are similar microfauna now in the Gulf of California, indicating that water temperatures were higher than today. All of the volcanic deposits are found below this layer.

The base of the Nodular Shale is also the top of the Sentous (Figure 2-8B). The Nodular Shale grades directly from the Sentous sandstone. This organic rich shale that is the source rock for much of the oil found here, and is approximately 150 feet thick; at the time of deposition, it may have been as much as 400 feet thick but has since been compressed. The mineralogy of the shale includes plagioclase derived from the volcanic rocks, and clay from the distant landmass depositing in the basin. The grain size became finer because of a decrease in the land-based sediment, leading to dominantly marine shale deposit. The Nodular Shale was deposited across the Los Angeles Basin. The subsidence of the Los Angeles Basin ceased approximately 2 million years ago.

Figure 2-8C Bradna Surface on Top

Figure 2-8D Bradna Surface with Faults

The Pasadenan Orogeny (mountain building) began about 2 million years ago. The activity led to uplift and rotation of plates, and a transition from a strike slip and extensional regime to a strike slip and compressive regime. Compaction and uplift forms the Baldwin Hills at this time, and is ongoing today. During this time period, the traps started to form; the folds and faults act as impermeable zones that allow oil to accumulate beneath them. It is believed that the Newport-Inglewood Fault may have originated as a normal fault giving it a steep angle, and accommodated the strike-slip motion. The two grey faults in Figure 2-8D are thrust faults accommodating the compression. This block was likely oriented NW-SE, but has rotated to E-W.

The Newport-Inglewood Fault is also shown in Figure 2-8D. The Newport-Inglewood Fault terminates in the northern portion of the field, as depicted in Figure 2-4, and Figure 2-8D expands on that termination. The fault likely transitions to folds or thrust faults as it approaches the Santa Monica Fault to the north.

The interval above the Nodular Shale at Inglewood includes the Bradna Shale. In other Los Angeles Basin fields, such as Long Beach and Beverly Hills, sands were deposited instead of the Bradna Shales. It is thought that the Inglewood area at the time formed a topographic high, such as a submarine knoll.

Figure 2-8E Moynier Surface on Top

Figure 2-8F Moynier Surface with Faults

The Moynier formation is shale, likely reflecting the submarine knoll that is more or less unique to the Inglewood Oil Field compared to other parts of the Los Angeles Basin. Some sand channels begin to appear in Moynier time, but they are minor.

Figure 2-8G Rubel Surface on Top

Figure 2-8H Rubel Surface with Faults

The Rubel marks the return of sand after the Sentous sandstone. It is the first major sand unit, and is the first major petroliferous zone at Inglewood. Approximately 90 percent of oil production is from the sandy submarine debris flow deposits (turbidities), first represented by the Rubel formation. These are deep-sea fans that funnel land-derived sands down to the deep ocean area. These formations are overlapping fans, and are currently active offshore of Southern California.

Figure 2-8I Rindge Surface on Top

Figure 2-8J Rindge Surface with Faults

The Rindge Formation is another productive sandstone for oil development. New structures are represented here in Figure 2-8J. These are interpreted as normal faults. The area was still dominantly strike slip with compression, but we interpret these as relatively shallow normal faults. These form the graben structure in the southeastern portion of the field. These could also be dominantly strike-slip faults with a normal component.

Figure 2-8K Vickers H-Sand Surface on Top

Figure 2-8L H-Sand Surface with Faults

The Vickers unit is another productive sandstone, similar to the description for the Rindge.

Figure 2-8M Vickers Surface on Top

Figure 2-8N Vickers Surface with Faults

Figure 2-8O UIHZ Surface on Top

Figure 2-8P UIHZ Surface with Faults

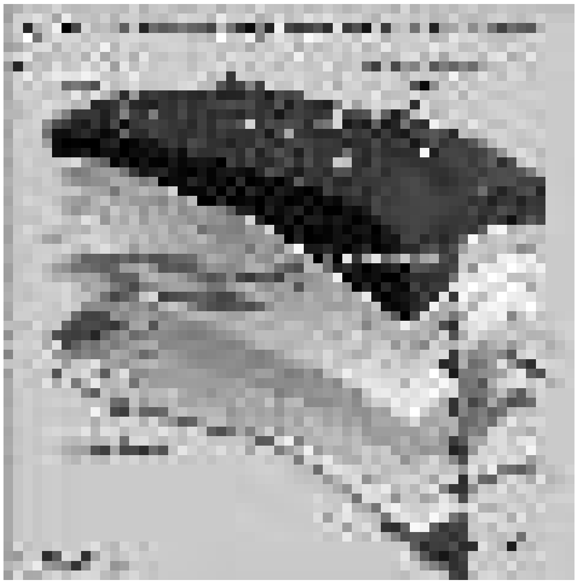


Figure 2-8Q Vickers Reservoir Hydrocarbon Seal with Faults

Figure 2-8R PICO Surface on Top

There is a prominent, relatively impermeable layer at the top of the Vickers, within the upper portion of the Pico Formation. The impermeable layer is more shale-rich than the underlying sandstones and forms a seal, inhibiting further upward migration of oil and gas. There are limited oil and gas deposits in the lowermost portion of the hydrocarbon seal; these are known as the Investment Zone. The folded and faulted units below act as traps beneath this seal. The depositional environment is still similar to that of the sandstones: submarine turbidite fans. However, this time may have been relatively less active, so the deposits are finer grained and formed a relatively impermeable shale instead of a sandstone.

Figure 2-8S PICO Surface with Faults

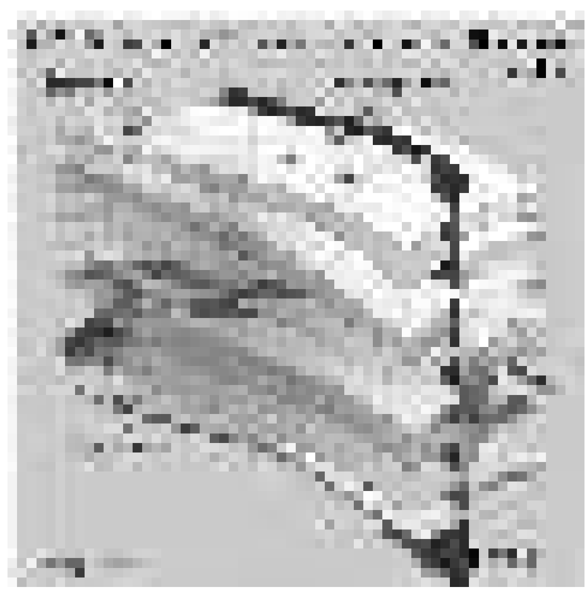


Figure 2-8T PICO Surface w/ Discontinuous Water Bodies & Faults

The top of the Pico Formation is also considered the Base of Fresh Water across much of the Los Angeles Basin. The Pico is a marine formation similar to the underlying units, and the formation water is salty. At shallower depths, above the Pico (Figure 2-8T), water is relatively fresh, but occurs in isolated, discontinuous water bearing zones that do not provide a sufficient yield for water supply, and are separated from the water-bearing zones elsewhere in the Los Angeles Basin. The aerial photograph of the Inglewood Oil Field is overlain on the geologic strata to provide reference (Figure 2-8U).

Figure 2-8U Ground Surface & Aerial Photo on Top

2.4.2 Petroleum Producing Zones

The field produces oil, natural gas, and saline water from interbedded sandstone and shale sediments ranging from Miocene Upper Topanga Formation (approximately 15 million years in age) to late Pliocene Upper Pico Formation (approximately 2 million years in age). See Table 2-3 for Geologic Time Scale. Production within the field is from nine zones that range in depth from about 900 to 10,000 feet. In order of increasing depth and increasing geologic age, the producing horizons are: Upper Investment-Investment, Vickers, Rindge, Rubel, Upper and Lower Moynier, Bradna, City of Inglewood, Nodular Shale, and the Sentous (refer to Figure 2-4, which illustrates the geology of the Baldwin Hills). The shallow reservoir zones (Vickers and Rindge zones) have been undergoing waterflood treatment since 1954. Each of the producing formations, along with the active wells completed in each zone, is summarized in Table 2-4.

Table 2-4 Summary of Active and Idle Wells within Each Oil and Gas-bearing Formation on the Inglewood Oil Field

Series	Formation	Average Depth Below Ground Surface	Number of Active Wells		Number of Idle Wells	
			Producer	Injector	Producer	Injector
Upper Pico	Upper Investment-Investment	1,000 feet	5		9	
	Vickers	2,000 feet	188	83	45	28
	Vickers-Rindge[1]	2,000 – 3,000 feet	167	67	27	33
Lower Repetto	Rindge	3,000 feet	10	3	5	1
	Rubel	4,000 feet	8	4	6	3
	Rubel-Moynier[2]	4,000 – 5,000 feet	16	5	5	
	Upper Moynier	5,000 feet	22	4	28	4
	Lower Moynier	5,500 feet				
Upper Puente	Bradna	6,000 feet	1			
	City of Inglewood	7,000 feet	1		2	1
	Nodular Shale	8,000 feet				
Upper Topanga	Sentous	8,500 feet	12			
Wells drilled within other transition areas between formations			39	2	14	1
		Total	**469**	**168**	**141**	**73**

Source: Fugro Consultants 2011, PXP 2012

[1]These wells are completed in both the Vickers and Rindge formations
[2]These wells are completed in both the Rubel and Moynier formations

A total of 1,475 oil wells have been drilled on the Inglewood Oil Field; these are active, idle, or plugged. Many have been directionally drilled and are non-vertical (i.e., drilled on a slant or angle). There were 469 active production wells and 168 active waterflood injection wells operating as of the writing of this study. Table 2-4 identifies the number of producing (pumping) well and injection wells in each zone, divided between active and idle wells. Plugged and abandoned wells are not included in Table 2-4.

Vickers and Rindge Formations

The Vickers and Rindge zones accounted for more than 74 percent of the total cumulative production at the Inglewood Oil Field in 2011 and 2012 (to date). Overall, the shallow and extensive Vickers and Rindge zones have produced more than half of all the oil produced over the life of the Inglewood Oil Field. In the context of this study, all of the high-rate gravel packs have been completed in these two zones.

The primary development focus in the Vickers and Rindge zones occurs between 2,000 and 4,000 feet below the ground surface; limited production from the Investment Zone occurs at approximately 1,000 feet. The formations are cut by faults, which act as barriers to fluid flow because they cut off permeable sand formations.

Nodular Shale Formation

The Nodular Shale is the name given to that portion of the Upper and Middle Miocene rocks of the Western part of the Los Angeles Basin that carry large phosphatic nodules. It is a subunit of the Monterey Formation. The Nodular Shale is known to underlie several oil fields of the Los Angeles Basin including Playa Del Rey (Hoots 1931, Wissler 1943), El Segundo (Porter 1938, Wissler 1943), Inglewood (Wissler 1943), Torrance (Wissler 1943), and Wilmington (Wissler 1943). It is also suspected to underlie the Beverly Oil Field (Hoots 1931) and the Lawndale Oil Field. In the context of this Hydraulic Fracturing Study, the only two high-volume hydraulic fracture completions that have occurred at the Inglewood Oil Field have been done within this formation.

The Nodular Shale is a highly organic, dark brown to black shale, and has produced small amounts of oil in several wells at Inglewood. This distinctive unit was deposited on deeply submerged offshore ridges and slopes through the slow accumulation of biological debris, diluted by clay particles carried in suspension by circulating ocean currents. The high organic content of the Nodular Shale indicates the presence of anaerobic conditions seen in the northern area of the Nodular deposition.

Sentous Formation

The Sentous Formation is the deepest unit produced at the Inglewood Oil Field, and is below the Nodular Formation at greater than 9,000 feet below the ground surface. The Sentous is the geologically oldest producing zone in the Inglewood Oil Field and also along the Newport-Inglewood Fault trend. Since the early 1990s, the exploration and development focus in the Inglewood Oil Field has been on the Lower Pliocene and Upper and Middle Miocene, particularly the Sentous. Sentous sands were deposited in approximately 1,000 feet water depth during the opening of the rifted basins of the Southern California continental borderland. Oil accumulated in the Sentous sands down the northwest plunge of the Inglewood anticline; however, the sands become impermeable higher up on the anticlinal crest due to filling of the pore spaces with calcite cement. This loss of permeability has created a stratigraphic trap for this reservoir (Halliburton 2012). In the context of the Hydraulic Fracturing Study, the conventional hydraulic fracture completions have been conducted either solely in the Sentous zone or combined in the Sentous and either the Moynier or the Bradna.

2.5 Future of Oil and Gas Development in the Los Angeles Basin

The Monterey Shale is the primary source of oil and natural gas found in Southern California. The organic-rich shale was heated and compressed during tectonic activity, producing oil and gas. Some of the oil and natural gas migrated upwards into the overlying, more permeable, sandstone layers, where the hydrocarbons were then trapped by overlying impermeable shales and faults. Across Southern California, the deep source rocks, approximately 2 miles below the ground surface, of the Monterey Formation are now an exploration objective. High-volume hydraulic fracturing is being explored as a possible well completion method to allow the extraction of oil and natural gas from this geologic formation.

At a 2012 meeting of the American Association of Petroleum Geologists, the U.S. Geological Survey presented an assessment of the amount of oil remaining in the Los Angeles Basin. They note that, during much of the twentieth century, discovery and development of the Los Angeles Basin oil fields went hand in hand with rapid urbanization, which impacted field development from the first day of drilling. In spite of one of the world's greatest concentrations of oil per unit area, the oil recovery efficiency in the major fields continues to decrease (Gautier et al. 2012). Many small fields have been covered by residential or commercial development while still in primary production. For example, along the Wilmington Anticline and Newport-Inglewood Fault Zone, at least six fields have estimated original oil volumes in excess of one billion barrels. These fields have been in production for about 90 years. However, future recovery in such major fields could reasonably be expected to almost equal the total amount of oil recovered so far. It is predicted that oil volumes well in excess of one billion barrels could be recovered going forward from existing fields in the Los Angeles Basin through widespread application of current best practice industry technology such as improved imaging, advanced directional drilling, and other techniques (Gautier et al. 2012).

Along with continued oil and gas development in the Los Angeles Basin, hydraulic fracturing has been occurring to explore the resource potential of the Monterey Shale throughout California and in the Los Angeles Basin. Hydraulic fracturing is likely to continue to be utilized during recovery of the remaining petroleum resources. Figure 2-9 displays the location of wells in the southern California where hydraulic fracturing was reported in either 2011 or 2012 (as reported on www.fracfocus.org).

Cardno ENTRIX

0 7.5 15 30 Miles

N

LEGEND

Well Completed by Hydraulic Fracturing

XXX Well Name

XX Multiple Wells Completed by Hydraulic Fracturing

PLAINS EXPLORATION & PRODUCTION COMPANY

Figure 2-9

2011 and 2012 Reported Hydraulic Fracturing Operations in Southern California

09 | 12 | 12

Chapter 3

Hydraulic Fracturing at Inglewood Oil Field: Past, Present, and Future

3.1 Oil and Gas Well Drilling, Including Hydraulic Fracturing Completions

Well drilling is the process of drilling a hole in the ground for the purposes of extracting a natural substance (e.g., water, oil, or natural gas). Drilling and completing a well consists of several sequential activities, which are listed below in order (note that these activities may be conducted multiple times during the drilling of a well, or be already completed and not needed for a particular well):

- Building the well pad and installing fluid handling equipment;
- Setting up the drilling rig and ancillary equipment and testing all equipment;
- Drilling the hole;
- Running formation evaluation logs and other instruments down the well;
- Running casing (steel pipe) to line the wellbore;
- Cementing the casing;
- Removing the drilling rig and ancillary equipment;
- Logging the casing to ensure bonding of cement to the formation and casing;
- Perforating the casing;
- Stimulating the well;
- Installing surface production equipment;
- Beginning production of the well;
- Monitoring well performance and integrity; and
- Reclaiming the parts of the drilling location that are no longer needed and removing equipment no longer used.

In the exploration and development of oil and natural gas fields, wells must be designed to carry the extracted fluids directly from the producing zone at depth to the surface completely within the well, without allowing fluid to escape into surrounding formations. Wells are designed and constructed to prevent any communication (migration and/or transport of fluids) between these subsurface layers, which have acted as a barrier for millions of years (API 2009).

In most parts of the Los Angeles Basin, including the Inglewood Oil Field, there are impermeable rock formations that lie between the hydrocarbon producing formations and shallow zones including groundwater-bearing formations and the land surface. These formations provide additional, natural protection against migration of oil and gas to the shallower formations. These impermeable formations and confining faults at the Inglewood Oil Field isolate, or trap, the hydrocarbons from the near surface formations. If these impermeable formations did not exist, the naturally buoyant oil would continue rising until reaching the surface, similar to areas such as the La Brea Tar Pits.

3.1.1 Drilling, Casing, and Cementing

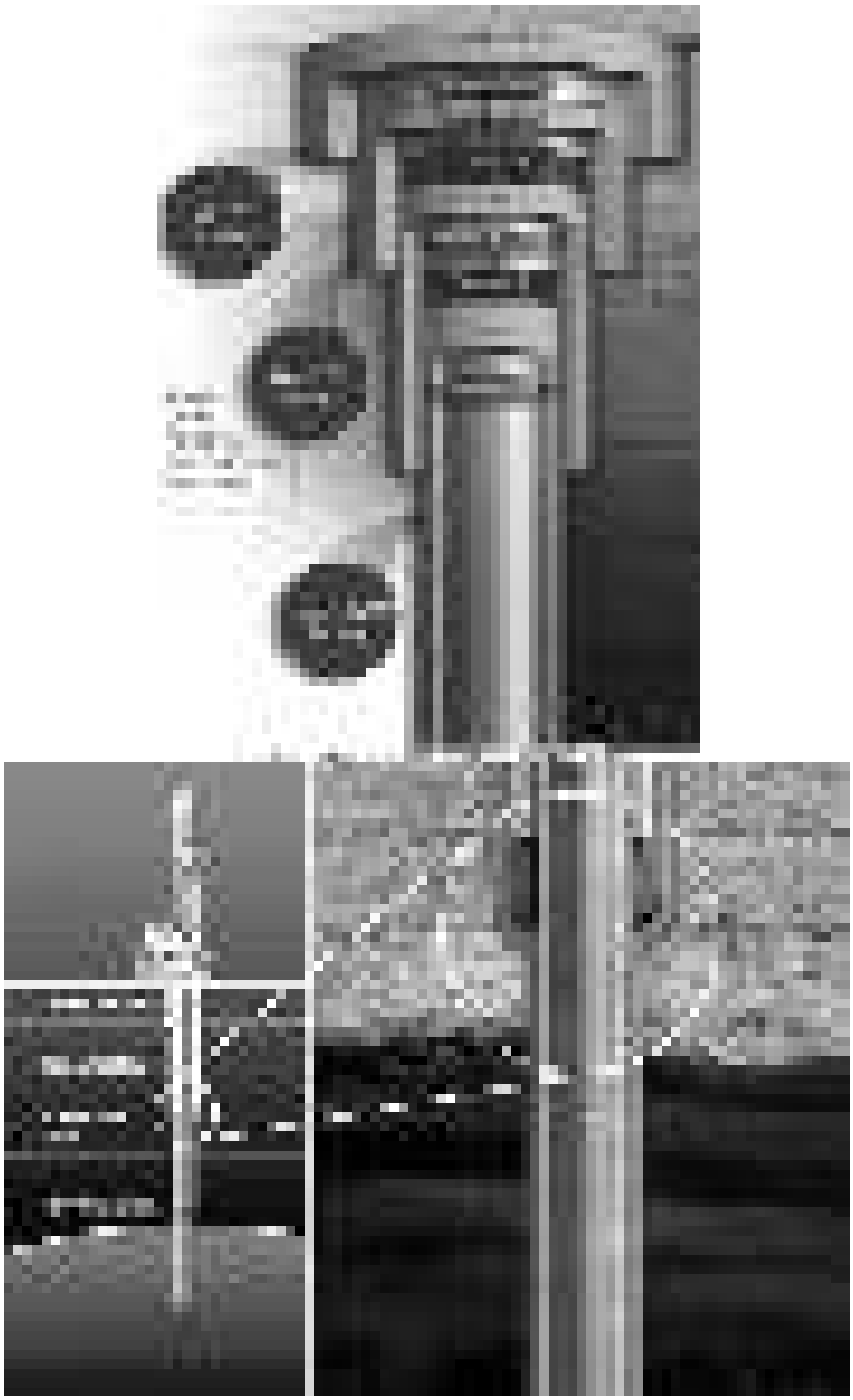
This section describes the methods used during drilling to ensure oil, natural gas, and water that are pumped from the deeper formations are brought to the surface without loss to shallower zones.

Wells are drilled using a drilling rig equipped with a drill string. The drill string consists of a drill bit, drill collars (heavy weight pipes that put weight on the bit so that it cuts through the formation), and a drill pipe. The drill string is assembled and suspended at the surface on a drilling derrick and run into the hole in the ground. It is then rotated using a turntable, or motor, in order to cause the drill bit to advance downward through the formations and thereby extend the hole deeper into the ground.

While the hole is drilled, fluid (drilling mud) is circulated down the drill string and up the space between the drill string and the hole. This drilling fluid serves to lubricate the drilling assembly, remove the sediments that are drilled, maintain pressure control of the well and stabilize the hole being drilled (prevent collapse of sediments back into the hole). Drilling fluid is generally a mixture of water, clays, and additives that prevent fluid loss, control density, and suspend the drilled cuttings. The first hole drilled is for installation of the surface protection casing. This is followed by sequentially deeper holes so that the well can be completed (API 2009).

The first step in completing a well is to case the hole (Figure 3-1). As the well is drilled and drilling fluid is removed, a series of steel pipes known as casings are inserted to prevent the boring from closing in on itself. Cemented casing also serves to isolate the well from the surrounding formation. Each length of casing along the well is often referred to as a casing string. The steel casing strings are a key part of well design and essential to isolating the formation zones and ensuring integrity of the well. Cemented casing strings protect against methane migration and protect groundwater resources (if present) by isolating these shallow resources from the oil, natural gas, and produced water (water produced during operation of a well) inside of the well. It is important to note that the shallow portions of the well have multiple strings of steel casing installed (Halliburton 2012, API 2009).

When drilling nears the base of fresh water, typically sealed naturally from deeper saline water by an impermeable confining layer, as at the Inglewood Oil Field, the casing is placed into the drilled hole. The design and selection of the casing is important since the casing has to be able to withstand various forces (for example compression by surrounding formation), as well as any pressure it might be subjected to during the well's life. The casing is threaded on each end that allows it to join to the next pipe. When several joints of casing are screwed together, they form a continuous string that isolates the hole.



Source: Halliburton 2012

Figure 3-1 Depiction of Casing Strings

The casing used in wells at the Inglewood Oil Field meet the American Petroleum Institute (API) standards. API standards entail strict requirements for compression, tension, collapse, burst resistance, quality, and consistency so that casing is able to withstand the anticipated pressure from well completion, fracturing and production, as well as environmental conditions that could cause corrosion (API 2009).

The space between the casing and the drilled hole (wellbore), called the annulus, is filled with cement, permanently holding the casing in place and further sealing off the interior of the well from the surrounding formation. Cementing is accomplished by pumping the cement (commonly known as slurry) down the inside of the casing into the well to displace the existing drilling fluids and to fill in the space between the casing and the actual sides of the drilled well. Once the cement has set, drilling continues to the next depth. This process is repeated, using smaller steel casing each time, until the targeted oil and gas-bearing reservoir is reached and cement is no longer used.

Oilfield cements are carefully designed products, formulated to meet the requirements of individual well designs. Cementing serves two purposes — it provides protection and structural support to the well while also providing zonal isolation between different formations, including full isolation of the groundwater. Cement is fundamental in maintaining integrity throughout the life of the well and protecting the casing from corrosion. Placement of the cement completely around the casing and at the proper height above the bottom of the drilled hole are two of the primary factors in achieving successful zone isolation and integrity. Proper isolation requires complete filling of the annulus and tight cement bonding to both the casing and the surrounding geologic formation. This bonding and the absence of voids prevents the development of migration pathways and isolates the production zone (Halliburton 2012, API 2009).

3.1.2 Hydraulic Fracturing as a Completion Technique

The final steps to a producing well are known as "well completion." Well completion includes perforations and any sort of well stimulation techniques, including hydraulic fracturing, sand control measures, installing the production tubing, and other downhole tools.

Perforating

Once the well is drilled to the target producing zone, cased and cemented in place, the areas outside the well are sealed off by the casing and cement. At this point in the process, there is a solid steel casing across the target producing zone. In order to pump out oil, natural gas, and water from this zone, a mesh of open space must be made in the casing. The process of creating the open holes within the target producing zone is called perforating; perforations are simply holes that are made through the casing. Perforating uses a series of small, specially designed shaped charges, which are lowered to the desired depth in the well and activated (Figure 3-2). These shaped charges create the holes in the steel casing that connect the inside of the production casing to the geological formation.

The perforations are isolated by the cement. Additionally, the producing zone itself is isolated outside the production casing by the cement above and below the zone. This isolation ensures that hydrocarbons and other fluids are unable to migrate anywhere except between the perforations and the wellbore.

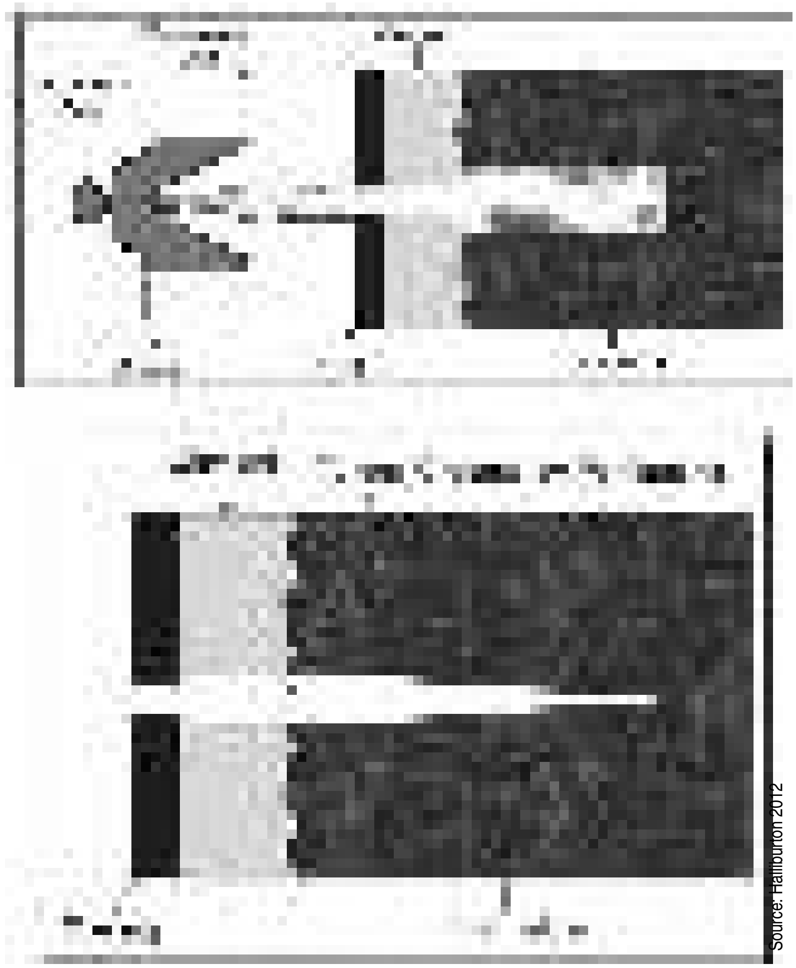


Figure 3-2 Perforation Process

Hydraulic Fracturing Process

Hydraulic fracturing is not part of the drilling process, but is a completion technique applied after the well is drilled, sealed, and perforated and the drilling rig has moved to another site. It is a well completion technology that results in the creation of fractures in rocks that allows oil and gas in the source rock to move more freely through the rock into the well. Hydraulic fracturing is a well stimulation process used to maximize the extraction of underground resources. Hydraulic fracturing is sometimes referred to as "fracking."

Hydraulic fracturing for stimulation of oil and natural gas wells was first tested in the United States in 1947. It was first used commercially in 1949, and was rapidly adopted because of increased well performance and increased yields of oil and gas from relatively impermeable rock units. It is now used worldwide in tens of thousands of oil and natural gas wells annually. The method has also been used at more shallow depths to assist in cleanup of contaminated industrial sites that have relatively impermeable zones.

In general, the process of hydraulic fracturing consists of injecting water, sand, and additives into the well over a short period of time (typically less than an hour) at pressures sufficient to fracture the rocks of a formation. Water and small granular solids such as sands and ceramic beads, called proppants, make up approximately 99 percent or more of the fluid used in a typical hydraulic fracturing operation (Halliburton 2012). This is consistent for both conventional and high-volume hydraulic fracturing. The flow of water acts as a delivery mechanism for the sand, which enters the newly-created fractures and props them open. If proppant does not enter a new fracture, then the pressure of the overlying rocks forces the fracture closed. These proppant-filled fractures allow oil and gas to be produced from reservoir formations that are otherwise too tight to allow flow.

The additives in the water help the sand to be carried farther into the fracture network. Such additives used to increase the viscosity of the water include gelling materials and/or foaming agents. Other liquid and solid additives that may be incorporated in the fracturing fluid are surfactants, a soap-like product designed to enhance water recovery, friction reducers, biocides to prevent microorganism growth, oxygen scavengers and other stabilizers to prevent corrosion of metal pipes, and acids to remove drilling mud damage. Figure 3-3 illustrates the composition of a typical fluid used in high-volume hydraulic fracturing. The specific products used at Inglewood Oil Field are described in Section 3.2.

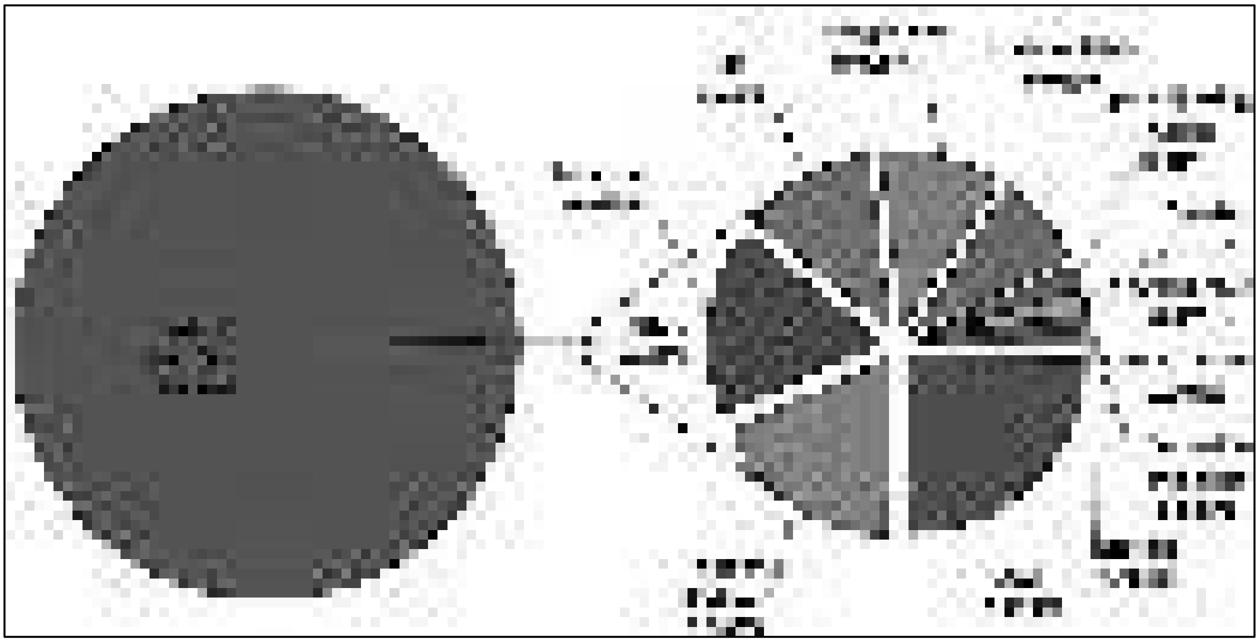


Figure 3-3 Composition of a Typical Fracturing Fluid

There are several steps during the hydraulic fracturing process. Taken together, these steps constitute one stage. Horizontal wells that are completed by hydraulic fracturing typically have several stages. Stages are not completed simultaneously. After the first stage is complete, the pressure is reduced, and the downhole equipment is moved to setup the second stage. When ready, the pressure is increased for the second stage. The following describes the steps that can be conducted during a hydraulic fracturing stage.

- **Step 1.** This optional step places water mixed with a dilute acid such as hydrochloric or muriatic acid into the sealed well. The volume of acid used is low and it is spent (used up) within inches of the fracture entry point and yields calcium chloride, water and small amount of CO_2. No acid is returned to the surface (King 2012). This step serves to clear cement debris in the wellbore and provide an open conduit for other hydraulic fracturing fluids by dissolving carbonate minerals and opening fractures near the wellbore. This step is not always performed, depending on the characteristics of the well and the formation.

- **Step 2.** The hydraulic fracturing fluid pad step (water with friction reducing additives) helps initiate and then propagate the fracture and assist in the placement of proppant material.
- **Step 3.** A proppant concentration step consists of several steps of adding water combined with proppant material (sand) to the well. This step may collectively use several hundred thousand gallons or more of water. Proppant material may vary from a finer particle size to a coarser particle size throughout this sequence and the proppant concentrations will vary during the treatment – starting with a lower concentration and then ramping to a higher concentration.
- **Step 4.** A flush step consists of a volume of fresh water or brine sufficient to flush the excess proppant from the wellbore.
- **Step 5.** Most of the fluid used for hydraulic fracturing is heated in the deep formation, becomes less viscous, flows more readily, and is recovered as it comes back up the well to the surface; this fluid is known as flowback. The amount recovered depends on the characteristics of the formation, and of the fluid used for hydraulic fracturing. The fluid that does not flow out of the well as flowback remains in the formation until the well is brought on production to pump and recover oil and gas. Any remaining fracturing fluids are also pumped out of the ground. Therefore, any remaining hydraulic fracturing fluid that does not return as flowback is captured by the pumping of the well. The only period of elevated pressure is during the brief (typically less than an hour) hydraulic fracturing operation itself (Halliburton 2012).

Uses of Hydraulic Fracturing

Within the last decade, the combination of horizontal wells installed with GPS-mounted drill heads to precisely guide the drill bit through relatively thin reservoir formations, and high-volume hydraulic fracturing completions has allowed the production of natural gas and oil from deep shale and tight sands deposits. Previously, the oil and gas-bearing shales were thought of as the source rocks of petroleum, from which oil and gas could not be economically produced directly. With the advent of new technology, companies now have the ability to precisely drill a horizontal well to be entirely within a relatively thin shale and tight sand bed using GPS technology, and then to precisely fracture that shale and prop open the fractures with sand to produce hydrocarbons from formations that previously were not economical.

This ability to capture hydrocarbon resources from zones that previously could not be produced is one form of development of “unconventional sources of oil and gas”. As applied to shale gas, this technique has completely changed the estimate of economic natural gas reserves. U.S. natural gas reserves had previously been thought to be in decline. To supply the nation’s energy needs, numerous plans to import gas from overseas as liquefied natural gas (LNG) were proposed between 2000 and 2005. Now, however, development of shale gas has led to the U.S. becoming the world’s largest producer of natural gas, surpassing Russia in 2009. The abundance of this relatively clean-burning fuel is beginning to displace the use of coal in U.S. generating stations, thus reducing greenhouse gas emissions.

At the Inglewood Oil Field, the uses of hydraulic fracturing are to create a permeable channel (propped fracture) within the shale and sandstone units so that the oil can be produced economically from these deeper formations. All activity occurs on an active, closely monitored oil and gas field using existing cleared areas for new wells whenever feasible. The activity differs from other parts of the country where areas not located in active oil and gas developments are converted to this use, and where the principal target is natural gas.

Types of Hydraulic Fracturing and Gravel Packing

Hydraulic Fracturing

Hydraulic fracturing as applied in oil and natural gas completions can take one of two forms, although some hybrid approaches are also in use. The process of fracturing in both forms is the same; the difference generally lies in the type of reservoir in which the fracturing is occurring, either tight sandstone or shale. The two forms are as follows:

- **Conventional Hydraulic Fracturing.** This completion approach uses water, sand, and additives to fracture and stimulate the producing formation to a distance of up to several hundred feet from the well. This method is intended to enhance the permeability of the target producing zone itself, and stimulate the reservoir. It is typically applied in tight sandstone formations and some shales.
- **High-Volume Hydraulic Fracturing.** This higher energy completion approach is generally applied to shales rather than sandstones. Sand and additives are used in the process similar to how they are used in conventional hydraulic fracturing; however, the primary distinguishing factor is the amount of fluid and pressure used in the process. Since shales have extremely low permeability, it is essential to increase the formation surface area contact with a permeable fracture channel. The high-volume hydraulic hydraulic fracturing process accomplishes this by increased treatment rates and material volumes.

Gravel Packing

In addition to hydraulic fracturing, the Settlement Agreement requires that gravel packing also be described and evaluated in this study. Gravel packing differs from hydraulic fracturing in that it is not intended to create fractures in the producing formation in order to pump out more water, oil, and gas. Rather, it is intended to place sand and gravel outside and adjacent to the well itself, with the intention of limiting the amount of fine-grained material that is pumped from the formation along with the fluids. As such, the purpose and techniques of gravel packing are distinctly different from hydraulic fracturing. Although the objective and techniques of gravel packing are very different from hydraulic fracturing, they are described in this study in accordance with the Settlement Agreement:

- **High-Rate Gravel Pack.** Since 2003, high-rate gravel packing has been conducted above the fracture pressure to improve well production performance through sand control. This operation uses much lower pressures than conventional and high-volume hydraulic fracturing. This completion approach, which is sometimes referred to as a “frack pack,” uses water, gravel, and additives to place sand and gravel near the well itself with the objective of limiting entry of formation sands and fine-grained material into the wellbore. In this process, the space between the formation and the outer casing of the well is packed, at a high-rate, with gravel that is small enough to prevent formation grains (sand) and fine particles from mixing and entering the wellbore with the produced fluids, but large enough to be held in place by the well perforations. This relatively low-energy completion approach can create limited fractures, using water, sand, and additives that improve the proper placement of the gravel filter. This process is not intended to increase the permeability of the producing formation, and it only affects the area near the well itself. Sand and finer particles that are entrained from the formation by pumping reduce the life of surface equipment such as valves, pipelines, and separators. In addition, produced sand can reduce oil production and impair the performance of injection wells.

- **Gravel Pack.** Prior to 2003, gravel packing was done at lower rates and lower applied pressures. The objective was the same as high-rate gravel packing, and the methods were also very similar, but the gravel packing process was always conducted at pressures less than the fracture pressure. In the past, some gravel packing was conducted using produced crude oil as part of the fluid mixture (a total of 11 completions); this oil was injected into the oil-producing formations themselves and not into shallow formations. Although PXP does not use oil as a fluid in gravel packing any more, it is noteworthy that such activity would not require an Underground Injection Control (UIC) permit since the operation did not use diesel fuel.

3.2 Hydraulic Fracturing at the Inglewood Oil Field

Conventional hydraulic fracturing has been conducted on 21 wells in the past at the Inglewood Oil Field. These completions were conducted in the Sentous Moynier, , Bradna ,City of Inglewood, Rubel, and Nodular shale formations. Combined, a total of approximately 65 stages of conventional hydraulic fracturing have occurred at the Inglewood Oil Field since 2003 when PXP began operating the field.

In conjunction with this Hydraulic Fracturing Study, PXP conducted high-volume hydraulic fracturing tests at two wells at the Inglewood Oil Field (VIC1-330 and VIC1-635). These are the only two high-volume hydraulic fracture jobs that have been performed on the Inglewood Oil Field.

Figure 3-4 shows the location of Inglewood Oil Field wells that have either been completed by high-volume hydraulic fracturing or conventional hydraulic fracturing since PXP took over field operations. All of the hydraulic fracturing has been completed on producing wells, that is, on pumping wells rather than injection wells.

3.2.1 Conventional Hydraulic Fracturing

Halliburton (2012) analyzed data from the past conventional hydraulic fracturing in the Sentous formation at the Inglewood Oil Field. The results of this analysis are summarized in this chapter to provide an indication of the feasibility and effectiveness of this technique at the Inglewood Oil Field.

Conventional hydraulic fracturing uses water, sand, and additives to fracture and stimulate the producing formation to a distance of up to several hundred feet from the well. This method is intended to affect the formation surrounding the perforated zone of the well, and enhance the hydrocarbon production of the target zone. It is typically applied in sandstone and some shale formations.

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0 500 1,000 2,000 Feet

LEGEND

Conventional Hydraulic Fracture

High Volume Hydraulic Fracture

Inglewood Oil Field Boundary

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Figure 3-4

Locations of Hydraulic Fracturing Operations at Inglewood Oil Field

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In this type of treatment, water is mixed with a polymer to increase the viscosity to the range of 10 to 40 centipoise (cp); for comparison, water viscosity is 1 cp. When ready to pump into the well, the water and polymer blend referred to as the "base gel" is blended further with a liquid additive that binds the polymer chains in the base gel increasing the viscosity to several thousand cp which aids in the suspension of the solids. This process is referred to as "cross-linking" the base gel. The cross-linked gel is mixed with the proppant and pumped into the well as slurry. The proppant, either natural (sand) or manmade (ceramic beads), is pumped along with the fluid and remains in the created fractures to hold it open. Additives designed to delay the degradation of the cross-linked gel are pumped along with the cross-linked gel and, in combination with the elevated temperature in the formation, return the cross-linked gel to a viscosity approaching that of water so that it can be recovered, or "flowed back" from the formation.

Conventional hydraulic fracturing has been used for every producing formation deeper than the Vickers and the Rindge at the Inglewood Oil Field. Most conventional hydraulic fracturing jobs were completed in the Sentous, the deepest producing formation at approximately 10,000 feet beneath the ground surface. Halliburton (2012) contains an analysis of the outcomes of hydraulic fracturing in the Sentous zone based on detailed analysis of two wells: TVIC-1033 and VIC2-1133. Figures 3-5A and 3-5B present different visualizations of the fracture geometries determined from the hydraulic fracturing treatments. The small rectangular area at the base of the diagram represents the calculated volume that received proppant. The figures include the relevant formation surfaces, ground surface, geologic structure, including major faults; water-bearing bodies near the surface are also depicted. The area affected by the conventional hydraulic fracturing remained in the Sentous formation, greater than 9,000 feet below the ground surface.

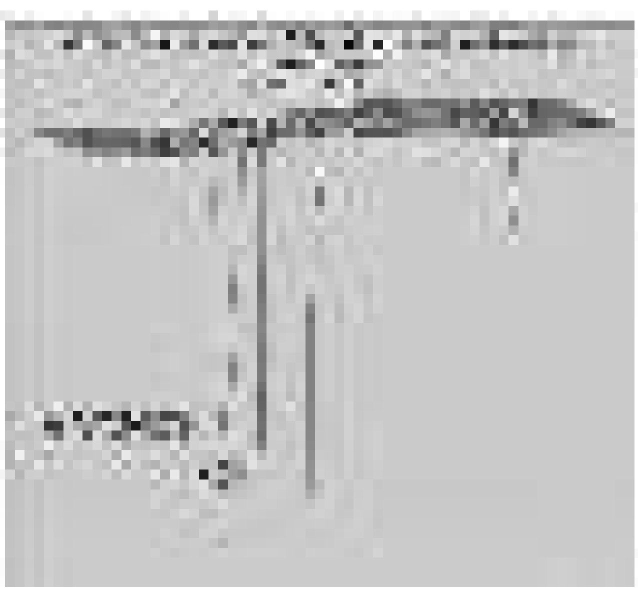

Figure 3-5A Side View of the Sentous Zone Modeled Fracture Geometries

Figure 3-5B Side View Showing Modeled Fracture Geometries for Study Well in the Sentous Zone Together with Structural Features (Faults)

3.2.2 High-Volume Hydraulic Fracturing

PXP contracted Halliburton Energy Services to conduct two high-volume hydraulic fracture jobs at separate wells on the Inglewood Oil Field for the purposes of addressing feasibility and potential impacts of hydraulic fracturing. The first hydraulic fracture completion was conducted on September 15 and 16, 2011, at the VIC1-330 well. The second completion was conducted on January 5 and 6, 2012, at the VIC1-635 well.

Only one hydraulically fractured stage was completed on each well during the operations. Both of these operations were conducted in the Nodular Shale, a subunit of the Monterey Shale, approximately 8,000 to 9,000 feet below ground surface. Halliburton (2012) contains a full report of these operations.

The conditions of the hydraulic fracture jobs are the same as those expected for any other future high-volume hydraulic fracturing to be conducted at the field. Therefore, the applied pressure, water use, and monitored effects are expected to be similar between these two stages of high-volume hydraulic fracture jobs and any future stages of high-volume hydraulic fracture jobs.

Future high-volume hydraulic fracturing completions would likely utilize more than one stage per well in the future. That is, any single hydraulic fracture job in the future could consist of more than one individual fracturing event. In hydraulic fracture jobs that consist of more than one stage, each stage would be conducted one after the other, never simultaneously. Therefore, any one stage will be similar to those stages described in this section. Cumulatively, the amount of water and chemicals used would be greater for a multi-stage completion than for a single-stage completion. However, the volumes required are still much less than the overall water usage at the field.

Although VIC1-330 and VIC1-635 are both vertical wells, PXP reports that in the future, high-volume hydraulic fracturing may be conducted via horizontal wells. This difference would not lead to any variation in the hydraulic fracture stage, or to the monitored effects; the only difference would be the construction of the well itself. Although a horizontal well can be much longer than a vertical well in the same formation, the hydraulic fracture completion targets an individual zone, and so the amount of water, sand, and additives used would be the same, stage for stage. A longer, horizontal well would also result in more than one stage, which as described above, would result in the use of greater volumes of water and chemicals.

This section describes the conditions and results of high-volume hydraulic fracturing of the VIC1-330 and VIC1-635 wells. Both hydraulic fracturing events were in the Nodular shale, at depths in excess of 8,000 feet below ground surface. Microseismic data are fist used to describe the hydraulic fracturing. Next, water demand, water reuse, and chemical use are described for both jobs.

Microseismic Monitoring Methods

A hydraulic fracture job generates microseismic events when the rock develops cracks. During the hydraulic fracturing treatment, these microseismic events are measured with seismic receivers or geophones placed at depth within a nearby well or wells. The events are so imperceptible, even by this sensitive equipment, that it must be placed at or near the depth of fracturing to detect them. Figure 3-6 shows the locations of the four wells, and the nearby wells used for microseismic monitoring of VIC1-330 and VIC1-635.

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0 500 1,000 2,000 Feet

LEGEND

High Rate Gravel Pack

High Volume Hydraulic Fracture

Monitoring Location

Inglewood Oil Field Boundary

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Figure 3-6

High-Volume Hydraulic Fracturing Operations with Microseismic Monitoring Locations

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Earthquakes and other seismic events are commonly measured using the Richter scale (Figure 3-7). The Richter scale is based on Magnitude; that is, an earthquake of Magnitude 6 is ten times stronger than an earthquake of Magnitude 5, as a result of the amplification of ground movements (e.g., soft soils overlying bedrock will strengthen the intensity of the ground movement). Events of Magnitude 3 to 4 are similar to vibrations caused by heavy traffic. Events of Magnitude 2 to 3 are typically not noticed by people. Events of Magnitude 1 to 2 are only detectable by seismographs and are not felt by people. For context, the Northridge earthquake of 1994 was Magnitude 6.4, and the San Fernando earthquake of 1970 was Magnitude 6.9.

During hydraulic fracturing, the microseismic events are generally less than Magnitude -2 or -3 on the Richter scale (Halliburton 2012). That is, they are about 1,000,000 times weaker than events that are typically felt by people. Although the pressures used in hydraulic fracturing are, by definition, high enough to fracture rock, the effects are very localized and do not induce further seismic effects. As discussed further in Section 4.5.6, recent studies by the U.S. Geological Survey and other organizations have consistently concluded that the forces generated by hydraulic fracturing do not cause earthquakes. These studies have shown that, under some conditions, injection of water or other fluids associated with wastewater disposal can, however, induce small tremors less than Richter Magnitude 3 or 4.

Source: South Dakota Geologic Survey 2012

Figure 3-7 Graphical Representation of Seismic Events as Recorded on the Richter Scale

Microseismic monitoring was conducted during hydraulic fracturing treatments for both VIC1-330 and VIC1-635. The results are used to determine the extent of fractured rock resulting from the treatment by mapping the locations of induced microseismic events. Figure 3-8A presents a detailed earth model side view visualization showing the locations of microseismic events detected during the mainstage fracture treatment. Each dot shown is a microseismic event, corresponding to a fracture. The rectangular area within the microseismic events represents the calculated volume that received proppant. The color of the microseismic events represents the time that they occurred. Taken together, the area affected by the microfractures is the zone affected by high-volume hydraulic fracturing. As depicted in Figure 3-8A, a few microseismic events occurred in the underlying Sentous Formation. However, the rectangular areas indicate that the proppant remained in the Nodular Shale, so the deeper fractures would seal after the high-volume hydraulic fracturing. Descriptions of fracture height and fracture length refer to the overall zone affected by fracturing; these are not the heights and lengths of individual fractures.

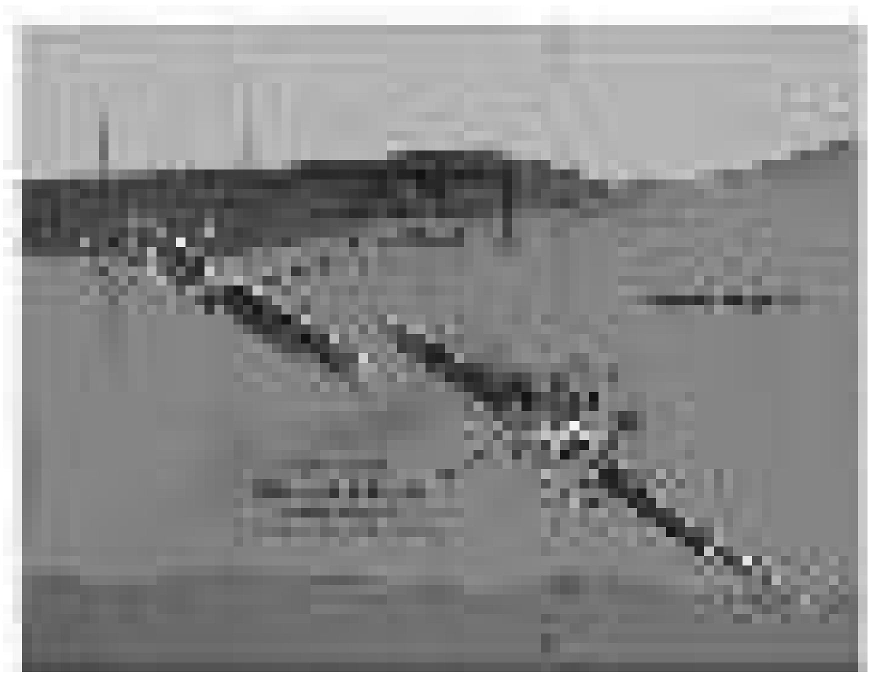


Figure 3-8A **Microseismic Events Detected During the Hydraulic Treatments in the Sentous Zone in Wells VIC1-330 and VIC1-635**

Figure 3-8B presents the 3-D model visualization of the microseismic events recorded during hydraulic fracture treatments in the Nodular Shale zone in wells VIC1-330 and VIC1-635. The distance between the top of the created fracture and the near-surface water bodies is approximately 7,700 feet. As shown in Figure 3-8B and described in the following sections, the fracture treatment stayed predominantly within the zone, and all proppant applied stayed within the zone (Halliburton 2012).

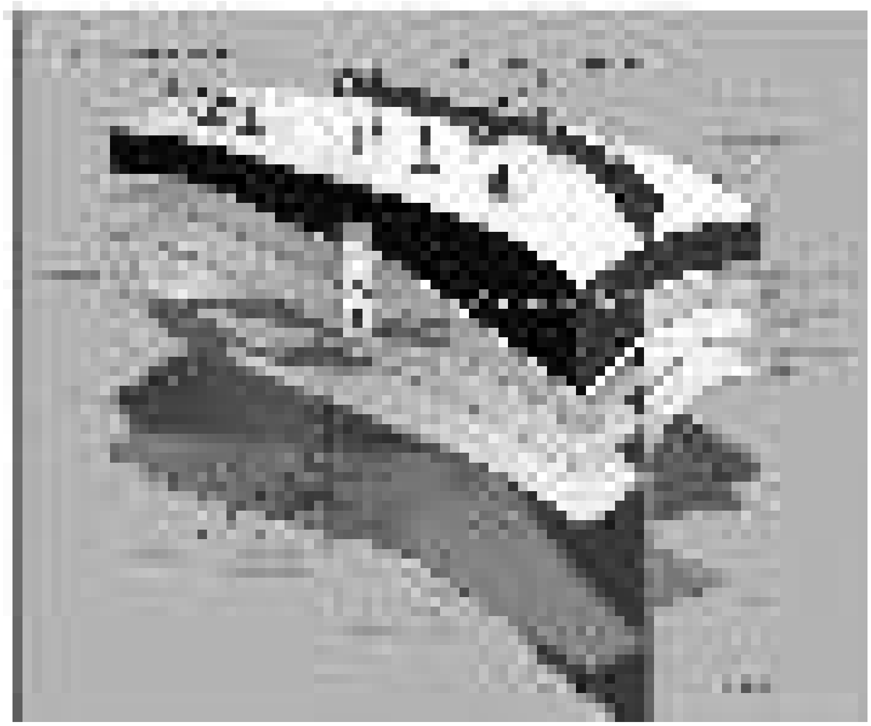


Figure 3-8B **Earth Model Visualization Showing the Microseismic Events Recorded during Hydraulic Fracture Treatment in the Nodular Shale Zone in Wells VIC1-330 and VIC1-635**

Well VIC1-330 Hydraulic Fracturing and Microseismic Monitoring

VIC1-330 well was hydraulically fractured between the depths of 8,030 to 8,050 feet below ground surface. The target formation was the Nodular Shale. The microseismic monitoring of VIC1-330 well was done from the VIC1-934 well using an array of geophones spaced 100 feet apart. The distance from the center of the geophone array to the perforations in the VIC1-330 treatment well is approximately 700 feet.

A total of 47 microseismic events were located during the hydraulic fracturing (Figure 3-9) operation. Based on the microseismic monitoring, the fractures are not radially distributed around the well, but follow three primary directions corresponding to the structure of the reservoir. Some fractures occur outside of the Nodular Shale, although most lie within the target unit (Schlumberger 2012b). Halliburton (2012) also models the distribution of proppant applied to the fractures in the target zone. Based on this model, all of the proppant stayed within the target zone of the Nodular Shale. The minor fractures that occurred outside the Nodular Shale did not receive proppant, and as such the minor fractures sealed based on the overburden pressure.

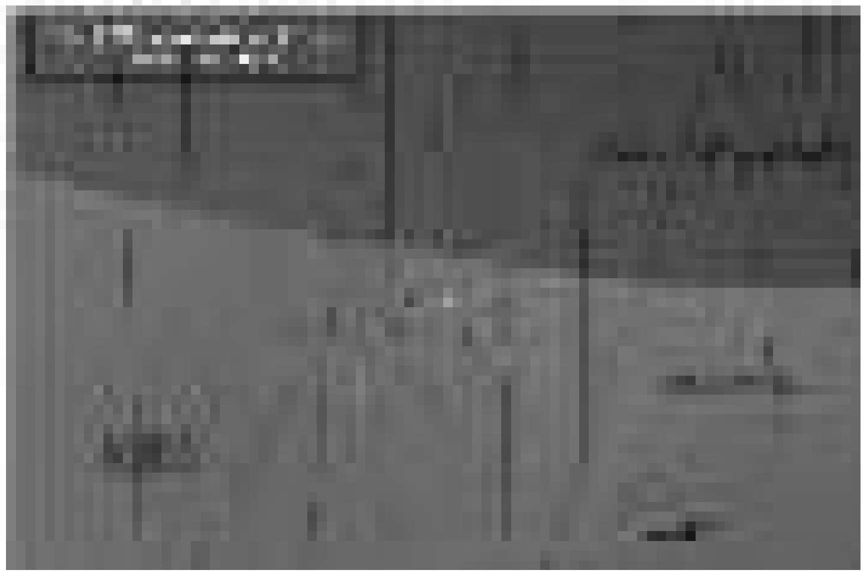


Figure 3-9 **Detailed Zoomed in Side View Visualization of the Microseismic Events Recorded during Fracture Treatment in the Sentous Zone in Well VIC1-330**

Well VIC1-635 Hydraulic Fracturing and Microseismic Monitoring

VIC1-635 well was hydraulically fractured between the depths of 8,430 to 8,450 feet below ground surface. The target formation was the Nodular Shale. The microseismic monitoring of VIC1-635 well was done from wells VIC1-735 and VIC1-935 using an array of geophones spaced 100 feet apart. Figure 3-10 depicts the microseismic events that were observed during the hydraulic fracturing treatment.

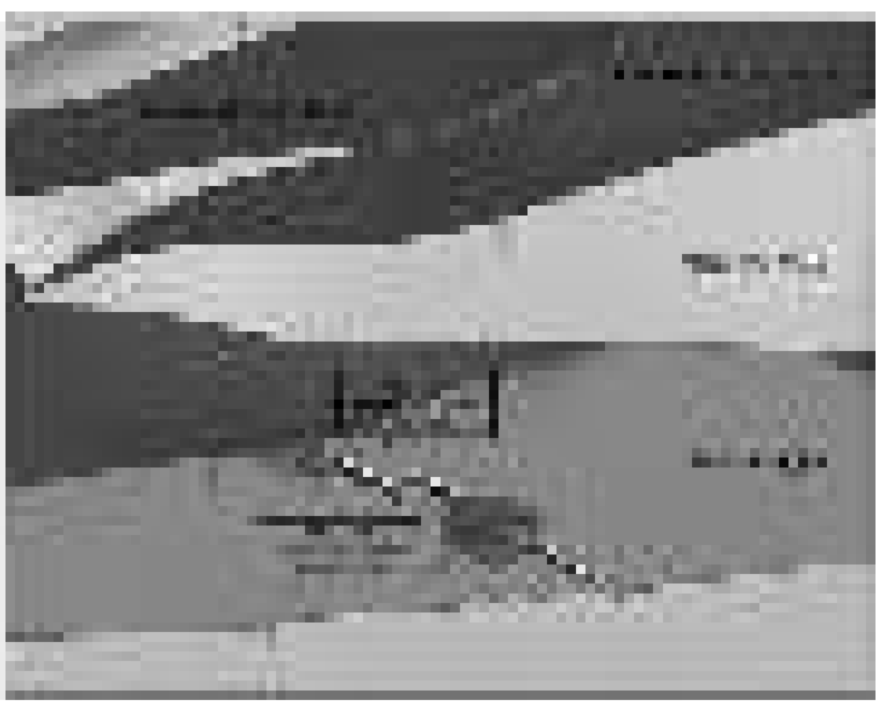


Figure 3-10 **Microseismic Events Detected during Mainstage Fracture Treatment, Top View**

Figure 3-11 shows the map view (left) and 2D Depth view (right) of the Mainstage high-volume hydraulic fracture treatment along with the microseismic events for the hydraulic fracture in well VIC1-635.

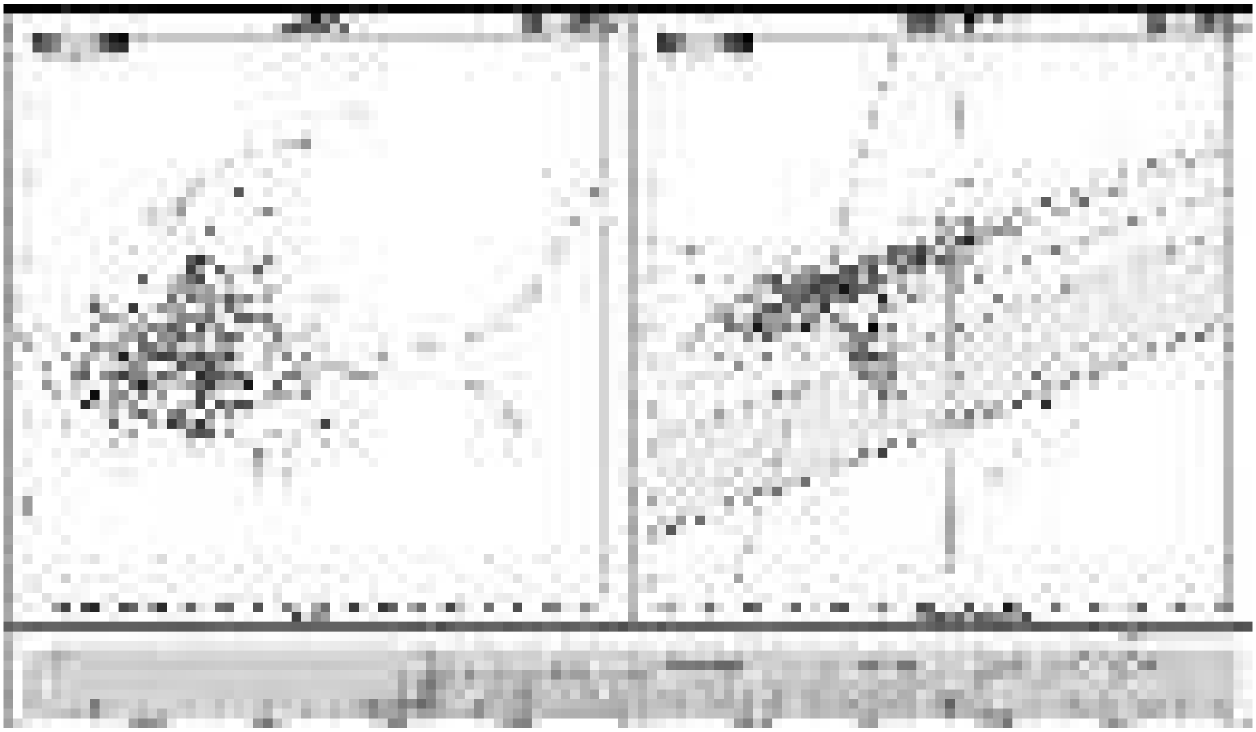


Figure 3-11 2D VIC1-635, VIC1-735 and VIC1-935 Surface Locations with Events Mapped

Earlier geologic control from well logs and structural mapping in the area indicated the Nodular Shale has dipping beds (~20°) from the northeast to the southwest. Based on the microseismic monitoring, the primary fracture network direction is considered east-west for the single stage mapped in the VIC1-635 well with a secondary fracture direction of N45°E. The microseismic mapping results indicate that the target zone of the Nodular Shale was effectively stimulated and fracture growth occurred along the formation dip of approximately 20 degrees. Growth appears to be asymmetric to the west based on the geophone array locations. The fracture network half-length was measured to be 750 feet. Fracture height was approximately 230 feet.

Almost all of the microseismic events occurred in the Nodular Shale; however, some microseismic events occurred outside the Nodular Shale and affected the Sentous Shale, underlying the Nodular. This appears to be related to pre-existing structure. Halliburton (2012) also models the distribution of proppant applied to the fractures in the target zone. Based on this model, all of the proppant stayed within the target zone of the Nodular Shale. Therefore the minor events (corresponding to microfractures) that occurred outside the Nodular Shale did not receive proppant, and as such they sealed based on the overburden pressure.

Water and Chemical Use during High-Volume Hydraulic Fracturing

Water Use and Source

Water for the hydraulic fracturing operations at the Inglewood Oil Field is provided from either produced water the field or, if a potassium chloride gel is used, fresh water provided by California American Water Company, the provider of all fresh water used at the Inglewood Oil Field. For both of the high-volume operations on the field, PXP used fresh water. Table 3-1 provides the volumes of water used during the high-volume hydraulic fracturing at the Inglewood Oil Field.

Table 3-1 Volumes of Water Used During High-Volume Hydraulic Fracturing Operations at the Inglewood Oil Field

Operation Type	Date	Well	Volume Water Used (gallons)	Water Source
High Volume	September 15-16, 2011	VIC1-330	123,354	Fresh Water
High Volume	January 5-6, 2012	VIC1-635	94,248	Fresh Water

Water Disposal

Water produced during hydraulic fracturing operations, known as flowback water and flush water, is transported by pipeline to the field water treatment plant where it is mixed with other produced water generated on the field. The treated water is then reinjected into the oil and gas producing formations as part of the waterflood process. This operation is in accordance with CSD Condition E.2.(i), which requires that all produced water and oil associated with production, processing, and storage be contained within closed systems at all times. The volume of water in the oil and gas producing zones is much greater than the volumes used for hydraulic fracturing and as such any residual additives are greatly diluted. In addition, many of the chemicals are soluble in oil and would be removed from the subsurface when the oil is sold.

Chemical Listing

Table 3-2 lists the additives that were mixed with the water and sand and injected into the formation during the two high-volume hydraulic fracture operations at the Inglewood Oil Field. Please refer to Appendix B for more detailed information regarding these additives, including volume injected and concentration.

Table 3-2 List of Additives Used During High-Volume Hydraulic Fracture Operations at the Inglewood Oil Field

Additive Type	Trade Name	Typical Main Compound Listed on Material Safety Data Sheet	Purpose
Water		Water	Base fluid carries proppant, also can be present in some additives
Biocide	BE-3S	Propionamide	Prevents or limits growth of bacteria which can cause formation of hydrogen sulfide and physically plug flow or oil and gas into the well
Gel	LGC-38 UC	Guar Gum	Thickens the water in order to suspend the sand
		Naptha hydrotreated heavy	
Breaker	SP Breaker	Sodium Persulfate	Allows for a delayed breakdown of the gel
Crosslinker	BC-140	Borate	Maintains fluid viscosity as a temperature increases

Table 3-2 List of Additives Used During High-Volume Hydraulic Fracture Operations at the Inglewood Oil Field

Additive Type	Trade Name	Typical Main Compound Listed on Material Safety Data Sheet	Purpose
pH Adjusting Agent	MO-67	Sodium Hydroxide	Adjusts pH to proper range for fluid to maintain the effectiveness of other fluid components
Surfactant	Losurf-300M	Ethanol	Aids in recovery of water used during fracturing operation by reducing surface tension
Clay control	Clayfix II Plus	Alkylated quaternary chloride	Clay-stabilization additive which helps prevent fluid interaction with formation clays
		Potassium chloride	
Proppant		Silica	Holds open fracture to allow oil and gas to flow to well

3.2.3 Images from January 2012 Completion Operations

This section presents photographs taken during high-rate gravel packing and high-volume hydraulic fracturing operations conducted at the Inglewood Oil Field in January 2012. The first photo above shows the overall requirements for a high-rate gravel pack completion; they are setting up near the wellhead.

After bringing the vehicles and equipment to the wellhead, hoses and pipes are connected to the various components of the test. The hydraulic fracturing is conducted at elevated pressure, so all components that bear pressure are steep pipes with wall thickness that provides a margin of safety. Hoses are used to connect water, sand, and chemicals prior to mixing and injection.

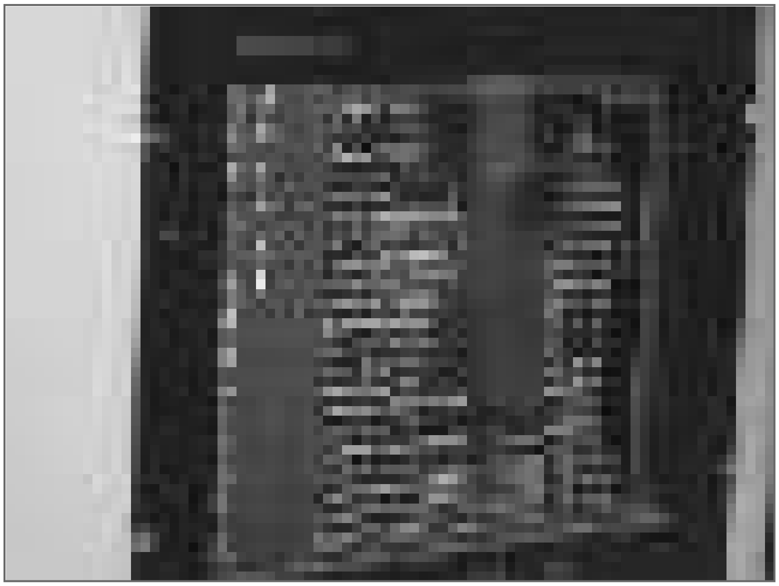
The blender unit is located behind the trailers in this image. The blender mixes the water, sand, and additives prior to introduction into the well for the completion process.

This is an image of the mixture of water, sand and additives used for the high-volume hydraulic fracturing at VIC1-635. This sample of the proppant/gelled water mixture is used to test for consistency with project specifications; samples are taken frequently during the course of the hydraulic fracture treatment for quality control purposes. Note that the food-grade gelling agents hold the sand in suspension, allowing the sand to be introduced into the fractures away from the well. Without the gel, the sand would settle out and not prop open the fractures formed by the completion process. The compound that causes this thickening, guar gum, is an additive used to thicken ice creams for human consumption (Halliburton 2012).

This image shows the VIC1-635 wellhead with a device for isolating the wellhead from the hydraulic fracturing equipment, set up to begin hydraulic fracturing. The green vertical pipe is the wellhead, and the two red pipes attached to the wellhead deliver the water-sand-additive fluid mixture down the well under pressure.

This image shows the VIC1-635 wellhead set up to begin hydraulic fracturing, looking the other direction from the previous image. The water-sand-additive fluid mixture is delivered down the well through the red pipes connected to the top of the well. The image shows the above-ground pumping unit to be connected after the well completion process (hydraulic fracturing), and the amount of equipment needed at the wellhead for hydraulic fracturing job.

A mobile Control Room is placed on site adjacent to the well to be hydraulically fractured at the Inglewood Oil Field. The control room has connections to all of the monitoring, allowing real-time adjustment of the hydraulic fracturing conditions as the job progresses. This control ensures that well integrity, pressures, proppant delivery, and all other attributes of the process can be adjusted to meet downhole conditions. The control room also has a small lab for testing the samples of gelled water and proppant material that are collected for quality control.



This image depicts a screen in the on-site mobile control room, monitoring downhole characteristics of the early stages of the VIC1-635 hydraulic fracture job in progress.

This image shows a graphical display of part of the VIC1-635 hydraulic fracture job in the mobile control room. The image on the screen shows the progress of adding the water-sand-additive mixture; the real-time monitoring using both numerical and graphical displays allows for modification or cessation of the hydraulic fracturing job if the performance does not meet the project design specifications.

3.3 Gravel Packs at the Inglewood Oil Field

In addition to hydraulic fracturing, the Settlement Agreement requires that gravel packing also be described and evaluated in this study. High-rate gravel packing uses water, gravel, and additives to limit entry of formation particles and sand into the wellbore. High-rate gravel packing is a technique that is used for sand control. High-rate gravel packing is ideal in formations that are already permeable. The gravel pack method uses a metal screen placed in the wellbore. The surrounding annulus, or the space between the well and the outer casing, is packed with gravel, water, and additives to limit entry of formation fines and sand into the wellbore. In this process, the space between the formation and the outer casing is packed, at a high-rate, with gravel that is sized small enough to prevent formation grains and fine particles from mixing and entering the wellbore with the produced fluids, but large enough to be held in place by screens. Sand and finer particles reduce the life of surface equipment such as valves, pipelines, and separators. In addition, produced sand can reduce oil production and impair the performance of injection wells.

3.3.1 Past Gravel Packs

Gravel packing is a completion approach that is specifically designed to prevent non-consolidated formation sands from flowing into the wellbore and preventing hydrocarbon production. In gravel packing operations, a steel screen is placed in the wellbore and the surrounding annulus packed with prepared gravel of a specific size designed to prevent the passage of formation sand. The primary objective is to stabilize the formation while causing minimal impairment to well productivity (Schlumberger 2012a). The gravel is circulated into place rather than pumped in under high pressure. Gravel packing does not exceed the fracture gradient.

The process of introducing a gravel pack has gone through several changes over time at the field. Prior to 2003, all of the gravel packs were conducted at pressures below the fracture gradient of the formation. Open hole gravel packs were used until 2003 in the Vickers-Rindge formation and were never installed above the fracture gradient of the surrounding formation. From the mid-1990s to 2003 in the Vickers-Rindge, the technique was modified to a cased-hole gravel pack; this improvement allowed the completion to target the specific producing zone. This had the effect of isolating high saline water producing zones so that the more oil-rich zones could be targeted. This method was also used in the Vickers-Rindge, and was never installed above the fracture gradient of the surrounding formation.

High-rate gravel packs were first used in 2003. At that time, this technique used the cased hole as before, and was limited to a 200-foot target interval as before. This method, however, was the first to exceed the fracture gradient in the surrounding formation, as well as introducing the gravel pack. The fractures would typically be less than 250 feet from the well. Eleven of the initial completions in 2004 used produced crude oil in the fluids in order to be more consistent with the oil in the formation, and potentially yield better well performance; however, analysis of well performance indicated that this was not the case and the use of oil was subsequently stopped. The crude oil had been previously pumped from the formation, and was only used for high-rate gravel packs targeting the oil producing zones. That is, crude oil was never used above, or near, the base of fresh water, but only in oil-bearing formations.

Table 3-3 lists the primary differences between high-rate gravel packs and conventional and high-volume hydraulic fracturing.

Table 3-3 Comparison of High-Rate Gravel Packs to Conventional Hydraulic Fracturing

High-Rate Gravel Packs	Hydraulic Fracturing
Wire wrapped screen is installed in the well	No wire wrapped screen in the well
Goal is not to pump entire sand / proppant volume in the formation but to prevent the entrance of sand into the wellbore	Goal is to pump entire sand / proppant volume into the formation
Sand and water mixture is placed within a short radius of the wellbore (normally 10-50 feet but can reach 250 feet)	Sand and water mixture can be pushed out well in excess of 500 feet from the wellbore

Source: Halliburton 2012

In addition, high-rate gravel pack treatments are usually smaller in terms of sand and fluid volumes and require less time to pump than an average conventional hydraulic fracturing treatment. To illustrate this difference, Table 3-4 provides a comparison of actual sand and fluid volumes pumped in the Inglewood Oil Field during a high-rate gravel pack treatment and the high-volume hydraulic fracturing treatments that were the subject of this study.

Table 3-4 Comparison of Sand and Fluid Volumes between High-Rate Gravel Pack and High-Volume Hydraulic Fracturing at the Inglewood Oil Field

Parameters	High-Rate Gravel Pack	Hydraulic Fracturing
Pump Time (minutes)	27.68	141.87
Clean Volume (bbl)	418.45	2992.18
Slurry Volume (bbl)	458.89	3210.35
Average Treating Pressure (psi)	768	6914
Max Treating Pressure (psi)	1343	8,818
Proppant Mass (100* lb)	373.79	2013.48

Source: Halliburton 2012

Since 2003, PXP has conducted high-rate gravel pack completions on approximately 166 wells in the Inglewood Oil Field, all in the Vickers and the Rindge formation, with a single completion in the Investment Zone (Figure 3-12). Each high-rate gravel pack includes an average of five stages per well; therefore, approximately 830 stages have been completed at the Inglewood Oil Field since 2003.

Halliburton (2012) studied four recent high-rate gravel pack completions in the Vickers and Rindge formations to assess applicability and feasibility: VRU-4243, TVIC-274, Stocker 461, and BC-285. The wells were selected because of their location within the field, including presence on both sides of the Newport-Inglewood Fault. Twenty-one independent high-rate gravel pack treatments from the four wells selected in the Vickers and Rindge zones were analyzed (Halliburton 2012).

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Figure 3-12
High-Rate Gravel Pack Completions
at the Inglewood Oil Field

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The results of the analysis showed the following:

- The fracture height created by the high-rate gravel packs in the Vickers and Rindge formations was, on average, in the range of 100 to 170 feet for the majority of the stages. The fracture height in several stages was around 200 to 240 feet.
- Fracture height is very small in relation to the depth of the fracture.
- The top of the created fracture is at least 1,000 feet below the bottom of the deepest perched water zones in the area that includes the Inglewood Oil Field.

Figures 3-13A and 3-13B present different visualizations of the fracture geometries produced by the high-rate gravel packs. The figures also show the relevant formation surfaces, ground surface, geologic structure including major faults, and groundwater-bearing bodies near the surface.

Figure 3-13A Side View Showing Modeled Fracture Geometries in the Vickers Zone

Figure 3-13B Side View Showing Modeled Fracture Geometries in the Vickers Zone and Structure (Faults)

3.3.2 Recent High-Rate Gravel Pack Completions

PXP also conducted high-rate gravel pack jobs at two wells on the Inglewood Oil Field to assess feasibility and potential impacts. The first high-rate gravel pack involving a five-stage completion was conducted on January 9, 2012, at the TVIC-221 well. The second high-rate gravel pack involved a six-stage completion and was conducted on the same day at a different well, TVIC-3254. Both of these operations were conducted in the Vickers and Rindge formations. The high-rate gravel pack operations were conducted by Halliburton with PXP oversight. The conditions of the high-rate gravel packs are similar to the well completions previously conducted across the field, and are also similar for any future high-rate gravel pack jobs that would be expected to be conducted at the oil field.

The maximum applied pressure during both high-rate gravel packs was 1,900 psi. In comparison the high-volume hydraulic fracturing projects had an average treatment pressure of 2,971 psi (VIC1-330) and 6,914 psi (VIC1-635). The high-rate gravel pack fracturing influences the zone within 100 to 250 feet of the well within the target oil-producing zone, compared to in excess of 500 feet for hydraulic fracturing.

Water and Chemical Use during High-Rate Gravel Pack

Water Demand/Source

Water for high-rate gravel packs at the Inglewood Oil Field has been provided either from produced water at the field or, if a potassium-chloride gel is used, fresh water provided by California American Water Company (the water service provider for all fresh water on the oil field). The majority of the high-rate gravel pack operations that have occurred since April 2011 have used produced water from the lease, including the two high-rate gravel pack examined in this study. Table 3-5 provides the volumes of water used during the two high-rate gravel pack fracture jobs at the Inglewood Oil Field.

Table 3-5 Volumes of Water Used During High-Rate Gravel Pack Hydraulic Fracturing Operations at the Inglewood Oil Field

Operation Type	Date	Well	Volume Water Used (gallons)	Water Source
High-rate gravel pack	January 5-6, 2012	TVIC 3254	33,357	Produced Water
High-rate gravel pack	January 5-6, 2012	TVIC 221	55,247	Produced Water

Chemical Listing

Table 3-6 below, lists the materials that have been injected into the formation during the high-rate gravel pack operations at the Inglewood Oil Field.

Table 3-6 List of Additives at Used During High-Rate Gravel Pack Operations at the Inglewood Oil Field

Additive Type	Trade Name	Typical Main Compound	Purpose
Water		▪ Water	Base fluid carries proppant, also can be present in some additives
Buffering Agent	BA-40L	▪ Potassium carbonate	pH buffer
Gel	LGC-36 UC	▪ Guar Gum ▪ Naphtha hydrotreated heavy	Thickens the water in order to suspend the sand
Breaker	SP Breaker	▪ Sodium Persulfate	Allows for a delayed breakdown of the gel
Crosslinker	K-38	▪ Disodium octoborate tetrahydrate	Maintains fluid viscosity as a temperature increases
pH Adjusting Agent	MO-67	▪ Sodium Hydroxide	Adjusts pH to proper range for fluid to maintain the effectiveness of other fluid components
Activator	CAT-3	▪ Copper chelate	Reduces viscosity
Surfactant	Losurf-300M	▪ Ethanol	Aids in recovery of water used during fracturing operation by reducing surface tension
Clay control	Clayfix II Plus	▪ Alkylated quaternary chloride ▪ Potassium chloride	Clay-stabilization additive which helps prevent fluid interaction with formation clays
Proppant		▪ Silica	Holds open fracture to allow oil and gas to flow to well

Water Reuse

As described for the high-volume hydraulic fracture operations, water produced during high-rate gravel pack operations is transported by pipeline to the field water treatment plant where it is mixed with other produced water generated on the field. The treated water is then reinjected into the oil and natural gas producing formations as part of the waterflood process. This operation is in accordance with CSD Condition E.2.(i) which requires that all produced water and oil associated with production, processing, and storage are contained within closed systems at all times. The volume of water in the oil and gas producing zones is much greater than the volumes used for hydraulic fracturing and as such any residual additives would be greatly diluted. In addition, many of the chemicals are soluble in oil and would be removed from the subsurface when the oil is sold.

3.4 Anticipated Future Use of Hydraulic Fracturing and Gravel Packing at the Inglewood Oil Field

PXP expects that, in the future, high-volume hydraulic fracturing and conventional hydraulic fracturing may be conducted in the relatively deep Rubel, Moynier, Bradna, City of Inglewood, Nodular, and Sentous zones (all located deeper than 4,000 feet below ground surface).

It is anticipated that high-rate gravel packing operations may be conducted on as many as 90 percent of all future production wells drilled in the Vickers and Rindge formations on the Inglewood Oil Field, as well as other permeable sandstone completions. This procedure results in less formation sand being drawn into the well during pumping, thus, the amount of formation sand that requires management at the surface is reduced and the procedure provides a longer life to the well.

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Chapter 4

Environmental Effects Monitored in Conjunction with Hydraulic Fracturing Tests

4.1 Introduction

As described in Chapter 4, the Inglewood Oil Field has an ongoing program of environmental monitoring and reporting for environmental factors such as water quality, air quality, and others. As part of this Hydraulic Fracturing Study, these ongoing monitoring programs were augmented to include other monitoring of environmental factors of importance. This chapter summarizes the results of this comprehensive environmental monitoring, as follows:

- Hydrogeology, Water Quantity and Quality
- Containment of Fractures to the Desired Zone
- Well Integrity
- Slope Stability, Subsidence, Ground Movement, Induced Seismicity
- Methane
- Other Emissions to Air
- Noise and Vibration
- Los Angeles County Department of Public Health Study

In addition, since high-volume hydraulic fracturing was first used for shale gas development in the northeastern United States and tight sands development in the Intermountain West, there has been extensive coverage of controversies surrounding its use. Although most of the news has been about the development of shale gas, tight sands and coalbed methane deposits rather than the type of oil and natural gas development that occurs at the Inglewood Oil Field, community outreach conducted as part of this study indicated that many of the concerns surrounding shale gas development are shared by the local community. Therefore, in this chapter we present the methods and results of environmental monitoring conducted at the Inglewood Oil Field, and we provide context by describing how these issues have been described as they relate to shale gas development. Although the focus of this chapter is the Inglewood Oil Field, the issues have been framed by the national attention given to shale gas development elsewhere in the country; thus, the context is relevant to the Inglewood Oil Field.

4.2 Hydrogeology, Water Quantity and Quality

4.2.1 Geologic Control on the Distribution of Groundwater-Bearing Zones

The geology of the Baldwin Hills constrains the occurrence and movement of groundwater, as described in the USGS groundwater model of the Los Angeles Basin (USGS 2003), the California Department of Water Resources groundwater assessment of the Los Angeles Basin

(DWR 1961), and studies specific to the Inglewood Oil Field summarized in this study. The USGS excludes the Baldwin Hills from the model domain, separating it by a no-flow boundary. The no-flow boundary condition means that groundwater neither flows in to or out of the Baldwin Hills; it is isolated from the remainder of the Los Angeles groundwater basin (Figure 4-1). In the definitive account of the groundwater geology of the Los Angeles Basin, the Department of Water Resources concludes that "the Baldwin Hills form a complete barrier to groundwater movement, where the essentially non-water bearing Pico Formation crops out" (DWR 1961). The results of the extensive site-specific study of the Baldwin Hills, including a groundwater monitoring array that traverses the entire zone of potential fresh water, summarized in this section, are in complete agreement with the findings of the USGS and DWR.

Figure 4-2 presents the standard model of the geology of the Baldwin Hills (Wright 1991), from the surface to a depth in excess of two miles. The center of the figure has a small area labeled "*See Figure 4-3C*" which represents the uppermost 500 feet at the Baldwin Hills; this area will be magnified in stages from Figure 4-3B to 4-3C.

Figure 4-3A shows the locations of all of the groundwater monitoring wells installed on the Inglewood Oil Field. All the oil producing formations, from the Investment Zone downwards, contain water too saline for direct use at the surface. Only the upper 500 feet, above the top of the Pico Formation, has any fresh water, albeit limited in extent and yield. For this reason, the top of the Pico Formation is known as the base of fresh water.

Figure 4-3B is the first level of magnification and shows the freshwater zones. In all parts of the world, fresh (not salty) groundwater lies at relatively shallow depths. At greater depths the water is saline, not drinkable, and is sometimes called formation water. The USEPA makes this distinction in the Safe Drinking Water Act by protecting the shallow, fresh water from contamination by deeper, saline formation water. In most of the Los Angeles Basin, the base of the fresh water zone, below which saline formation water is found, is defined by a geological unit called the Pico Formation, a marine unit shown in Figure 4-3B (developed from USGS 2003). Overlying the Pico Formation are the aquifer systems in the Los Angeles Basin located away from the Baldwin Hills: the Inglewood Formation, the Silverado Formation, and the Lakewood Formation. In many parts of the Los Angeles Basin, these formations are aquifers for water supply wells. The box labeled "*See Figure 4-3C*" depicts how these formations became folded and faulted in the geological uplift of the Baldwin Hills (within the labeled box). As a result of this uplift, these formations are not water-bearing beneath the Baldwin Hills, and are in fact exposed at the surface. Their disruption by the uplift of the Baldwin Hills has disconnected them from the groundwater-bearing formations of the Los Angeles Basin (USGS 2003, DWR 1961).

Finally, Figure 4-3C shows the uppermost 500 feet beneath the Baldwin Hills. The ground surface is shown as the undulatory line at the top of the figure. The vertical black lines represent groundwater wells drilled at the Inglewood Oil Field from 1993 to present. Although they are shown along a single line in the figure, the wells are distributed across the oil field; their locations have been projected to a single line to aid in the presentation of the results. The length of the line shows the depth of drilling. If any groundwater was detected, the depth is shown with an upside-down triangle and the estimated extent of the groundwater is shown with the blue color surrounding the well (black line). If no groundwater was detected, that observation is noted with an upside-down triangle with a red circle around it, and a red cross-out.

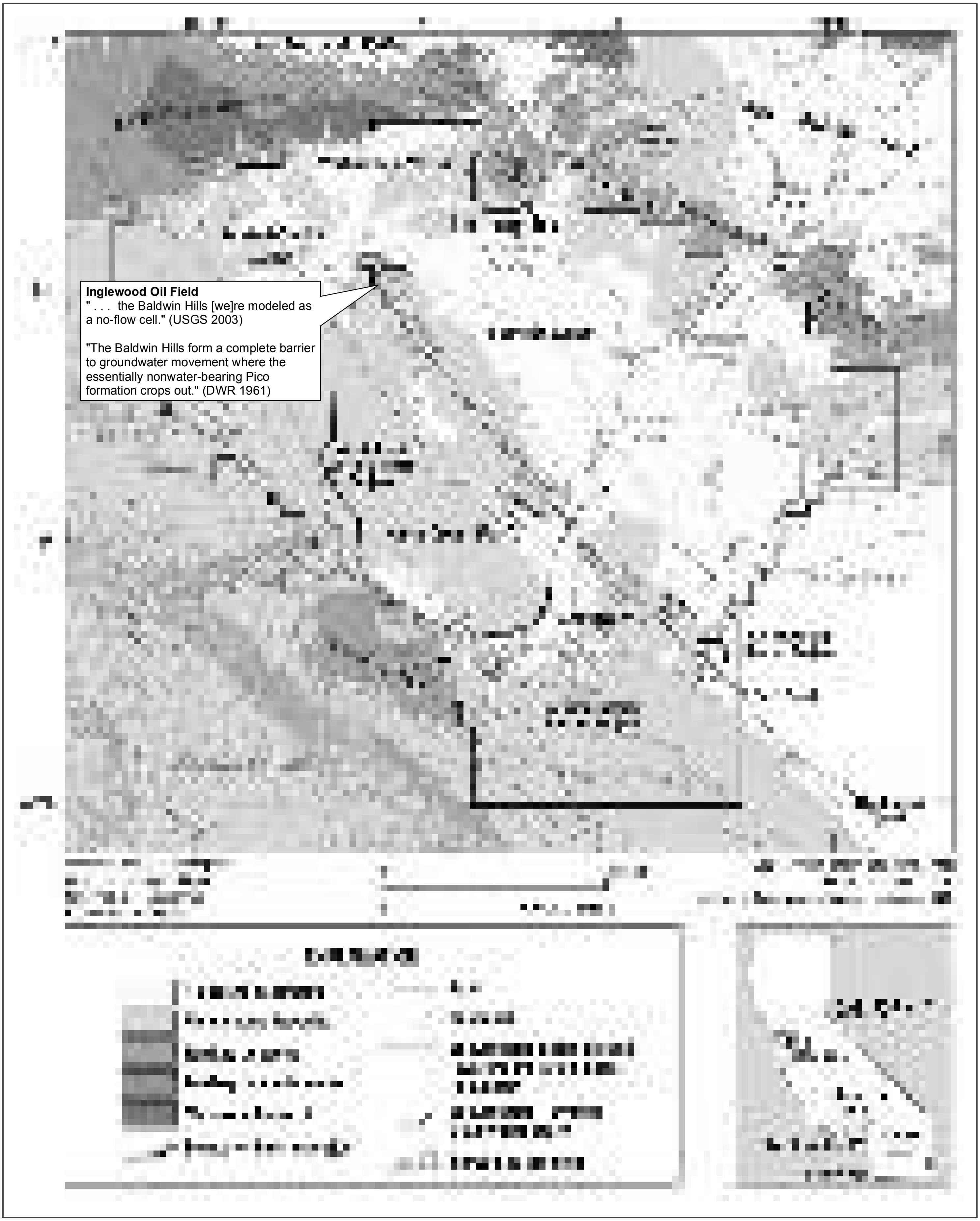

Figure 4-1
Groundwater Basins in the Vicinity of the Inglewood Oil Field

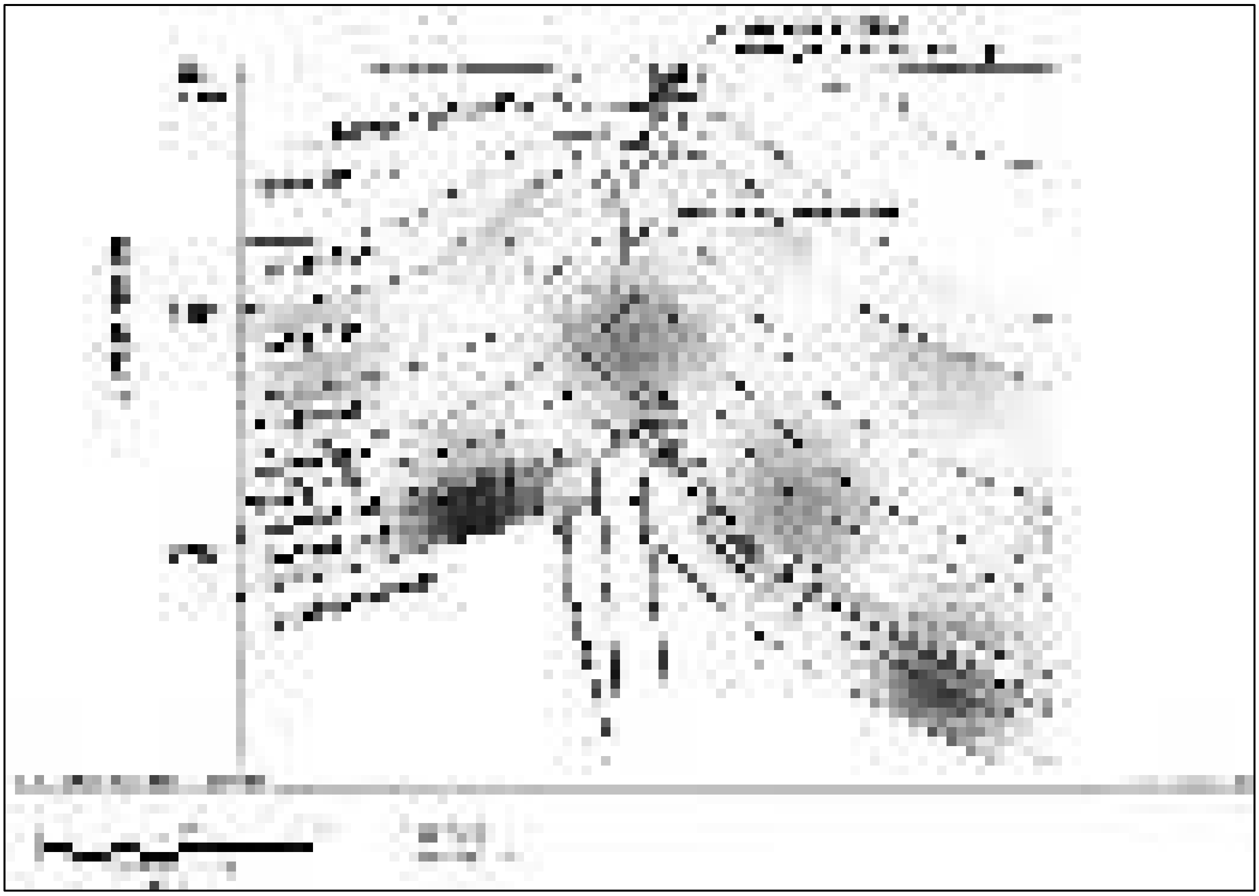

Figure 4-2 Location of Inglewood Oil Field in Relation to Known Fault Lines

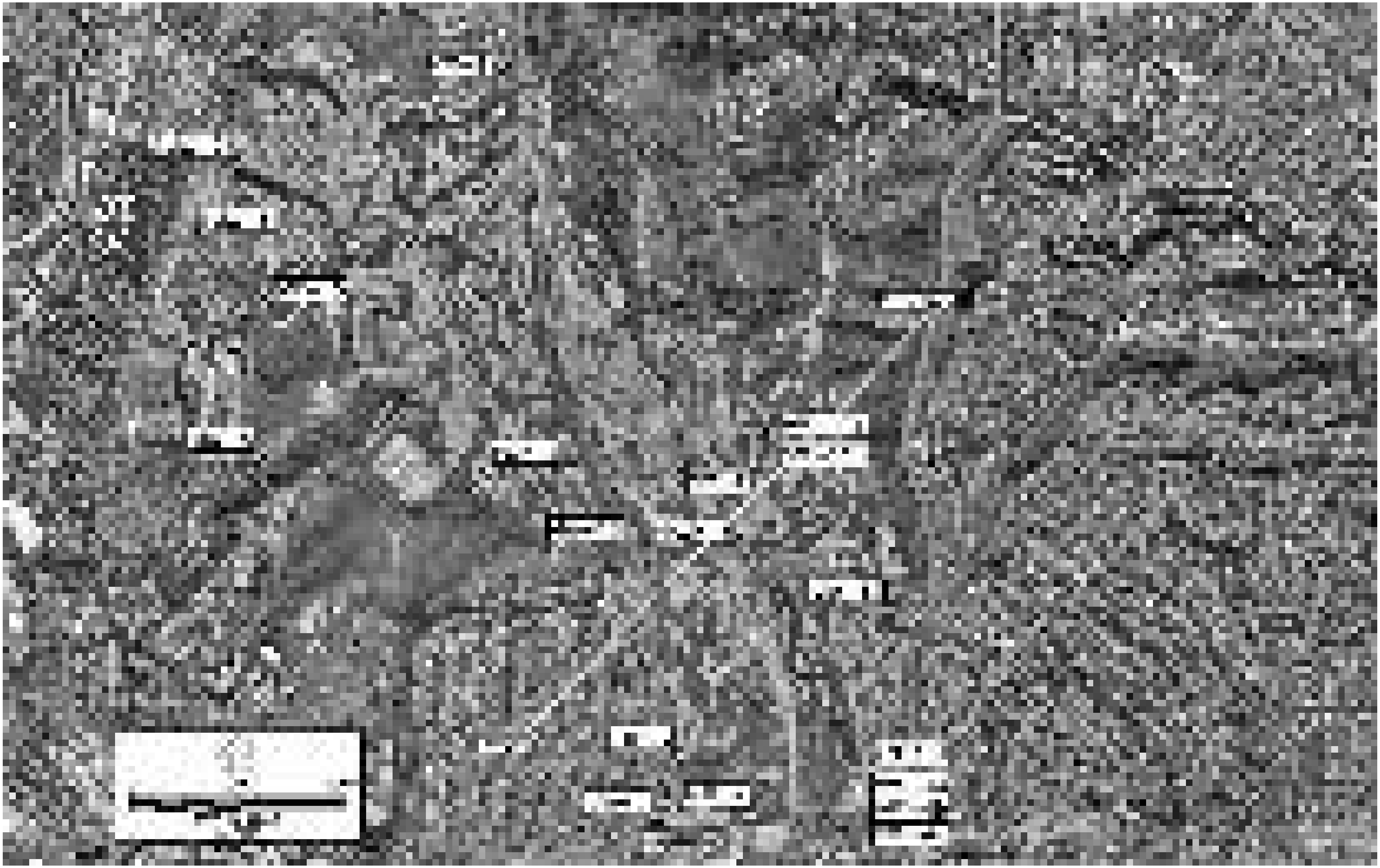

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Figure 4-3A

Cross Section Location

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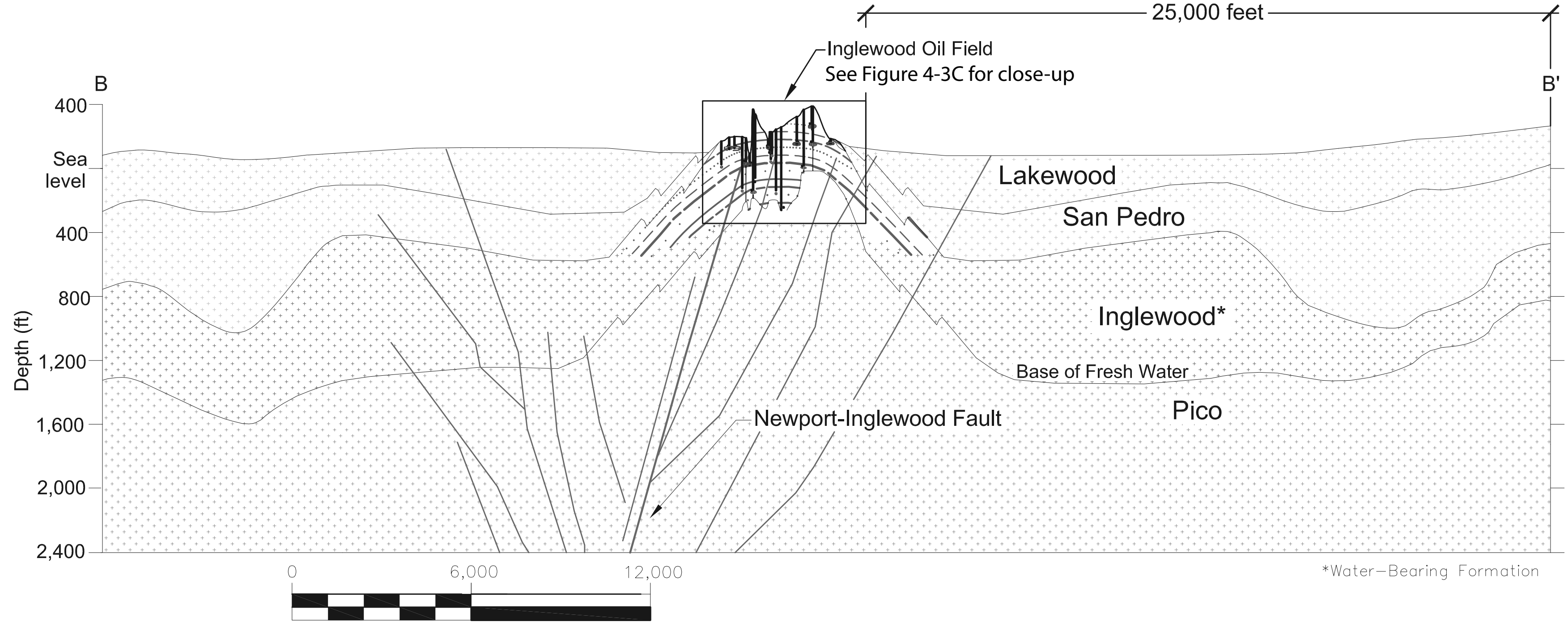

LEGEND

- Fault
- Fold Axis
- Geologic Formation
- Cross–Section

NOTES: 1. Vertical scale greatly exaggerated.
2. Both San Pedro and Inglewood formations are present at the oil field. Depiction is an estimate, as both formations crop out and have complex folds.
3. Cross–section depicted located as shown in inset map.

Source: USGS 2003

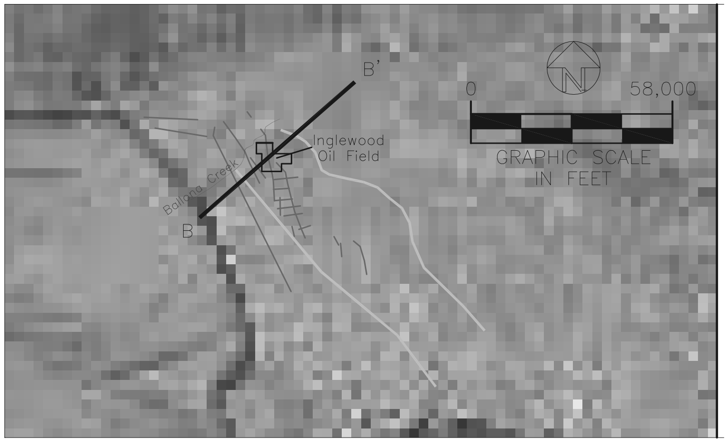

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Figure 4-3B

Monitoring and Drinking Water Wells in the Vicinity of the Inglewood Oil Field

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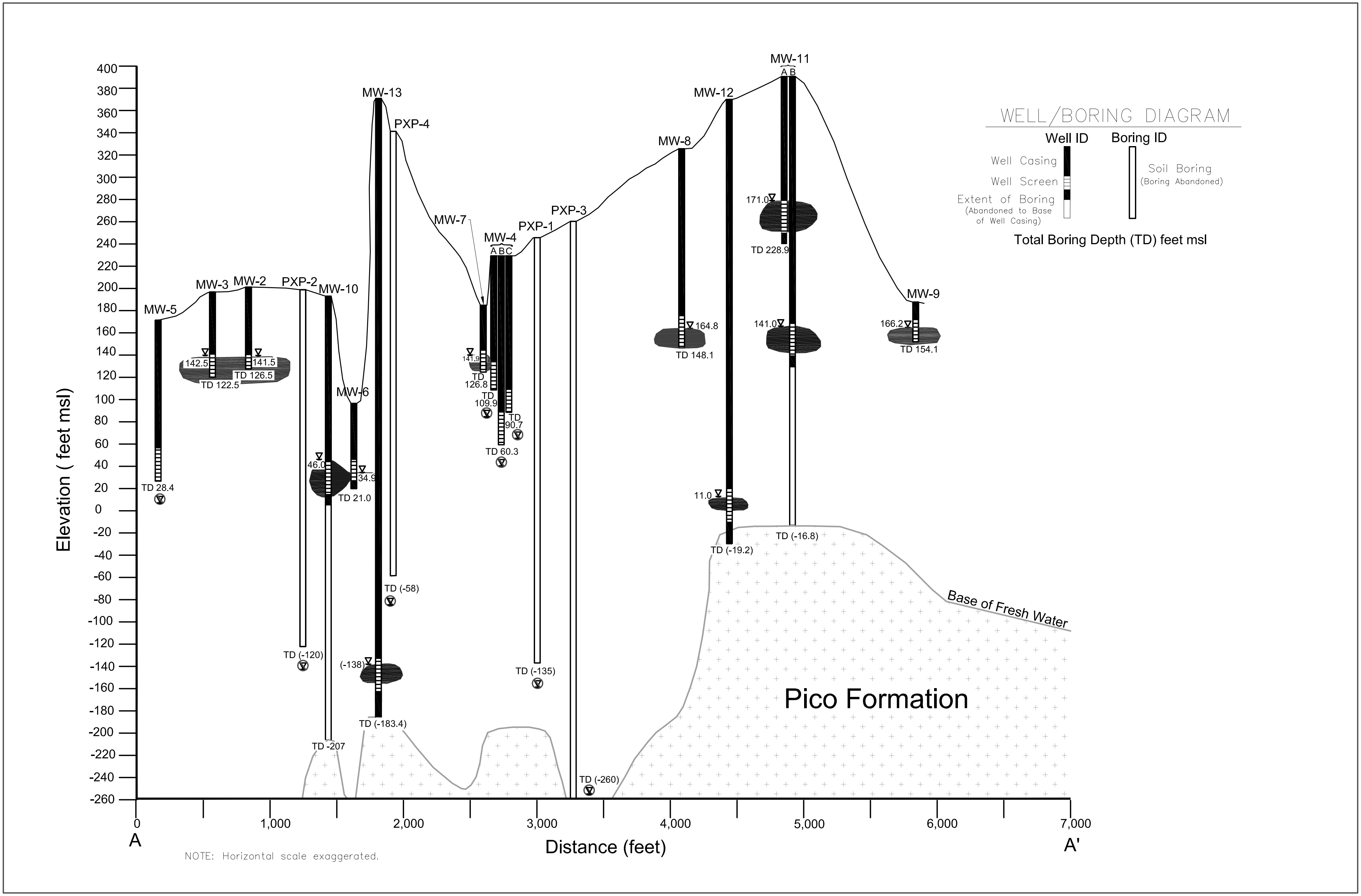

LEGEND

No Groundwater Found

Static Groundwater Level (feet msl)

Groundwater

Pico Geologic Formation

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Figure 4-3C

Groundwater Beneath the Oil Field

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Figure 4-3C is based on measured conditions beneath the field, with 15 groundwater monitoring wells and four deep supply wells that were attempted by PXP, but that did not encounter water and were abandoned. Note that the same characteristics shown in Figure 4-3C are also shown in more detail by the 3-D model prepared by Halliburton in Figure 2-8H. The data show that the water bearing zones are not continuous across the field because they occur at different depths, or do not occur at all.

All of the wells have very low yield; the shallow wells and all but two of the deeper wells pump dry in less than 30 minutes at low pumping rates. The two that can sustain higher initial pumping rates show declining yields when pumped for more than a day, indicating that the water bearing zone from which they draw is limited in extent. None of the units show evidence of a connection to the aquifers of the Los Angeles Basin (shown in Figure 4-3C).

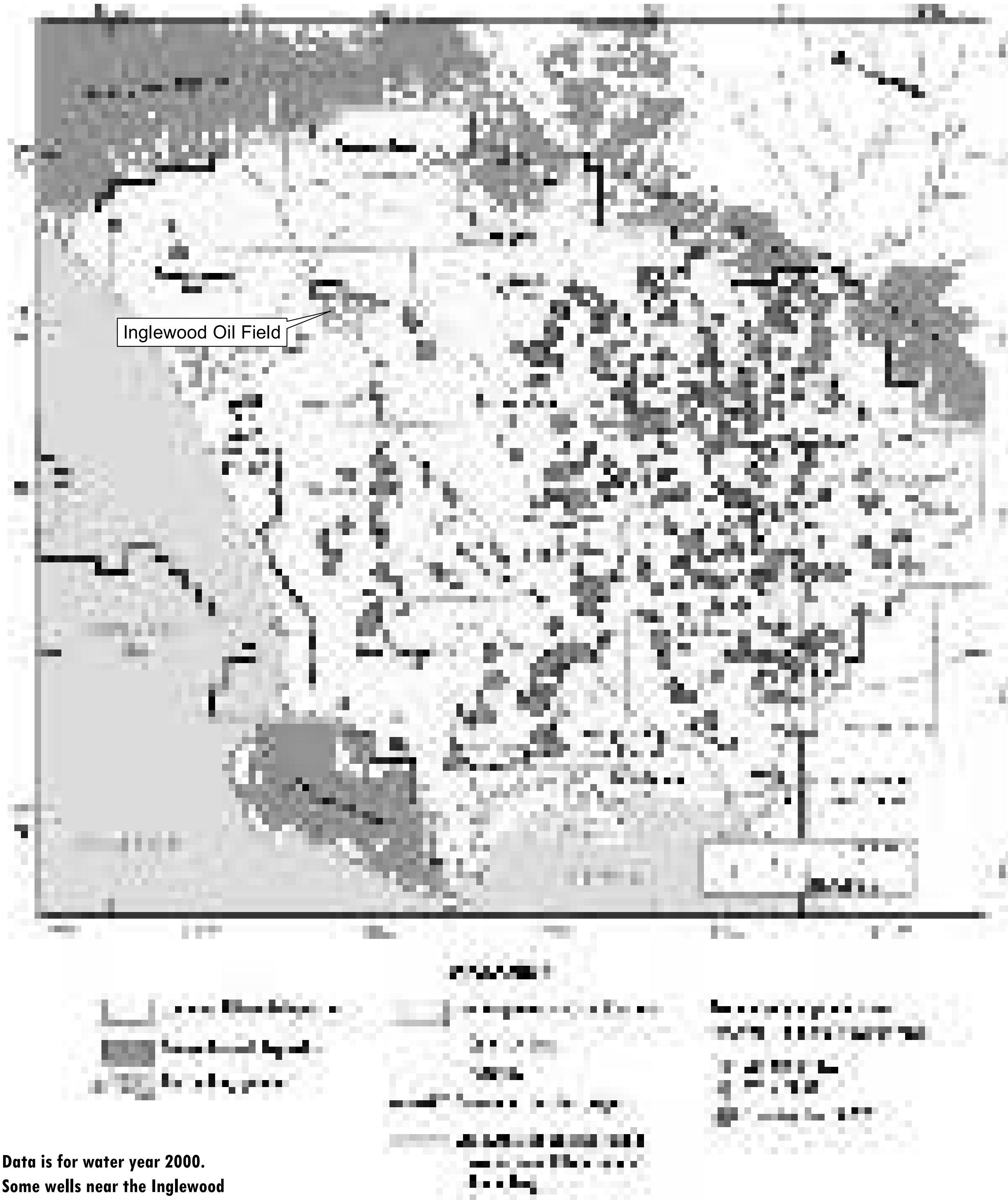
4.2.2 Hydrogeology

The description of conditions beneath the Inglewood Oil Field depicted in Figure 4-3C and described in the previous section is based on data collected from 15 groundwater monitoring wells installed to test for the presence and quality of groundwater beneath the site, ranging in depth from 30 feet to 500 feet beneath the ground surface. The four deepest wells were installed to reach the "base of the fresh water zone", that is, the top of the Pico Formation. As such, the understanding of groundwater hydrogeology and water quality is based on investigation of the *entire* zone beneath the surface that has any potential to contain fresh water.

The Baldwin Hills are generally comprised of non-water-bearing rock layers that straddle the West Coast, Central, and Santa Monica groundwater basins (Figure 4-4). As shown in this figure, where groundwater is pumped for groundwater supply, it is principally in areas to the east of the Baldwin Hills. The data used by the USGS to construct this figure were based on water year 2000; currently, the only wells in the vicinity of the Inglewood Oil Field are either no longer active (Environmental Data Resources, Inc. 2012) or located greater than 1.5 miles from the field boundary.

Studies of the Baldwin Hills have concluded that the tectonic uplift of the Baldwin Hills by folding and faulting has disconnected water-bearing sediments from groundwater supplies in the Los Angeles Basin (USGS 2003, DWR 1961). The geological formations that may produce usable quantities of groundwater from aquifers elsewhere in the Los Angeles Basin are folded, faulted, and either dry or have practically no water supply potential beneath the Baldwin Hills. Because of a lack of water, the geological formations beneath the Baldwin Hills are not suitable for water supply (DWR 1961, USGS 2003, County of Los Angeles 2008).

For example, the prominent aquifers in some portions of the Los Angeles Basin, which lie greater than 100 feet below the surface in the flat portion of the Los Angeles Basin (refer to Figure 4-1, Figure 4-3B), have been brought to the surface by folding and faulting of the Baldwin Hills. The units are exposed at the surface, do not contain water, and are not connected to the surrounding basin. In groundwater models of freshwater flow in the Los Angeles Basin aquifer systems recently prepared by the U.S. Geological Survey (USGS 2003), the Baldwin Hills is modeled as a "no flow" zone; that is, since the sediments beneath the Baldwin Hills are disconnected from the regional aquifers, groundwater flow is discontinuous across the Baldwin Hills.



Data is for water year 2000.
Some wells near the Inglewood Oil Field are no longer active.

Source: USGS 2003

The five water wells (MW-10, MW-11A, MW-11B, MW-12, and MW-13) drilled down to the base of fresh water in 2012 are intended to provide data for the deepest freshwater zones in the Baldwin Hills. Only two encountered water at the deepest levels, and were completed in the only zones containing water. Maps showing the top of the Pico used for oil exploration defined the top of the Pico Formation, and drilling progressed to that depth, in some places up to 500 feet below the ground surface. In addition, geophysical logs were run after drilling to thoroughly search for water. Two of the locations had only a single thin water-bearing zone, and the wells quickly pumped dry at low flow rates. One location had water near the top of the Pico Formation and was initially capable of sustaining pumping rates of eight gallons per minute. Over three days of pumping the yield diminished significantly, indicating a limited areal extent of the water-bearing zone. The fourth location of the deep water wells identified two thin water-bearing zones. One pumped dry readily, while the other well initially sustained pumping rates of one gallon per minute. Over three days of pumping the yield diminished significantly indicating a limited areal extent of the water-bearing zone.

At the Inglewood Oil Field perched groundwater (groundwater that is discontinuous and occurs in small pores within the rock layers) has been measured at depths ranging from approximately 45 to 500 feet below ground surface (Figure 4-3C). Groundwater monitoring on the field suggests that rainfall and irrigation water from nearby residences appear to be the only source of this groundwater because water levels respond to the presence of water in catch basins.

4.2.3 Water Quality

Regulatory Limits

The LARWQCB Water Quality Control Plan, or Basin Plan, establishes beneficial uses of surface and groundwater in the Los Angeles Basin. Based on the State Board Resolution No. 88-63, "Sources of Drinking Water Policy", all groundwater in the state must be considered a potential source of drinking water, and carry a beneficial use designation of Municipal Supply (or MUN). The only exceptions are as follows:

- Total Dissolved Solids (TDS) exceeds 3,000 mg/l (5,000 µS/cm electrical conductivity);
- Contamination exists, that cannot reasonably be treated for domestic use; and
- The source is not sufficient to supply an average sustained yield of 200 gallons per day.

The LARWQCB would also require that, for any groundwater to have the MUN designation removed that there be a formal process to de-designate the aquifer. There are only two de-designated areas in the LARWQCB jurisdiction: limited coastal areas beneath the Port of Los Angeles and the Port of Long Beach, and a limited area of El Segundo seaward of a series of injection wells to limit saltwater intrusion. Because the Inglewood Oil Field has not gone through a de-designation process, any water that may be encountered beneath the field has the beneficial use designation of MUN, and the drinking water standards are applied regardless of the low yield.

Chemical Disclosure and Environmental Effects of Chemical Additives

When new oil and natural gas development using high-volume hydraulic fracturing was initially introduced in the northeastern United States (areas dependent on shallow groundwater resources or water from relatively pristine watersheds), public concern was initially related to the policy of oilfield service companies to maintain confidentiality of the precise chemical names and concentrations used in hydraulic fracturing fluids. The initial lack of full disclosure on the part of the oilfield service companies increased public concerns about the chemicals. As a result of these concerns, several states initiated independent reviews of the environmental impacts of hydraulic fracturing with an emphasis on water quality and chemical disclosure. In addition, USEPA initiated two ongoing reviews, one focused on the potential effects of hydraulic fracturing on drinking water supplies, and the other focused on the definition of "diesel fuel" as part of a review of the 2005 EPAct provisions (USEPA 2011c, USEPA 2011d). The 2005 EPAct reaffirmed that hydraulic fracturing is a well completion process regulated at the state level, and therefore does not require an underground injection control permit under the Safe Drinking Water Act. The 2005 EPAct did require a UIC permit in cases where diesel fuel is used in hydraulic fracturing fluid.

Since the passage of the EPAct, many states have adopted regulations or passed legislation requiring operators to disclose the composition of the fluids used in the hydraulic fracturing process. In California, legislation (AB 591 - Wieckowski) requiring operators to post a complete list of the chemical constituents used in the hydraulic fracturing process was introduced during the 2011 – 2012 legislative session but failed to pass. The bill would have required that operators involved in hydraulic fracturing provide a complete list of the chemical constituents used in the hydraulic fracturing fluid, as well as the following additional information to DOGGR:

- the source and amount of water used in the exploration or production of the well;
- data on the use, recovery and disposal of any radiological components or tracers injected into the well; and
- if hydraulic fracturing is used, disclosure of the chemical information data described above.

The chemical additives used in hydraulic fracturing, typically 0.5 percent of the total fluids, are necessary to ensure that the fracturing job is as effective and efficient as possible. The various chemicals are used as friction reducers, biocides to prevent microorganism growth, oxygen scavengers and other stabilizers to prevent corrosion of metal pipes, and acids to remove drilling mud damage. The consequences of not using additives in the fluids include higher engine emissions as a result of greater loads, increasing pipe corrosion (and, in turn, compromised integrity of the well), increased water use, and decreased hydrocarbon recovery.

PXP reported the full chemical listing of the two recent high-volume hydraulic fracture operations on the FracFocus.org website. This website offers the opportunity to comply with the standard chemical disclosure regulations in effect in other parts the country. The level of disclosure used in this Hydraulic Fracturing Study would comply with the terms of AB 591 as drafted at the time this study was prepared. Diesel fuel is not used as a chemical additive for the hydraulic fracturing conducted at the Inglewood Oil Field; therefore, a UIC permit is not required.

Quarterly Water Quality Testing Prior to High-Volume Hydraulic Fracturing

Fifteen groundwater wells to test for the presence and quality of groundwater beneath the Inglewood Oil Field have been drilled since 1993. These vary in depth from 30 feet to 500 feet below the ground surface. In addition to the 15 groundwater monitoring wells, four wells were drilled as potential water supply wells for the oil field, but because they were dry, the wells were abandoned and sealed. Of the fifteen wells, six did not encounter groundwater but the wells remain in place. Four were installed in 2012 and were drilled to the base of fresh water in order to characterize the entire fresh water zone. The remaining five all sample the shallowest water on the field and are monitored on a quarterly basis, in accordance with CSD Condition 19.

Quarterly monitoring reports for 2010 and 2011 provide a baseline indication of existing groundwater quality. Prior to the hydraulic fracturing operations of January 2012, a total of nine monitoring wells were tested as part of the groundwater monitoring effort that took place on November 22, 2011. Specifically, the monitoring effort involved wells MW-2, MW-3, MW-4A, MW-4B, MW-4C, and MW-5, MW-6, MW-7, and MW-8 as part of the monitoring program. The sampling involved the collection of depth-to-water measurements and groundwater samples from monitor wells MW-2, MW-6, MW-7, and MW-8. Monitor wells MW-3, MW-4A, MW-4B, MW-4C and MW-5 were not sampled since they were dry or contained insufficient water at the time the monitoring was conducted.

Groundwater analytical results indicated no results were above the California Maximum Contaminant Level (MCL), with the exception of arsenic. Arsenic levels are believed to correspond to naturally occurring arsenic found in soil and rock formations throughout the Los Angeles Basin. As documented by USEPA, when "compared to the rest of the United States, western states have more systems with arsenic levels greater than the [US]EPA's standard of 10 parts per billion (ppb)" (USEPA 2012a). Arsenic delineation maps produced by the USGS in 2011 have documented increased levels of arsenic in both Los Angeles County and Southern California as a whole (Gronberg 2011). These data are also consistent with soils data from the 2008 California Department of Toxic Substance Control (DTSC) memo "Determination of a Southern California Regional Background Arsenic Concentration in Soil" (Chernoff et al. 2008). Areas in Southern California have been shown to have higher than average levels of arsenic present in soil and thus, through the release of naturally occurring arsenic in sediments, levels can be inferred to also be higher than average in groundwater resources throughout Southern California.

A summary of the baseline groundwater analytical results is as follows:

- TDS was measured at 590 mg/L in MW-2, 2,000 mg/L in MW-6, 2,500 mg/L in MW-7, and 1,500 mg/L in MW-8.
- pH was measured at 7.5 in MW-2, 7.0 in MW-6, 7.0 in MW-7, and 7.0 in MW-8.
- BOD_5 was measured at 38.5 mg/L in MW-2, 30.4 mg/L in MW-6, 25.6 mg/L in MW-7, and 15.7 mg/L in MW-8.
- Low levels of TPH in MW-2, MW-6 and MW-7. The silica gel filtering method, which removes nonpetroleum materials such as fats, was run on all groundwater samples. Results indicate TPH concentration of 0.35 mg/L in MW-2, and below the detectable limit of

0.10 mg/L in wells MW-6 and MW-7. These levels are within the range of drinking water standards for taste and odor commonly applied for TPH (between 0.050 and 1 mg/L).

- TRPH were below the detection limit of 0.50 mg/L in all samples.
- BTEX and MTBE were below detection limits in all samples.
- Nitrate was detected at a concentration of 0.34 mg/L in MW-2 and 3.8 mg/L in MW-7, both below the state MCL of 45 mg/L.
- Barium was detected in MW-6 at a concentration of 56 µg/L, MW-7 at a concentration of 60 µg/L, and in MW-8 at a concentration of 170 µg/L. These levels are all below the state MCL of 1,000 µg/L.
- Arsenic was detected at a concentration of 37 µg/L in MW-2 and 4.2 µg/L in MW-8. The concentration of arsenic in MW-2 is above the state MCL of 10 µg/L and is likely due to naturally occurring arsenic found in soil and rock formations as described previously in this section.

4.2.4 Groundwater Monitoring Associated with High-Volume Hydraulic Fracturing

The groundwater monitoring wells have been sampled twice since the high-volume hydraulic fracturing was completed. Results from baseline (pre-hydraulic fracturing) groundwater sampling are compared with results of sampling the same wells after hydraulic fracturing. Since the deep wells do not have a baseline, the results of two rounds of sampling the deep wells are compared to the same two rounds of the pre-existing wells.

The Inglewood Oil Field is required to sample and analyze groundwater on a quarterly basis in compliance with CSD Section E.19. This sampling will continue irrespective of whether hydraulic fracturing operations are conducted in the future, but this study focuses on the sample rounds at the end of 2011 (pre-hydraulic fracturing) compared to the two rounds of samples collected so far in 2012 (post-hydraulic fracturing).

Comparison of Baseline to Post-Hydraulic Fracturing Operation Water Quality

The monitoring wells with sample results prior to 2012 were sampled for the same analytes after the occurrence of hydraulic fracturing in January. The water was analyzed for the following constituents: pH, TPH, benzene, toluene, ethylbenzene, total xylenes, methyl-tributyl ethylene (MTBE), total recoverable petroleum hydrocarbons (TRPH), total dissolved solids (TDS), nitrate, nitrite, metals, and biological oxygen demand (BOD_5). These chemicals include compounds in the hydraulic fracturing fluids.

Analysis of samples taken post-fracturing (April and August 2012) indicated no results above the state MCL for any constituents, with the exception of arsenic, which is likely due to naturally occurring arsenic found in soil and rock formations that was present prior to fracturing. These results are consistent with earlier, pre-hydraulic fracturing sample results from these wells.

In most cases there were either no changes in the concentrations of the analytes sampled for, or there was a decrease in concentration after hydraulic fracturing. One of the wells, MW-7, had an increase of one compound, chromium, in the samples after the hydraulic fracturing (2.7 µg/L to

3.0 µg/L, both results well below the 50 µg/L state standard). Chromium is not associated with hydraulic fracturing additives.

The following analytes showed no major changes in concentration when comparing the data that was obtained prior to and after hydraulic fracturing:

- pH
- TPH-DRO[1]
- TRPH
- TDS
- Nitrite
- Cobalt

The following analytes were below detection prior to the hydraulic fracturing, and then showed concentrations above the detection limit in January 2012, with levels returning to below detection in April 2012 and remaining below detection in August 2012. All analytes were below the state MCL.

- MW-6 and MW-7 TPH-DRO (with Silica Gel)
- MW-3 Benzene, Toluene, Ethylbenzene, Total Xylenes, and MW-8 Toluene
- MW-2 and MW-6 Zinc

The following analytes showed an instance where there was a slight concentration increase after the hydraulic fracturing:

- MW-7 Chromium showed an increase from 2.7 to 3.0 µg/L. Both results are well below the 50 µg/L state standard. Chromium is not associated with hydraulic fracturing additives.

The following analytes showed a decrease after hydraulic fracturing:

- MW-2 and MW-6 Nitrate
- MW-7 Barium
- MW-8 Arsenic
- MW-7 Copper showed below detection limit immediately after wells were installed, then had a slight increase in concentration 6 months later, before returning to below the detection limit.
- MW-8 Lead showed below detection limit immediately after wells were installed, then had a slight increase in concentration 6 months later before returning to below the detection limit.

In summary, the hydraulic fracturing did not have a detectable change to groundwater quality based on the comparison of baseline results to post-hydraulic fracturing results. Any variations are within the ranges detected over the course of the monitoring. Figure 4-5 summarizes the monitoring results. The horizontal axis lists any detected compounds. The vertical axis divides the amount detected by the drinking water standard for that compound; a value of 1 means that the detected amount equaled the drinking water standard. The horizontal line corresponding to 1 on the vertical axis divides the chart between detections that meet the drinking water standard

[1] With the exception of MW-8, which showed below the detection limit after the well was installed in February 2012 then a slight increase to .34 mg/L in April 2012, before returning to below the detection limit in August 2012.

(all compounds except arsenic) from arsenic, which has a high background in Southern California that exceeds the drinking water standard.

Comparison of New Wells to Pre-Existing Wells

In addition to the existing monitor well array prior to hydraulic fracturing, five new wells (MW-10, MW-11A, MW-11B, MW-12, MW-13) were installed at the field in order to fully investigate the occurrence and quality of groundwater from the base of fresh water (the top of the Pico Formation at approximately 500 feet below ground surface) to the shallowest occurrence of water (approximately 30 feet below ground surface).

Groundwater sampled from these wells was analyzed for the following constituents: pH, TPH, benzene, toluene, ethylbenzene, total xylenes, MTBE, TRPH, TDS, nitrate, nitrite, metals, and BOD_5. A comparison of the sample results from the pre-existing wells to the new deeper wells show generally consistent results. In most cases, the results for the new deeper wells were within the ranges found at the pre-existing wells. All analytes were below the state MCL for Drinking Water Standards, with the exception of arsenic, which is likely due to naturally occurring arsenic found in soil and rock formations as described previously in this section.

The following analytes showed no major changes in concentration:

- pH
- Benzene, toluene, ethylbenzene, total xylenes, MTBE
- TRPH
- Nitrate and nitrite
- Cobalt
- Lead
- Barium
- Copper, below detection in most cases, but had similar ranges for those analytes that showed above detection.

The following analytes showed ranges that were greater in the new deep wells in comparison to pre-existing wells; all ranges were below the state MCL:

- TDS in the shallow wells showed a concentration range of 510 to 2,500 mg/L vs. 1,400 to 3,900 mg/L for deep wells.
- Zinc in the shallow wells showed a concentration range of 18 to 120 µg/L vs. 20 to 280 µg/L for deep wells.
- BOD in the shallow wells showed a concentration range of 9.91 to 43.4 mg/L vs. 11.5 to 92.6 mg/L for deep wells.

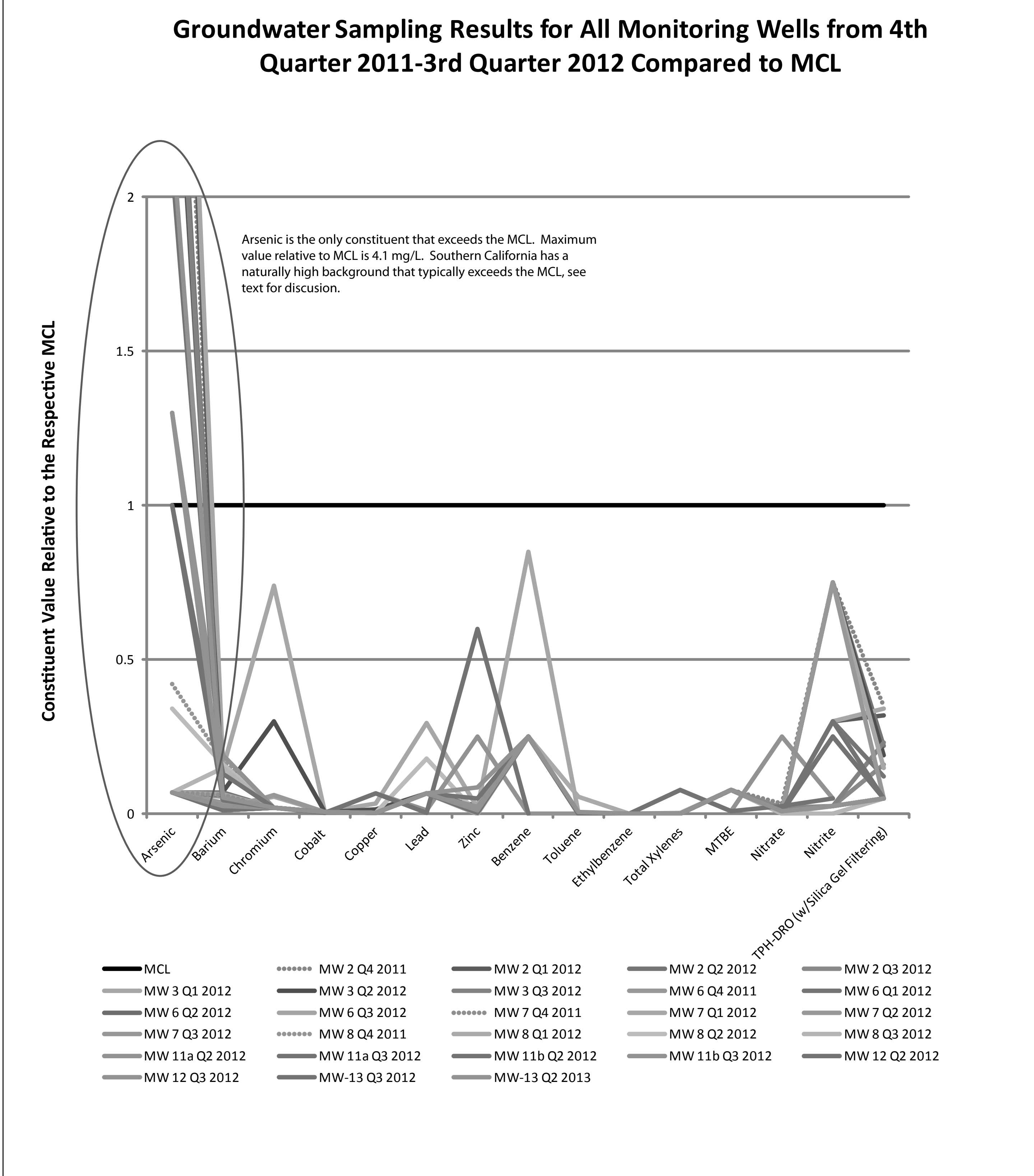


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Figure 4-5

Comparison of Baseline to Post-Hydraulic Fracturing Operation Water Quality

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The following chemical concentrations in the new deep wells were lower than the concentration in the pre-existing wells. As a note, all ranges were below the state MCL with the exception of arsenic, which is likely due to naturally occurring arsenic found in soil and rock formations:

- Arsenic in the shallow wells showed a concentration range of up to 37 µg/L vs. 21 µg/L for deep wells.
- TPH DRO in the shallow wells showed a concentration range of up to 2.1 mg/L vs. 0.77 mg/L for deep wells.
- Chromium in the shallow wells showed a concentration rage of up to 32 mg/L vs. 12 mg/L for deep wells.

In summary, the new wells, intended to investigate the deepest zones of fresh water beneath the Baldwin Hills, had similar groundwater quality results compared to the pre-existing wells that have been sampled on a quarterly basis prior to high-volume hydraulic fracturing. Accordingly, the new wells do not show a detectable environmental effect of high-volume hydraulic fracturing. The values are within the ranges detected over the course of the monitoring (Cardno ENTRIX 2012).

4.2.5 Surface Water

No perennial or intermittent streams as defined by the USGS are present at the Inglewood Oil Field (County of Los Angeles 2008), although Ballona Creek lies to the north and west. Surface runoff occurs primarily as sheet flow across the land surface, eventually flowing into ephemeral gullies and drainage ditches to six surface water detention basins. Runoff from these basins is discharged to the Los Angeles County stormwater system near the boundaries of the field in accordance with protections, sampling and analysis, and monitoring overseen by the LARWQCB (NPDES No. CA0057827). No discharge of surface water occurred during hydraulic fracturing operations; thus, there was no effect on surface waters draining from the oil field.

4.2.6 Sources of Drinking Water to the Local Community

West Basin Water District (District) provides water to the City of Inglewood, Culver City, and the unincorporated communities of South Ladera Heights, Lennox, Athens, Howard and Ross-Sexon, View Park, Windsor Hills, and others near the Inglewood Oil Field, either itself or through sale of water to retail service provides such as California American Water Company, California Water Service Company, and Golden State Water Company, among others.

Approximately 66 percent of the District's water supply is imported from either the Colorado River or from the San Joaquin Delta in Northern California. Approximately 20 percent of the District's supply is from groundwater; however, the nearest active supply well is 1.5 miles from the Inglewood Oil Field, and most of the groundwater supply is from significantly further than 1.5 miles from the Inglewood Oil Field (USGS 2003, West Basin Municipal Water District 2011).

Among the other communities in the vicinity of the Baldwin Hills, the City of Culver City relies on imported sources of water and does not have a groundwater supply (Culver City 2010). The City of Inglewood has four groundwater supply wells in the West Basin, all of which are greater than 1.5 miles from the oil field. None of these wells serve the community around the field (City of Inglewood 2010). Golden State Water Company has 13 wells within the West Basin groundwater basin with the closest one located 1.5 miles from the Inglewood Oil Field (Golden

State Water Company 2011). California American Water Company which services the Baldwin Hills provides water purchased from West Basin Water District as well as groundwater pumped from the Central Basin (California American Water 2011b).

All of the water service providers to the communities surrounding the Baldwin Hills must test their water at least 4 times a year and report the results to the water users. The constituents that are tested include, but are not limited to, the following:

- Turbidity
- Aluminum
- Arsenic
- Fluoride
- Nitrate
- Gross Alpha Activity
- Gross Beta Activity
- Uranium
- Color
- Chloride
- Odor
- Specific Conductance
- Sulfate
- Total Dissolved Solids
- N-nitrosodimethylamine
- Alkalinity
- Calcium
- Hardness
- Magnesium
- pH
- Potassium
- Sodium
- Total Coliform Bacteria
- Bromate
- Chlorine Haloacetic acids
- TTHMs
- Copper

A review of the 2011 Annual Water Quality Reports for the local community (which includes testing in the 4th quarter following the high-volume hydraulic fracture of VIC1-330), indicate that the community receives water that meets USEPA and California drinking water standards. The continued monitoring four times per year ensures that the water supply will continue to meet these standards (California American Water 2011a, Golden State Water Company 2012). The data for 2012 has not yet been posted as of the time of this study, and is expected to be posted in January 2013.

There are no domestic or industrial water supply wells located within the active surface oil field boundary. Potable water aquifers nearest to the Baldwin Hills include the Silverado Aquifer, which is located along the north-northwest boundary of the Baldwin Hills and extends to depths between 200 and 450 feet below ground surface, and several aquifers to the east of the Field including (in descending order) the Exposition, Gage, Lynwood, Silverado, and Sunnyside aquifers that extend to depths of approximately 800 feet. These aquifers are underlain by the non-water-bearing Pico Formation (see Figure 4-3C, DWR 1961). As described in Section 4.2.1, the USGS (2003), DWR (1961) and this study have determined that because the Baldwin Hills are uplifted, the formations do not allow groundwater to flow in to or out of the Baldwin Hills; it is isolated from the remainder of the Los Angeles groundwater basin and are considered to form a complete barrier to groundwater movement. The groundwater monitoring and analysis has shown that hydraulic fracturing did not have a discernible effect on groundwater quality beneath the Baldwin Hills; the isolated nature of this groundwater further indicates that there would be no effect on the groundwater in surrounding aquifers of the Los Angeles Basin.

In summary, over much of the Baldwin Hills there is limited or no groundwater within the freshwater interval above the Pico Formation, and where groundwater does occur, it is not connected to the aquifers of the Los Angeles basin. The groundwater is not used for water supply of any type, nor is it present in sufficient quantity to provide a water supply. The local community receives most of its water from sources in northern California (the Delta) or the Colorado River. Therefore, activities associated with oil and gas development in the Baldwin Hills do not affect the community's drinking water supply. The water supplied to the local community (as well as any community in California) must be sampled on a quarterly basis, with the results reported to the community.

4.2.7 Water Supply Concerns Related to Shale Gas Development Elsewhere in the United States

National Issue

There have been several studies published in 2011 and 2012 that examine the potential effects of hydraulic fracturing and shale gas development on private water wells in their respective study areas. Specific concerns with regard to groundwater contamination include: risk of migration and contamination from fracturing fluids; and, risk of migration and contamination from gas, oil, or other compounds (e.g., arsenic, methane).

2011 Duke Study: Methane Contamination of Drinking Water

The first Duke University study was conducted in response to the widespread public concern in Pennsylvania about drinking water contamination from drilling and fracturing, and lack of scientific evidence as to whether these activities posed an actual risk. The study described itself as the first scientific review of water contamination near hydraulic fracturing operations. The study, which collected and analyzed samples from 68 drinking water wells in the Marcellus and Utica formations in Pennsylvania and New York, aimed to evaluate the potential impacts of natural gas drilling and hydraulic fracturing on shallow groundwater quality by comparing areas with active drilling and fracturing to areas that are not currently being drilled. This study found that methane contamination in private drinking wells systematically occurred in areas where hydraulic fracturing of shale gas takes place (Osborn et al. 2011).

The study also indicated that methane was detected in 85 percent of the drinking water wells across the region, regardless of gas industry operations, thus demonstrating a regional background in this area of natural gas. The concentration of methane in water collected from the drinking water wells in areas with active natural gas drilling and extraction were approximately 17 times higher on average than those further away. Average and maximum methane concentrations were found to be higher in wells located within approximately 1,000 meters (3,280 feet) of active drilling sites. Although methane gas is known to occur naturally in shallow groundwater aquifers in both of these regions, the testing determined that the gas found in the wells was consistent with methane gas that originated at depths associated with the reservoirs that were drilled. However, although the testing showed elevated methane levels in the wells, isotopic analyses conducted on the same wells did not find any indication that hydraulic fracturing fluids or saline produced water had polluted the groundwater aquifers (Osborn et al. 2011).

Critiques of the study cite the lack of baseline data. While the study finds higher methane concentrations near active wells and concludes that these increases are the result of hydraulic fracturing operations, it does not compare its data to wells sampled prior to the occurrence of hydraulic fracturing. Other criticisms have asserted that the data set is not large enough to draw any definitive conclusions, and that the results are likely to vary regionally. Furthermore, critiques point to areas without drilling where methane was detected in wells, suggesting that claims that hydraulic fracturing caused methane contamination are not scientifically supportable (Bauers 2011).

2012 Duke Study: Natural Migration of Brines

Researchers at Duke University published a second study in response to continued concerns related to reports of potential drinking water contamination related to hydraulic fracturing in Pennsylvania. The study aimed to examine hydraulic conductivity between shale gas formations and the shallower drinking water aquifers in Pennsylvania.

The study analyzed the chemical content of 426 groundwater samples collected from six counties in Northeastern Pennsylvania that did not have links to drilling activities. The study then compared the salts present in the samples to the salts present in brine water from the Marcellus Shale. For some samples, they found that the salts in the groundwater had the same chemical composition as the salts in the Marcellus Shale brine, suggesting that there are naturally occurring hydrogeological pathways in the Marcellus shale that could allow migration from the shale to shallower aquifers. The authors report that there is no link between the salinity of the samples and proximity to Marcellus Shale gas wells, stating that "it is unlikely that hydraulic fracturing for shale gas caused this salinization and that it is instead a naturally occurring phenomenon that occurs over longer timescales." The report speculates that "these areas could be at greater risk of contamination from shale gas development because of a preexisting network of cross-formational pathways that has enhanced hydraulic connectivity to deeper geological formations" (Warner et al. 2012).

2011 Pennsylvania Methane Study

Cabot Oil and Gas Corporation (Cabot) conducted baseline "pre-drill" groundwater sampling and analysis throughout Susquehanna County, PA for various water quality parameters, including dissolved gases and other water quality parameters related to drinking water standards. The baseline analysis was performed in advance of proposed drilling in accordance with Pennsylvania regulations or anticipated regulations. The study concluded that (1) there is consistent evidence of elevated methane in shallow groundwater, (2) concentrations of methane seem to correlate with surface topography (i.e. more methane was found in wells that were in lowland valleys than on hilltops), (3) there was no relationship between methane concentrations in non-productions areas versus historical gas production areas (i.e. higher methane concentrations were not related to gas well drilling) (Molofsky et al. 2011).

In addition, a smaller subset of wells was sampled to conduct an isotope analysis, which helps distinguish between sources of natural gas. This portion of the study was initiated by the Pennsylvania Department of Environmental Protection (DEP) and Cabot. The study concluded that the methane is naturally occurring, unrelated to gas drilling, and not from the shale gas-producing Marcellus shale (Molofsky et al. 2011).

Finally, the report criticized the 2011 Duke study of methane contamination, arguing against the conclusion by the Duke researchers that the thermogenic methane identified in their water samples was consistent with the Marcellus shale. Instead, the isotopic fingerprints of the Duke samples, and other hydrogeological evidence, suggested that the methane found may have been from shallower sources rather than the deeper Marcellus formation and are not related to hydraulic fracturing activities (Molofsky et al. 2011).

2012 Dimock Study

Dimock, Pennsylvania was portrayed in the 2010 movie "Gasland," and included interviews with residents who feared their water was contaminated by natural gas drilling. In early January 2012, USEPA responded to complaints of drinking water quality in Dimock. Residents complained of cloudy, foul smelling water since 2009 after Cabot Oil & Gas Corporation began hydraulic fracturing operations to extract natural gas from reserves near the Marcellus Shale. The USEPA sampled waters from 64 homes in Dimock and concluded the set of samples did not indicate levels of contaminants that would foster further action by USEPA. The USEPA released the final data set on May 11, 2012, of 59 homes. Since USEPA sampling began, contaminants were found in some wells, but USEPA stated the levels of contamination in the wells were considered safe and did not pose a threat to human health. The USEPA also resampled four wells where previous data showed contamination. At one of those wells, USEPA found elevated levels of manganese (a naturally-occurring substance) in untreated well water, but the two homes serviced by that well had water treatment systems the reduced the level of manganese to sage levels. None of the other wells contained levels of contaminants that would require action. The USEPA did find one well containing hits for methane, but USEPA declined to verify the source of pollution, as methane is documented to be a naturally occurring gas in the surrounding area. USEPA has released all sampling results to residents in Dimock and has no further plans to conduct additional sampling (USEPA 2012b). Representatives for Cabot have publicly contended the contaminants found in some of the wells are likely from background levels or other activities unrelated to hydraulic fracturing activities (Gardner 2012).

2008 Bainbridge Township, Ohio Study

In December 2007, the Ohio Department of Natural Resource, Division of Mineral Resources Management (DMRM) initiated an investigation after there was an explosion at a house. Responders quickly recognized that natural gas was entering homes through water wells; either unvented water wells located in basements, abandoned and unplugged water wells in basements, or wells with indoor well pumps. The Ohio Valley Energy Systems Corp, which had recently completed a nearby oil and gas well, English No. 1, assumed responsibility for the natural gas contamination and resulting explosion.

Further investigation by DMRM concluded that three factors were likely to have contributed to the gas invasion of the shallow aquifers: (1) inadequate cementing of the production casing around that well, (2) proceeding with hydraulic fracturing without addressing the casing deficiencies, and (3) the month long period after hydraulic fracturing during which the annular space between the surface and the production casing was shut in, confining high-pressure gas in the restricted space. The over-pressurized condition cause the migration of natural gas from the well annulus into the natural fractures in the bedrock located below the base of the cemented surface casing. It is believed that the natural gas traveled vertically through the fractures into overlying aquifers and into local water wells (ODNR 2008).

2011 Pavilion Study

The USEPA released a draft report in December 2011 examining the potential of a link between groundwater contamination in Pavillion, Wyoming and local hydraulic fracturing operations. The USEPA's draft report found that groundwater samples taken from two deep test wells contained benzene and at least 10 other compounds known to be used in hydraulic fracturing fluid. The draft report theorized that the fluids seeped up from improperly sealed gas wells. Several months after the USEPA issued its draft report, reviews by both critics and proponents of hydraulic fracturing provided additional expert opinion on the draft. The critical reviews contend that the data collection processes were faulty and accordingly no valid conclusions could be drawn from USEPA's study. Notable criticisms of USEPA's draft report are as follows:

- The pollution detected by USEPA that was linked to hydraulic fracturing was found in deep water monitoring wells, not the shallower monitoring wells that are more comparable to the drinking water supply wells. The link between pollution in deep monitoring wells and shallow water wells is uncertain.
- Contamination in shallow monitoring wells was strongly linked to contamination from waste disposal pits, rather than migration of deeper sources of contamination.
- USEPA's monitoring wells were drilled directly into gas bearing zones; approximately 200 to 275 meters bgs (656 to 902 feet); therefore, reviewers suggest that it is not unusual that elevated levels of methane, hydrocarbons, and benzene were detected (Petroleum Association of Wyoming 2011). Along the same line, methane is naturally occurring near the surface of the Wind River Formation and many residents recall the presence of methane in well water prior to the occurrence of energy production activities in Pavillion (EnCana Oil & Gas Inc. 2009)
- To the extent that drilling chemicals were detected in deep monitoring wells, USEPA acknowledges the possibility of poor wellbore design and integrity, resulting in vertical and lateral movement of contaminants to surrounding groundwater. The study stated that only two gas production wells in the Wind River Formation have surface casings that extend below the depth of domestic wells. Shallow surface casings in conjunction with little or no cement or sporadic bonding of production casings can facilitate upward gas and fluid migration. In addition, poorly sealed domestic water wells are a known concern in Pavillion and an improper seal can create a migration pathway for gas and fluids into domestic wells.

Another subsequent study of USEPA's draft report commissioned by NRDC, the Wyoming Outdoor Council, Sierra Club and the Oil and Gas Accountability Project largely supported the agency's findings. The report, by Nevada-based hydrologic consultant Tom Myers, was careful to state that more testing is needed, though he said USEPA's preliminary conclusion that hydraulic fracturing had polluted the area's groundwater was sound. Myers said hydraulic fracturing fluids could move up a number of ways in the region — from compromised gas wells, past thin layers of sandstone, or through out-of-formation rock fissures. The natural gas wells in the area, he stated, often lack metal casings or cement, allowing natural gas and fluids to travel up into groundwater. The USEPA, Myers said, also found a number of compounds during testing that aren't found naturally, including isopropanol and diethylene glycol. "*The [US]EPA is correct in its conclusion that there is no acceptable alternative explanation and the most likely source of these contaminants is fracking fluid*," Myers wrote. Myers also disputed that cement and drilling mud contaminated water samples, stating that neither could raise the pH level to the range found — between 11.2 and 12.0. Myers recommended that USEPA should continue collecting data, including from new and deeper monitoring wells, to try to replicate and verify its findings (Myers 2012b).

While USEPA's draft report identified potential links between hydraulic fracturing and contamination in the Pavillion water wells, the report remains a draft and USEPA is continuing its review. At a hearing before the House Subcommittee on Energy and Environment, in February 2012, USEPA Region 8 administrator Jim Martin stated the following in response to mischaracterizations of the draft report: "*We make clear that the causal link [of water contamination] to hydraulic fracturing has not been demonstrated conclusively,*" adding that USEPA's draft report "*should not be assumed to apply to fracturing in other geologic settings*" (Martin 2012).

In March 2012, USEPA agreed that additional testing was needed in the Pavillion before a final report could be issued. The USEPA in conjunction with the State of Wyoming is currently conducting further sampling of water wells in the area. In a joint statement, USEPA Administrator Lisa Jackson, Wyoming Governor Matt Mead, and the Northern Arapaho and Eastern Shoshone Tribes said: "*The USEPA, the State of Wyoming, and the Tribes recognize that further sampling of the deep monitoring wells drilled for the Agency's groundwater study is important to clarify questions about the initial monitoring results*" (USEPA 2012c).

2012 Myers Model Study of the Marcellus Shale

Tom Myers, a Nevada-based hydrologic consultant, published a study in spring 2012, which uses a model to characterize the risks associated with contaminants travelling through natural vertical pathways from fractured shale to shallower drinking water aquifers. The study analyzes two potential hydrogeological pathways – advective transport through bedrock and preferential flow through fractures. Myers assigned various factors to model contaminant flow including groundwater flow, conductivity of the substrate, and changes in conductivity of the substrates based on regional shale hydrogeology, high density fracturing, and faulting; and high-volume injection.

The study acknowledges that the model simplifies a complex underground system, but the results suggest that a combination of the factors described above could decrease transport times from the Marcellus Shale to shallower aquifers from geological times scales to only tens of years, and that preferential flow through natural and hydraulic fracturing induced fractures could further reduce transport times to as little as just a few years (Myers 2012a). However, the study is a modeling exercise that is theoretical in nature and specific to contaminant transport in the Marcellus shale. Following publication of this study, Syracuse University hydrogeology Dr. Don Siegel released a critique of its assumptions and conclusions. Myers developed "an implausible model" that produced "completely wrong results," Siegel wrote. According to Siegel, the Myers model is based on mistaken assumptions about the kind of rock that lies above the Marcellus Shale, the way groundwater moves through sedimentary basins, and the length of the fissures created by hydraulic fracturing (Siegel 2012).

Relevance to Inglewood Oil Field

Of the studies noted above, USEPA's ongoing study of Pavillion, Wyoming has received the most media attention nationally and continues to be the subject of varying interpretations. In evaluating the draft report it is important to note that hydraulic fracturing operations and geology at the Pavillion Natural Gas Field are distinctly different from those under consideration at the Inglewood Oil Field. For comparison purposes, the drinking water aquifer sits directly atop the production zone at the Pavillion Natural Gas Field and high-volume hydraulic fracturing is used to extract gas from as shallow as 372 meters (1,220 feet) bgs, in close range to the domestic water wells that are screened as deep as 244 meters (800 feet) bgs. Consequently, there is a negligible separation between the hydraulic fracturing operations and groundwater. In contrast, the high-volume hydraulic fracturing jobs PXP conducted occurred over 2,500 meters (8,202 feet) bgs to retrieve shale oil, while the perched water formation is located approximately 120 meters (393 feet) bgs, a separation of over one mile.

Furthermore, as noted in Section 4.2.1, there is no groundwater that can sustain a water supply beneath the Baldwin Hills. The supply to the local area is primarily from sources outside of Los Angeles, whereas in Pavillion groundwater is integral to the community's water supply. The water supply in Los Angeles is subject to quarterly testing and public reporting. Due to uplift in the Baldwin Hills and associated folding and faulting, water tables are sporadic and shallow throughout the production zone on the oil field itself and there is no sustainable groundwater resources located within perimeters of the oil field. In groundwater models of freshwater flow in the Los Angeles Basin aquifer systems (USGS 2003), the Baldwin Hills is modeled as a "no flow" zone; that is, since the sediments beneath the Baldwin Hills are disconnected from the regional aquifers, groundwater flow is discontinuous across the Baldwin Hills (see Figures 4-1 and 4-3C).

4.3 Containment of Hydraulic Fractures to the Desired Zone

A significant amount of discussion has taken place about the vertical growth of hydraulic fractures, particularly in gas shales, tight sands, and shallow reservoirs in regards to whether these hydraulic fractures can create pathways for the fracturing fluids or hydrocarbons to migrate upward and contaminate groundwater supplies.

The vertical extent that a created fracture can propagate is controlled by the upper confining zone or formation, and the volume, rate, and pressure of the fluid that is pumped. The confining zone will limit the vertical growth of a fracture because it either possesses sufficient strength or elasticity to contain the pressure of the injected fluids or an insufficient volume of fluid has been pumped. This is important to note because the greater the distance between the fractured formation and the groundwater or water-bearing zones, the more likely it is that multiple formations will possess the qualities necessary to impede the growth of hydraulic fractures.

Microseismic and micro-deformation mapping has been conducted on thousands of hydraulic fracturing jobs nationwide (Fisher and Warpinski 2011) and indicate that the growth of fractures vertically is relatively well-contained. Figure 4-6 is taken from Fisher and Warpinski (2011) and depicts the depth and vertical height affected by hydraulic fracture jobs conducted in the Barnett Shale of Texas (inclined multi-colored lines), compared to the depth of water (blue horizontal line at top of chart). This relationship was validated during observations of microseismic results at the Inglewood Oil Field (see Figure 3-11).

Fracture lengths of a typical hydraulic fracture operation can sometimes exceed 1,000 feet when contained within a relatively homogenous layer, but fracture heights, because of the layered geological environment and other physical parameters are typically much smaller, usually measured in ten-foot or hundred-foot intervals (Warpinski et al. 2011).

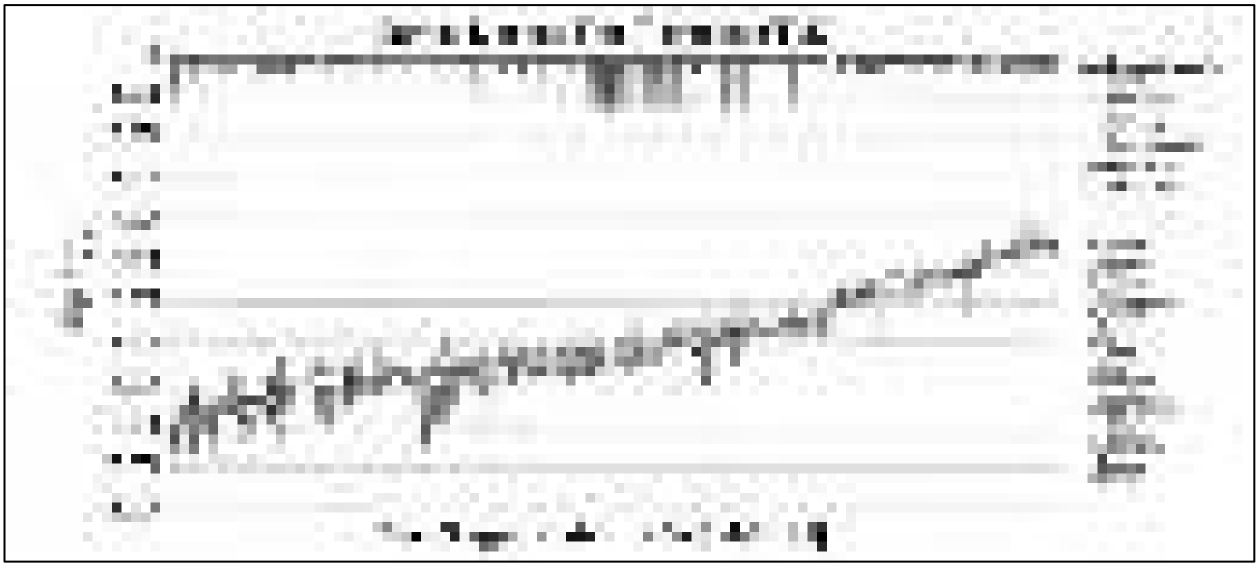


Figure 4-6 Barnett Mapped Frac Treatments/TVD

Similar results have been found by Fisher and Warpinski (2011) in their study of the real fracture growth data mapped during thousands of fracturing treatments in tight sands and shales. They supplemented their data with an in-depth discussion of fracture-growth limited mechanisms augmented by mine back tests and other studies. They also note that fractures and fracture networks tend to be complex; the complexity tends to shorten the network as the energy dissipates.

At the Inglewood Oil Field, the measured distribution of fractures caused by the hydraulic fracture completions at VIC1-330 and VIC1-635 were less than 1,000 feet horizontally from the well, and were almost entirely within the target zone with limited vertical fracture growth (less than 250 feet). Fractures grew either horizontally from the well or at angles less than 20 degrees depending on the local dip angle of the geological formations. The high-volume hydraulic fracture completions were conducted between 8,000 and 9,000 feet below the ground surface and microseismic analysis of the operations indicate that fractures did not form at shallower depths than approximately 8,000 feet below the ground surface (see Section 3.2.2). By comparison, the deepest groundwater encountered that had relatively low salinity was at a depth of 500 feet below the ground surface, corresponding to the base of fresh water beneath the Inglewood Oil Field.

During hydraulic fracturing, the pressure applied to the rock by the water/sand/additive mixture exceeds the fracture strength of the rock, and portions of the rock fractures. As measured by the microseismic data in Section 3.2.2, the induced fractures follow the bedding planes at approximately 20-degree angles. Halliburton (2012) also models the distribution of proppant applied to the fractures in the target zone. Based on this model, all of the proppant stayed within the target zone of the Nodular Shale. The minor fractures indicated by the microseismic data that occurred outside the Nodular Shale were in the underlying, oil-bearing Sentous Formation. These fractures did not receive proppant, and as such they sealed based on the overburden pressure.

4.4 Well Integrity

The proper construction of active oil wells and the condition of idle, plugged, and abandoned wells ensures that protections to fresh water-bearing zones are intact. As described in Section 4.2.1, the water-bearing zones above the base of fresh water beneath the Baldwin Hills do not have sufficient yield to support a water supply, and the local community receives their water supply primarily from sources that are distant from the Los Angeles Basin. However, well integrity was still investigated as part of this study at the scale of the wells subject to hydraulic fracturing, as well as at the scale of the entire oil field.

The term "well integrity" refers to the containment of hydrocarbons within a well from the producing formation all the way to the surface. The rock formations that lie between the hydrocarbon producing formations and the groundwater have isolated the groundwater over millions of years. The well construction process uses a combination of steel casing, cement sheaths, and other mechanical isolation devices to prevent the migration and transport of fluids between these subsurface layers. These construction and engineering controls provide multiple layers of groundwater protection throughout the life of the well. The wells within injection zones at Inglewood Oil Field are constructed in accordance with API guidelines, ensuring that all Federal and State regulations are met and groundwater is protected (API 2009).DOGGR well integrity regulations require a facility to keep records of the size, weight, grade, and condition of all casings and any equipment attached to the casing, pursuant to California Code of Regulations (CCR), Chapter 4, Article 3, §1724.

To supplement more broadly applicable statutory and regulatory requirements, the State Oil and Gas Supervisor may establish Field Rules for any oil and gas pool or zone in a field when sufficient geologic and engineering data are available from previous drilling operations, pursuant to CCR Title 14, Division 2, Chapter 4, § 1722 (k). Each Field Rule is specific to a field, and in many cases, specific to Areas and Zones or Pools within a field. DOGGR has established Field Rules for those fields where geologic and engineering information is available to accurately describe subsurface conditions. These Field Rules identify downhole conditions and well construction information that oil and gas operators should consider when drilling and completing onshore oil and gas wells. Field Rules have been established for the Inglewood Oil Field in two areas – west of the Newport-Inglewood fault and east of the Newport-Inglewood fault (DOGGR 2007).

With regard to abandoned wells, Term 10 of the Settlement Agreement requires PXP to install a 150-foot cement plug at the surface of the well, which exceeds DOGGR standards 6-fold (DOGGR requires only installation of a 25-foot plug). This supplemental requirement provides enhanced protection of any surface resources.

Based on review of records maintained by PXP, the active wells at the Inglewood Oil Field meet modern well construction and casing standards, which protect against releases to the environment, pursuant to State Regulations and Field Rules. Idle wells are tested annually and reports are submitted to DOGGR, in accordance with CCR, Chapter 4, Article 3, §1723.9.

Prior to commencing injection operations, each injection well must pass pressure tests to confirm the integrity of the casing. A Mechanical Integrity Test (MIT) must be performed on all injection wells to verify that injected fluid is confined to the approved zone(s). California regulations mandate that "data shall be maintained to show performance of the project and to establish that no damage to health, life, property, or natural resources is occurring by reason of the project. Injection shall be stopped if there is evidence of such damage, or loss of hydrocarbons. All project data shall be available for periodic inspection by Division personnel" (CCR 1724.10 (h)).

Active injection wells at the Inglewood Oil Field are surveyed annually (and pressure tested after each well work) per DOGGR requirements pursuant to CCR, Chapter 4, Article 3, §1724.10(j)3. In addition, PXP monitors active injection wells weekly for injection rates and pressures (what also indicates the integrity of the wellbore and confinement of fluids to the injection zone) and reports to DOGGR on a monthly basis, pursuant to CCR Chapter 4, Article 3, §1724.10(c).

PXP also measures the pressure of the annulus after they idle production or injection at a well and reports this data to DOGGR, as required by Chapter 4, Article 3, §1724.1. Each idle well (production or injection) is subject to DOGGR Idle Wells Testing program according to California Code of Regulations. The testing is done in two parts. Initially it is determined if wellbore fluids (i.e. formation fluids) are above or below the designated base of fresh water. If the fluids are below, then no further testing is required until the next testing cycle. If the fluid in the wellbore is at or above the designated base of fresh water, thus creating a potential for migrating into the designated fresh water formation, then the next part of testing takes place. The integrity of the casing (steel pipe and cement) at this time is evaluated by one of the following methods: running static temperature and spinner surveys, pressure testing the casing, or a nitrogen fluid level depression tests. Both fluid level in the wellbore location determination, and subsequent (if needed) integrity testing are subject to DOGGR witnessing. Any problems determined during an annual Idle Wells Testing are addressed by either repairing or abandoning the well.

Additional well integrity monitoring is provided through PXP's active production well monitoring. In accordance with Public Resources Code Division 3, Article 1, Section 3227, PXP provides monthly production reports that indicate the amount of oil/gas produced from each well, composition of produced water (e.g., salinity), amount of injection fluid, and any other information requested by the Division. While reports are only produced on a monthly basis, PXP monitors active production wells daily for oil and water production rates and pump behavior. Although PXP reports that this is not intended to be a well integrity monitoring program, PXP notes that this monitoring allows them to quickly identify, isolate, and correct any potential problems.

Well integrity is further monitored prior to and during each stage of the hydraulic fracturing operations. The well casing of the subject well is tested to ensure integrity prior to injection of fracturing fluids (Halliburton 2012). This is accomplished by pressure testing the well up to 70 percent of the strength of the casing. Offset wells, production wells, and injection wells are also tested for proper zonal isolation (i.e., annular cement) prior to any hydraulic fracturing operations. Halliburton's post-job reports for hydraulic fracturing operations indicate that all measurements of well integrity conducted for this study show that there were no losses in pressure. Furthermore, as shown in the microseismic results (see Section 3.6), none of the fractures encroach on nearby wells. The applied energy of the hydraulic fracturing rapidly decreases away from the completed well, and as such, surrounding wells would not be adversely affected by the operation.

Prior to issuing a permit for any new injection well, or converting an existing production well to an injection well, DOGGR conducts an Area of Review (AOR) as required by the Underground

Injection Control Program (see Section 5.4.1 for greater discussion of this program). The AOR investigates the condition of every well within one-quarter mile radius of the proposed new injection well. The AOR includes all active, idle, plugged, and abandoned wells and determines the casing and cement intervals. The purpose of the review is to ensure that all wells in the area are completed or abandoned in such a way as to contain injected fluids within the zones that are approved for injection and isolate freshwater zones from the injected fluids.

4.5 Slope Stability, Subsidence, Ground Movement, Seismicity

4.5.1 Slope Stability

Slope stability is a primary geologic concern in the Baldwin Hills. The California Department of Conservation, Division of Mines and Geology, has previously reported on slope stability and geological issues of the Baldwin Hills (CDMG 1982). The purpose of the program was to identify the nature and cause of slope failures across the state, and to provide the information to local governments within whose jurisdictions the failures occurred so that they can plan action to mitigate the problems. The 1982 study was solely focused on the Baldwin Hills and included investigations starting in 1969. The study included detailed mapping of areas with slope instability in the Baldwin Hills, investigation into the causes of the failures, and recommended mitigation.

The study notes widespread damage from slope failures caused by rains in 1969, 1978, and 1980, and less widespread damage in other years. The study concludes that there are two reasons why slope stability is a substantial problem in the Baldwin Hills:

- The terrain that has been developed consists mostly of steep natural slopes underlain by soft sedimentary rocks. This combination will lead to slope instabilities. The result is that graded and natural slopes with slope angles up to 45 degrees or steeper occur without proper drainage devices and retaining walls.
- Much of the Baldwin Hills were developed prior to the enactment of strict grading codes by local government, and as such lack adequate protections. These protections include lower slope angles, requirements for compaction of fills, and structural requirements.

The study notes that the Inglewood Formation is susceptible to slope instability because the surficial soils developed on the formation are clay-rich, while the Culver Sands are particularly susceptible to erosion. The study also notes the presence of ancient, apparently large, landslides. Most of the mapped slope failures damaged more than one property.

The study also notes that in the three most densely developed portions of the Baldwin Hills, approximately 21 percent of the properties have been damaged by rainfall-induced slope failures. As described in the CDMG report, approximately 93 percent of the residential properties have the potential for at least minor damage from slopes failure during or after large storms in the future (CDMG 1982).

Monitoring for vibration and subsidence did not detect a change due to hydraulic fracturing. As such, hydraulic fracturing would not affect surface slope stability.

4.5.2 Subsidence

Subsidence is another geological concern in the Baldwin Hills. As described in the Baldwin Hills CSD EIR, prior to 1971 the maximum cumulative subsidence of any of the areas along the Newport-Inglewood fault zone was centered over the Inglewood Oil Field, where 67,000 acre-feet of oil, water, and sand had been withdrawn from shallow production horizons. Water injection into the shallow production horizons began in 1957 and as of 1971, effectively eliminated subsidence associated with oil and gas production (County of Los Angeles 2008). The Inglewood Oil Field has an ongoing program of annual subsidence monitoring that is reported in the framework of the CSD. To date, although minor subsidence has been detected, no changes in ground surface are attributed to oil and gas production activities (Fugro Consultants 2012). Measurements of subsidence before and after high-volume hydraulic fracturing did not detect a measurable change.

Subsidence has also been theorized to be one factor associated with the failure of the former 20-acre Baldwin Hills Reservoir in 1963. The north embankment of the Baldwin Hills Reservoir failed causing property damage and loss of life. One of the leading theories for the reservoir's failure is that it was undermined by seepage along a fault which was known prior to construction of the reservoir, and which is related to the active Inglewood fault system. The dam's failure has been attributed to different causes: oil-field subsidence (Castle and Yerkes 1969); tectonic faulting (Hudson and Scott 1965); water injection in the nearby oil field (Hamilton and Meehan 1971); and construction related factors (Wright 1987). An innovative design was intended to prevent tectonic subsidence and water injection from jeopardizing the reservoir. In a study of the reservoir failure, Wright (1987b) presents records that document that a field change to the design during construction undermined most of the features intended to accommodate the original design protections. As such, it has been theorized that the design changes also played a role in the dam's eventual collapse (Casagrande et al. 1972).

4.5.3 Monitoring of Ground Movement

The CSD requires an annual ground movement survey at the Inglewood Oil Field. Surveying for both vertical and horizontal ground movement is accomplished using satellite-based Global Positioning System (GPS) technology. Accumulated subsidence or uplift is measured using repeat pass Differentially Interferometric Synthetic Aperture Radar (inSAR) technology. The data are then evaluated to determine whether Inglewood Oil Field operations are related to any detected ground motions or subsidence. According to the ground movement survey covering the 2011/2012 monitoring period there is no correlation between measured elevation changes and field activities (Fugro Consultants 2012, Fugro NPA 2012, Psomas 2012). For this period, inSAR imagery was collected on January 15, 2012, and elevation data at each survey location was collected in February 2012. Note that hydraulic fracturing operations occurred in September 2011 and January 5-6, 2012, and were captured in this survey period. Fugro Consultants compiled a list of the Inglewood Oil Field production and injection wells within a 1,000-foot radius of each survey location and PXP supplied the annual production and injection volumes from all active wells across the field. The database included 456 production wells and 213 waterflood injection wells. Some stations recorded settlement where the injected fluid exceeded the produced volume, some monuments recorded elevation gains where the produced water volume exceeded injected volume, and others showed changes where no active wells are within a 1,000-foot radius. The majority of movements is less than the 0.05 foot measurement threshold and, therefore, at or less than the limit that can be detected (Fugro Consultants 2012, Fugro NPA 2012, Psomas 2012). None of the ground movement was attributable to high-volume hydraulic fracturing.

4.5.4 Vibration and Seismicity During Hydraulic Fracturing

PXP retained Matheson Mining Consultants, Inc. to conduct vibration and ground surface monitoring during the high-volume hydraulic fracturing operations at the VIC1-330 well on September 15 and 16; the VIC1-635 well on January 5 and 6; and at TVIC-221 and TVIC-3254 high-rate gravel pack operations on January 7, 9, 10, and 11.

Vibration records for the VIC1-330 and VIC1-635 wells were collected using four and eight seismographs, respectively, installed at different locations in relation to the high-volume hydraulic fracture operations. The TVIC-221 and TVIC-3254 wells are directly adjacent to one another; therefore, the same seismographs were used to monitor the high-rate gravel pack operations at these wells. Vibration records for these wells were collected using eight seismographs installed at different locations between 218 and 1,000 feet from the wells. Seismographs were placed near the subject wells. All of the seismographs were put in place early enough to allow the collection of 24 hours of baseline data prior to recording vibrations of the hydraulic fracturing operation and high-rate gravel pack operation. Each device was set to the lowest trigger level possible (0.005 in/sec) in order to detect all vibrations.

Seismic events (imperceptible to humans and below the limit which could cause any structural damage) caused by vehicles and other oil field activities at the surface were noted during the baseline period conducted prior to each hydraulic fracture event. These events were then compared to events recorded during both the pumping test and hydraulic fracturing time periods. Table 4-1 displays the highest level vibration recorded during the baseline period and each hydraulic fracture operation. Based on this comparison, Matheson Mining concluded that no seismic activity was produced by any of the high-volume hydraulic fracturing or high-rate gravel pack operations for which seismicity was recorded (Matheson Mining 2012a, b, c).

Table 4-1 Comparison of Vibration Levels Recorded During Baseline Monitoring and Hydraulic Fracturing Operations

Subject Well	Maximum Vibration Record (in/sec)	
	Baseline Monitoring	During Hydraulic Fracturing Operation
VIC1-330	0.0062	0.0119
VIC1-635	0.0075	0.0162
TVIC-221 and 3254	0.025	0.0194

Source: Matheson Mine Consultants 2012 a-c

4.5.5 Induced Seismicity and Additional Seismic Monitoring During Hydraulic Fracturing

Microseismicity was measured directly during and after the hydraulic fracturing. However, the public has expressed concern related to induced seismicity along the Newport-Inglewood Fault potentially resulting from hydraulic fracturing or water injection. This section addresses this topic using data from the Newport-Inglewood Fault across southern California, and measurements made at the field as part of the California Institute of Technology's monitoring program for the region.

The Newport-Inglewood Fault is discernible at the surface by the chain of low hills extending from Culver City (the Baldwin Hills) to Signal Hill. According to the Southern California Earthquake Center (Petersen and Wesnousky 1994), the fault is not marked by a sharp zone, but instead is marked by a broad zone of seismicity centered on the fault trace. Faults of the Newport-Inglewood zone of deformation are predominantly defined in the subsurface from oil-well data and groundwater data. Petersen and Wesnousky (1994) evaluated all seismic events greater than Magnitude 2 in the Newport-Inglewood Fault zone, and determined that most epicenters are located at depths between 3.5 miles and 12 miles below ground surface (Petersen and Wesnousky 1994, Hauksson 1987).

In comparison, the waterflood operation at the Inglewood Oil Field extends to depths of up to 3,000 feet (0.57 mile) and the deepest hydraulic fracturing occurs at less than 10,000 feet depth (1.9 miles). The very small, not discernible, microseismic effects of fracturing are located 1.6 miles above the zone where most epicenters are located, and the waterflood is 2.9 miles above this zone. Based on distHance alone, there would be little or no relationship between the location of Inglewood Oil Field activities and the much deeper epicenters of most earthquakes along the Newport-Inglewood fault zone.

In addition to the seismic monitoring conducted by Matheson Mining Consultants, Inc., seismic data collected by the permanently installed California Institute of Technology (Cal-Tech) accelerometer (seismometer) at the CI.BHP Baldwin Hills location (adjacent to the PXP field office at 5640 South Fairfax Avenue, approximately 6,300 feet southeast of VIC1-635 and 7,806 feet southeast of VIC1-330, and 9,620 feet from the TVIC wells) was reviewed for the time periods before and during the hydraulic fracturing and high-rate gravel pack operations. The data collected from the seismograph during the VIC1-635 operation showed two minor spikes during the time period reviewed (the largest of which measured 0.0012 inch per second). Analysis of the data by Dr. Hauksson, a Senior Research Associate in Geophysics with the Cal-Tech Seismological Laboratory indicates that no seismic events above background levels (0.0003 to 0.0006 inch per second) were recorded. These spikes tend to occur randomly every two or three hours and could be related to local traffic. According to Dr. Hauksson, these spikes are common in urban areas and not considered significant (Matheson Mining Consultants, Inc. 2012a). No data above background levels was recorded on the Cal-Tech seismograph during the VIC1-330 operation (Matheson Mining Consultants, Inc. 2012b). The data collected from the seismograph during the TVIC high-rate gravel pack operations showed some spikes during the time period reviewed but no significant signals above the background levels. As with the VIC1-635 operation, analysis of the data by Dr. Hauksson indicates that the noise recorded on the seismograph during the time period of the hydraulic fracturing operation, even the spikes, did not exceed background levels (Matheson Mining Consultants, Inc. 2012a).

The utilization of these data is relevant in addressing public concerns about the potential for ground movement triggered through induced seismicity as a result of hydraulic fracturing and high-rate gravel pack operations at the Inglewood Oil Field. Based on an analysis of the data, all tests indicate that the hydraulic fracturing analyzed in this study did not induce seismic activity. Any microseismicity as a result of the hydraulic fracturing was imperceptible at the surface. In addition, any effects of oil field operations are much shallower than the zones typically associated with earthquake epicenters along the Newport-Inglewood Fault zone. The Baldwin Hills CSD includes provisions that address the effects of earthquakes on the field. These include construction provisions, and provisions to cease operations after large earthquakes and conduct inspections. For example, the Magnitude 6.4 Coalinga earthquake of 1983, in the San Joaquin Valley of California, caused damage to some oilfield facilities. Most of the damage was to surface facilities, with very minor subsurface damage. Fourteen of 1,725 active wells had some damage (Hughes et al. 1990). This event led to enhanced safety measures. The Baldwin Hills CSD requires an accelerometer on the field for purposes of monitoring seismic activity and triggering inspections.

4.5.6 Potential for Induced Seismicity at Other Areas in the United States

National Issue

Several earthquakes in Mahoning County, Ohio (an area that is not historically seismically active) prompted Ohio's Department of Natural Resources to shut down five deep underground injection wells in January 2012, due to concerns that wastewater injected into the wells under pressure triggered the earthquakes. Similarly, a 5.6-Magnitude earthquake shook Oklahoma in November 2011, following a series of smaller quakes over the preceding months that may also have been attributed to wastewater injection.

All agencies that have reviewed the question have determined that hydraulic fracturing itself is not the cause and is likely not capable of producing an earthquake event of any notable size. Seismologists at the U.S. Geologic Survey have found that hydraulic fracturing "itself probably does not put enough energy into the ground to trigger an earthquake" (USGS 2012). Review of the source studies for the articles found that many point to energy-related activities other than hydraulic fracturing (e.g., injection for waste disposal) as the source of induced seismicity.

As recently as June 2012, the National Research Council, a division of the National Academies of Science, released a report titled *Induced Seismicity Potential in Energy Technologies.* The report found that only one felt event in England had been confirmed and attributed to hydraulic fracturing globally. This case, caused by Cuadrilla in England in 2011, recorded two earthquakes (one Magnitude 2.3 and one Magnitude 1.5) that Cuadrilla believes was due to hydraulic fracturing. These are below a level that would be felt. The cause was thought to be injection of large volumes of sand and proppant.

Of the 35,000 shale gas wells that had been hydraulically fractured, only one case was suspected, but not confirmed, to be attributed to hydraulic fracturing connected to shale gas development. Two other cases connected to conventional oil and gas development were associated with, but never confirmed to stem from, the application of hydraulic fracturing technologies. The report found that, "the very low number of felt events relative to the large number of hydraulically fractured wells for shale gas is likely due to the short duration of injection of fluids and the limited fluid volumes in a small spatial area" (NRC 2012).

Seismic activity as a result of energy-related activities is not a new phenomenon. According to the USDOE Lawrence Berkeley National Laboratory, energy-related activities have been linked to isolated events of induced seismicity since the 1930s, which marks the start of large-scale fluid extraction (USDOE 2012). USDOE has found that hydraulic fracturing is known to cause slight tremors when fluid is injected into the ground under high pressure, but these are on the order of Magnitude -3 and -4 and are practically imperceptible.

Most of the USDOE research conducted to date however has pointed to the injection of fluids into deep wells for waste disposal as a cause of induced seismicity, as well as the use of reservoirs for water supplies, carbon sequestration, and geothermal energy operation. Injection into deep wells could cause seismic events capable of being felt if fluids migrate into neighboring rock formations. The deep, old rocks that surround injection wells have many faults that have reached equilibrium over hundreds of millions of years, but migrating fluid due to wastewater injection could disrupt this equilibrium and trigger shaking (de Pater and Baisch 2011).

Similarly, USGS studies have indicated that hydraulic fracturing did not cause increased seismicity in the midcontinent United States. As part of an effort to understand the potential impacts from U.S. energy production, the USGS has been investigating the recent increase in the number of earthquakes in the midcontinent United States with a Magnitude of three or greater on the Richter scale. Scientists looked carefully at regions where energy production activities have changed during recent years. The results of the studies suggest that hydraulic fracturing has not caused the increased frequency of earthquakes; however, in some instances, the increase in seismicity was linked to deep underground injection wells. The USGS indicates that it is still unclear whether the increased seismic activity is related to changes in production methodology or the increased rate of oil and gas production. For example, the USGS has previously reported that oil and gas extraction can cause earthquakes when removal of large quantities of oil, gas, or water changes underground stresses. The studies also note that not all underground injection causes earthquakes and that there have not been conclusive examples that underground injection may trigger large, major earthquakes even if located near a fault (Hayes 2012).

The recent National Resource Council study of induced seismicity (NRC 2012) finds that many factors are important in the relationship between human activity and induced seismicity: the depth, rate, and net volume of injected or extracted fluids, bottom-hole pressure, permeability of the relevant geologic layers, locations and properties of faults, and crustal stress conditions. Moreover, in a recent survey of earthquake activity and injection wells in Texas, the results suggested that injection rates, pressures, geological substrate permeability as well as fault location and underlying fault stress could influence the probability of fluid injection creating earthquakes (Frohlich 2012). In the siting of many injection wells, these factors are not well known. At the Inglewood Oil Field however the geological conditions are well known and there is a long history of successful waterflood operations.

Ohio Case Study

Although Mahoning County, Ohio, is historically not a seismically active area, nine low-Magnitude earthquakes were observed beginning in early 2011. Initial media coverage following the earthquakes pointed to hydraulic fracturing as the cause (Fountain 2011, Palmer 2012). Seismologists later plotted the quakes however and determined that their epicenters corresponded to Northstar 1, a 9,000-foot deep Class II injection well used to dispose of brine and wastes from natural gas hydraulic fracturing operations. The state shut down four deep disposal wells in January 2011, after a Magnitude-4 earthquake occurred. The Ohio Department of Natural Resources concluded that the earthquakes were caused by deep underground injection at Northstar 1, not the hydraulic fracturing operations, for the following reasons: injection operations began at Northstar 1 shortly before the first seismic events were recorded in the area; seismic events were clustered around the wellbore (the focal depths of the events were 4,000 feet laterally and 2,500 vertically from the well bore terminus); and there is evidence of fractures and permeable zones in the surrounding formation. Further modeling and analysis of the Northstar 1 well and the surrounding geology are required to establish a better understanding of what happened. Additional studies are underway by ODNR and cooperating agencies (ODNR 2012).

Texas Case Study

In 2011 and 2012, a retrospective survey was conducted near the Dallas-Fort Worth area in Texas. The study's intent was to review the relationship between detectable earthquakes and the characteristics of nearby injection wells. The study reviewed earthquake activity between November 2009 and September 2011 over a 70-km area of the Barnett Shale using temporary seismographs installed under the National Science Foundation's funded EarthScope U.S. Array program. It identified that there were a sizable number (67) of earthquakes Magnitudes 1.5 and higher that could be identified under the U.S. Array program. Only one-eighth of these were reported by the National Earthquake Information Center, however.

The earthquakes identified seemed to have varying relationships with the surrounding injection wells. The rates of injection wells nearest the strongest cluster of earthquakes typically exceeded 150,000 barrels of water per month. However, 90 percent of wells that had injection rates exceeding 150,000 barrels of water per month did not have related earthquakes. The study suggested that earthquakes may more likely be triggered if the injection reaches a critical rate, but that the rate could depend on localized geologic conditions. The geological substrate permeability as well as fault location and underlying fault stress could influence the probability of fluid injection triggering earthquakes (Frohlich 2012).

Oklahoma Case Study

In January 2011, the Oklahoma Geologic Service was contacted with reports of multiple earthquakes observed in the Garvin County area within a 24-hour period. Following the reports, the Oklahoma Geologic Service confirmed that in fact, over 50 earthquakes ranging in Magnitude 1.0 to 2.8 were recorded in this area. A review of activity in the area also confirmed that a hydraulic fracturing event had taken place that day at the nearby Eola field. This area of south-central Oklahoma has historically been seismically active; therefore, a network of seismic stations was installed in 1977 which allowed for the accurate reporting and determination of epicenters for this series of quakes. The Eola field is located in an area where several fault blocks are located between major faults (the Eola, Reagan, and Mill Creek faults). The Oklahoma Geologic Survey conducted its own study to determine if the reported earthquakes were in fact induced by the hydraulic fracturing that had taken place on the field. The study involved a series of model simulations and statistical analyses using the data records by the seismic monitors as well as data collected from the field operator regarding the hydraulic fracturing event itself. The Oklahoma Geologic Service found that there was a clear correlation between the hydraulic fracturing event and the observed seismicity, and that all of the epicenters of the seismic events were within 5 km (3.1 miles) of the Eola field and that some of the earthquakes occurred at similar depths as the reservoirs which were fractured (approximately 630 meters or 2,066 feet). However, the service could not confirm that the fluid pressure at the hypocentral location of the earthquakes was enough to generate seismicity and given the extensive seismic history in the area, the Service could not determine if the hydraulic fracturing had actually induced the earthquakes. The study also noted that the earthquakes observed were extremely low Magnitude in nature and were felt by only one individual (Holland 2011).

Basel, Switzerland Case Study

One of the most publicized instances of induced seismicity that is cited by critiques of hydraulic fracturing occurred at a geothermal project in Basel, Switzerland. In 2006, during the course of the development of an enhanced geothermal reservoir at a depth of about 5 km (3.1 miles) underneath the city, a Magnitude 3.4 earthquake was triggered. The earthquake occurred after 11,500 cubic meters of liquid (3.03 million gallons) were injected into a 5-km (3.1 miles) deep injection well. A steady increase in seismicity was detected in response to a gradual increase in flow rate and wellhead pressure. Monitors recorded more than 10,000 seismic events during the injection phase. After water had been injected for about 16 hours, a Magnitude 2.6 event occurred within the reservoir, which exceeded the safety threshold for continued well stimulation. In response, injection was halted prematurely. Two additional seismic events of Magnitude-2.7 and 3.4 occurred several hours later. At that point, the well was opened and the water was allowed to flow back. The seismic activity declined quickly thereafter. The well was officially shut down in 2009 (Deichmann and Giardini 2009).

A study was commissioned by the Canton of Basel-Stadt and the Swiss federal government to assess the seismic risk resulting from continued development and operation of the geothermal system. The study addressed the effect of continued development and operation of the geothermal facility and its effect on induced seismicity, as well as the effects of operations on natural seismicity in the Basel Region; i.e., “triggered seismicity.” A 3-D geological model was used to map eight relevant faults in the vicinity of the Basel geothermal reservoir. The seismic activity (time intervals when large earthquakes could be expected) of each fault was estimated, and it was determined that the presence of the geothermal reservoir could have a direct impact on the recurrence time of these natural earthquakes by modifying subsurface stresses but that variation would be small. The study also found that there is a possibility that earthquakes exceeding the strength of previous seismic activity would occur during continued development and operation of the facility. Based on model simulations, the largest “triggered” seismic event was predicted to have a Magnitude 4.5 (Baisch et. al 2009).

Rocky Mountain Arsenal

One of the first records of induced seismicity linked to deep underground injection was at the Rocky Mountain Arsenal, where a deep injection well constructed in 1961 was used to dispose of wastes from the U.S. Army’s chemical weapon testing operations. The well was drilled to a depth of 12,045 feet. It was cased and sealed to 11,975 feet, and the remaining 70 feet were left as an open hole for fluid injection. 165 million gallons of Basin F liquid waste, consisting of salty water that includes some metals, chlorides, wastewater and toxic organics was injected into the well from 1962–1966. During that time period, there were several small earthquakes in the area, and in 1966 a correlation was noticed between the frequency of earthquakes and the volume of water being pumped. Pumping was halted in 1966 due to the possibility that the fluid injection was triggering the earthquakes in the area. Over the next two years earthquakes continued to occur as far as 6 km (3.7 miles) from the injection well as the pressure front caused by injection dissipated (Nicholson and Wesson 1990). The well remained unused for almost twenty years until the army permanently sealed it in 1985.

Relevance to Inglewood Oil Field

The studies of the potential link between hydraulic fracturing and earthquakes have all concluded that the hydraulic fracturing produces small, imperceptible microseismic events (like dropping a milk bottle on the floor according to Stanford geophysicist, Mark Zoback [COGA 2012]) as part of the process itself. These microseismic events were recorded by the microseismicity monitoring conducted during the VIC1-330 and VIC1-635 hydraulic fracturing operations completed as a part of this study, and most were restricted to the target oil producing zone. These microseismic events did not cause any recordable event at the surface, based on two types of vibration monitoring and the Cal-Tech accelerometer.

The Inglewood Oil Field does not inject wastewater in the manner where small (Magnitude 3 or 4) earthquakes have been detected in Ohio and elsewhere. In those cases, the wastewater is injected into a formation other than the gas-producing zone. At Inglewood, the waterflood operation injects treated produced water into the depressurized oil-bearing formation. The waterflood therefore does not increase the subsurface pressure. The waterflood has been conducted waterflood operations since 1954, and since 1971 at a rate to halt subsidence. No earthquakes on the Newport-Inglewood Fault zone, or any other fault zone, have been attributed to the waterflood operation. This history is validated by the National Research Council study which found, “the potential for felt induced seismicity due to secondary recovery and EOR is low” (NRC 2012). As part of the CSD conditions, ground motion, vibration, and seismicity are monitored to determine whether there is a connection. In the two years of monitoring so far, there has been no connection between oil field operations, including the waterflood, high-rate gravel pack, or high-volume hydraulic fracturing operations, and seismicity, vibration, or ground movement.

Petersen and Wesnousky (1994) evaluated all seismic events greater than Magnitude-2 on the Newport-Inglewood Fault zone, and determined that most epicenters are located at depths between 3.5 miles and 12 miles depth (Petersen and Wesnousky 1994, Hauksson 1987). In comparison, the waterflood operation at the Inglewood Oil Field extends to depths of up to 3,000 feet (0.57 mile) and the deepest hydraulic fracturing occurs at less than 10,000 feet depth (1.9 miles). Therefore, any effects of oil field operations are much shallower than the zones typically associated with earthquake epicenters along the Newport-Inglewood Fault zone.

4.6 Methane

4.6.1 Subsurface Occurrence of Methane

As described in Chapter 2, the Los Angeles Basin is the richest oil basin in the world based on the volume of hydrocarbons per volume of sedimentary fill (Biddle 1991). Most of the oil and gas lies trapped beneath both shales and faults, allowing it to accumulate at depth. However, some surface seeps do occur, as at the La Brea Tar Pits, and methane also migrates to the surface.

There are three types of gases that may exist within the geological and soil units underlying the active surface of the Inglewood Oil Field, including biogenic (swamp or sewer) gas, thermogenic (field) gas, and processed natural (or piped) gas.

Biogenic gas is primarily methane with carbon dioxide and sulfide gases that result from decomposition of organic material, such as from former marshy areas or from sewers. Although biogenic gas contains of mostly methane and carbon dioxide, these gases also consist of lesser amounts of ethane, propane, and butane, as well as trace amounts of hydrogen sulfide and ammonia. In the active surface field area, marshy areas were formerly present immediately north of the Baldwin Hills, in the former floodplain of Ballona Creek (Hsu et al. 1982). In addition, the large-diameter (approximately 15-foot) City of Los Angeles North Outfall Replacement Sewer underlies the active surface field boundary. Both of these features are potential sources of biogenic gas.

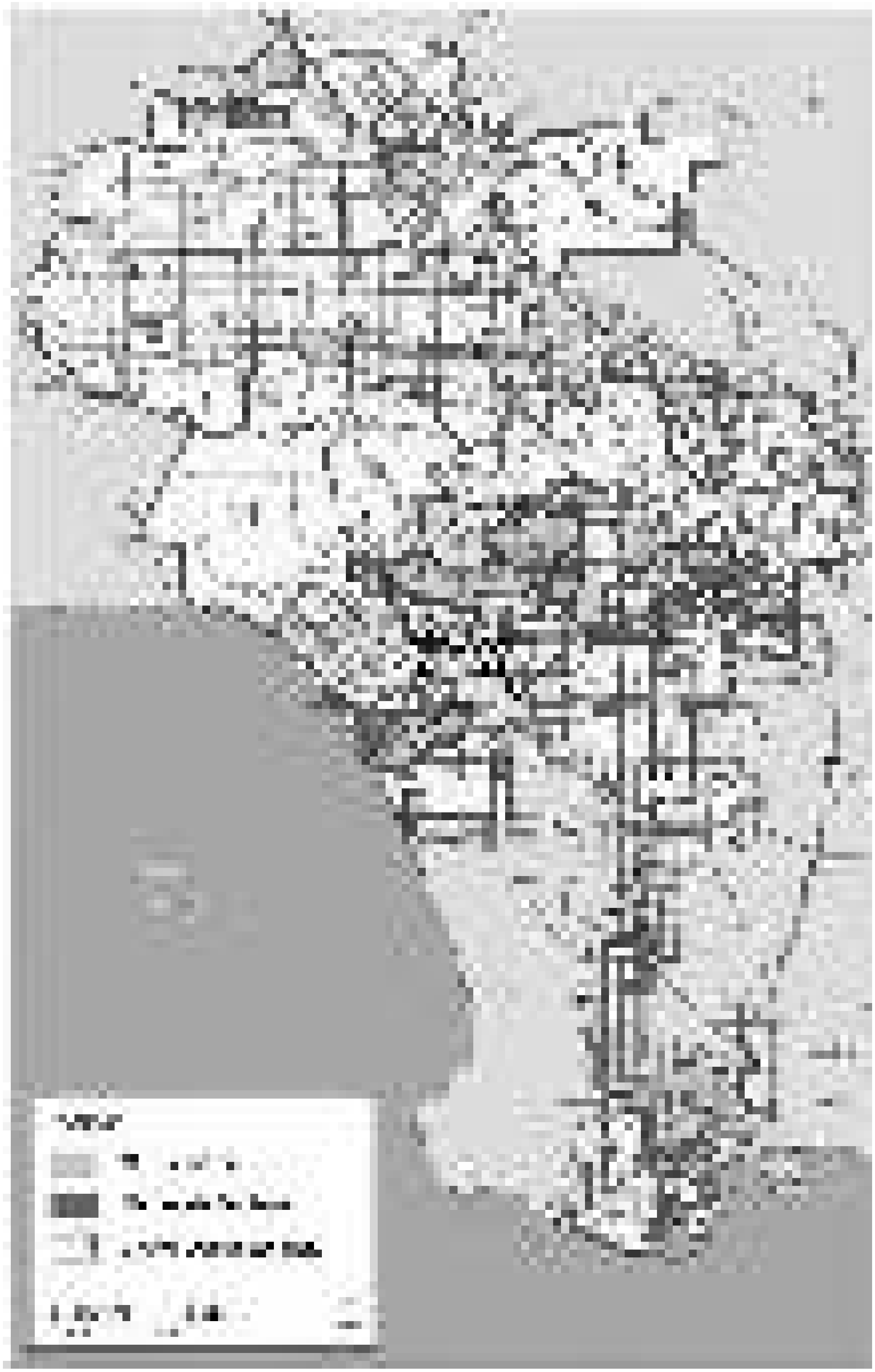
Thermogenic gas is generated at depth when increased temperatures and pressures alter organic material to form gases. Similar to biogenic gas, thermogenic gas contains a broad range of gas components including methane, ethane, propane, and butane, as well as trace amounts of toxic gases, including hydrogen sulfide. Activities at the Inglewood Oil Field produce oil and associated thermogenic gas.

Natural gas at the field is processed and sold to the BP Carson refinery, sold to Southern California Gas Company, or utilized for field use. Processed natural gas began as thermogenic gas derived from the oil and gas producing zones, and then had most non-methane components removed and reused.

These various types of gases exhibit distinct chemical characteristics, which permits "finger-printing" of gases, or differentiation between gas types (California Public Utilities Commission 2004).

4.6.2 Regulatory Framework for Methane

Due to the probability of methane gas releases from naturally occurring thermogenic and biogenic sources in this prolific oil and gas province, the City of Los Angeles has established a zoning ordinance identifying two zones, a Methane Zone and a Methane Buffer Zone (Figure 4-7). Special requirements for new construction, existing construction, and monitoring for methane have been established for these zones. The Baldwin Hills are not in the City of Los Angeles, and therefore are not classified on the methane map. However, the field is surrounded by such zones, and there is likelihood that methane conditions beneath the field are consistent with the relatively high background levels of methane in the Los Angeles Basin.



Source: City of Los Angeles 2004

Figure 4-7 Methane Zone Map

Following an explosion at the Ross Department Store in the Wilshire-Fairfax district of Los Angeles, and in an effort to avoid land use conflicts between oil field operations and urban environments, Senate Bill 1458 (Roberti) in 1986 directed the Department of Conservation and DOGGR to identify areas with the greatest potential for gas migration into structures, which could cause potential health and safety issues. A *Study of Abandoned Oil and Gas Wells and Methane and Other Hazardous Gas Accumulations* (Geoscience Analytical, Inc. 1986) identified eight high risk areas in the Southern California region that have the potential to cause a health and safety issue. These areas are categorized based on their locations within urban areas, having a history of seeps, and history of having plugged and abandoned wells within their boundaries. The Inglewood Oil Field was not identified as a high risk area in the study. The areas identified include: Salt Lake Oil Field (City of Los Angeles – Fairfax/Wilshire District); Newport Oil Field (City of Newport Beach); Santa Fe Springs Oil Field (City of Santa Fe Springs); the Rideout Heights area of the Whittier Oil Field (City of Whittier); Los Angeles City Oil Field (City of Los Angeles); Brea-Olinda Oil Field (City of Brea); Summerland Oil Field (City of Summerland); and Huntington Beach Oil Field (City of Huntington Beach) (Geoscience 1986).

Gas samples were collected at all high risk locations in the DOGGR study and analyzed to determine the hydrocarbon gas content and the origin of the soil gases. Of all the samples collected and indicating gas seepage, only two had the potential of originating from old oil and gas wells. In these two locations (Newport Beach and Huntington Beach) it was suspected that structures were built over old wells that were not plugged and abandoned to current standards. Although these old wells could have been the cause of the gas seepage, gas analysis indicated that the gas was biogenic in nature (i.e., not related to the oil and gas productive zone in the wells) and therefore the wells may have only been a conduit for the shallow biogenic gas (DOGGR, personal communication 2008 reported in CSD EIR).

Hamilton and Meehan (1992) also examined the causes of methane migration and the explosion in the Ross Store, as well as another natural gas vent in the Fairfax District near the La Brea Tar Pits. They reported that the methane was thermogenic in origin (that is, from the underlying oil-producing zone), but proposed that an additional scenario could account for the subsurface migration of methane: overpressuring of the oil-producing zone, leading to fracturing of the surrounding rocks and movement of methane along those newly-formed fractures. They recommended that DOGGR monitor injection operations to ensure that injection above the fracture pressure during produced water injection not exceed the formation fracture stress.

Chilingar and Endres (2005) have also evaluated methane migration in oil and gas producing areas, principally the many urban oil fields in Southern California. They conclude that "virtually all leaks can be traced to the poor well completion and/or abandonment procedures (i.e., poor cementing practices)." They advocate the evaluation of the integrity of old wells in the urban setting as a means to reduce this risk.

DOGGR reviews all applications for water injection wells under the authority of the UIC program. Injection wells for oil and gas development are Class 2 wells in this program, and DOGGR must evaluate the proposed injection pressures, the surrounding geology, and the well integrity of wells within one-quarter mile of any new proposed injection well. Because the field is contiguous and not interspersed with urban and residential development, the active Inglewood Oil Field is well positioned to address these issues in the Baldwin Hills. Because the field is active, it ensures that any unidentified issues will be addressed during field development, in comparison to orphan wells elsewhere in the state for which there is no identified owner.

4.6.3 Gas Monitoring Prior to Hydraulic Fracturing at the Inglewood Oil Field

Background soil gas methane concentrations throughout Southern California are typically 50 parts per million volume (ppmv) or less, although in Los Angeles certain areas are known to have higher background concentrations and have been identified on City Methane Zone Maps.

Since 2007, PXP has conducted annual soil gas surveys throughout the Inglewood Oil Field to test for methane concentrations and potential gas leaks from abandoned and idle wells. In 2007, GeoScience Analytical, Inc. sampled 94 locations, probing soil to a depth of four feet. The majority of these sampling locations were in the vicinity of idled or abandoned wells. Soil gases were extracted from each of the soil probes and transported to the laboratory for analyses of C_1 (methane) up to C_7 hydrocarbons, and hydrogen sulfide. The same 94 sample locations were tested in 2008 and 2009, with the addition of two other sites (GeoScience Analytical, Inc. 2009). Figure 4-8 depicts the sampling locations.

Methane concentrations detected in 2007 ranged from 1.0 ppmv to a high of 981,400 ppmv in the case of location #7, located near well LAI 1-130, which was an idled well. Given the high value for this location, additional soil gas vapor testing was done at 12 sites located around well LAI 1-130. The results of this additional sampling indicated that the source of the gas was most likely well LAI 1-130. The well was subsequently abandoned to the current DOGGR standards.

4.6.4 Gas Monitoring After High-Volume Hydraulic Fracturing

Soil gas testing was conducted again in 2011, following the high-volume hydraulic fracture operation of VIC1-330 in early September. During this sample event, 31 soil samples were taken and tested for C_1 to C_7 hydrocarbons and hydrogen sulfide. Of the samples tested, only two had readings greater than 500 ppmv (1,346 and 551 ppmv); both of which were well under the level for concern (12,500 ppmv) (GeoScience Analytical, Inc. 2011). GeoScience Analytical, Inc. concluded that the soil gases detected on the field were most likely the result of bacterial decomposition of crude oil in the near surface soils, i.e. biogenic (GeoScience Analytical, Inc. 2011). Isotopic analysis of three of the shallow soil gas samples was conducted to validate this finding. Carbon and hydrogen isotopic ratios were measured, and the results confirm a biogenic source for shallow soil gas (Figure 4-9).

There was no detected correlation between the hydraulic fracturing operation and the detected soil gas on the field.

Cardno ENTRIX

N

0 500 1,000 2,000 Feet

LEGEND

2012 Sample Locations

Location Not Sampled in 2012
(Exempt per CSD — 2 Years Below Background 50 PPM)

Inglewood Oil Field Boundary

PLAINS EXPLORATION & PRODUCTION COMPANY

Figure 4-8

Soil Gas Survey Sample Locations

07 | 05 | 12

4.6.5 Groundwater Monitoring for Methane after Hydraulic Fracturing

Methane in groundwater was tested during the quarterly sampling conducted on the Inglewood Oil Field (see Section 4.2). Groundwater was never measured for methane prior to high-volume hydraulic fracturing. Samples collected after high-volume hydraulic fracturing detected dissolved methane in all but one well (MW-7), at concentrations ranging from 0.01 to 9.7 mg/L. Wells MW-8, MW-11A, MW-11B, and MW-13 are located across the center of the field, and had concentrations ranging from 3.5 to 9.7 mg/L methane; all other concentrations were below 0.190 mg/L. Methane in water is not toxic and therefore, there is no drinking water standard (MCL) established. The City of Los Angeles methane zoning ordinance does not address methane in groundwater; the ordinance only addresses levels in soil gas and applies construction standards (Ordinance No. 175750). In water supplies, methane volatilizes from water, and at very high concentrations can displace oxygen. The U.S. Office of Surface Mining considers 28 mg/L in a water supply well as indicative that action be taken to reduce the concentration before use. Concentrations below 10 mg/L are considered safe, and between 10 and 28 mg/L the U.S. Office of Surface Mining suggests monitoring. The U.S. Bureau of Land Management also lists 10 mg/L as a safe concentration (U.S. Office of Surface Mining 2001). Therefore, all methane detections noted in groundwater samples within the oil field were within the level considered safe for any conditions. None of the water beneath the Baldwin Hills is used as a water supply, nor does it supply water at a yield suitable for a water supply.

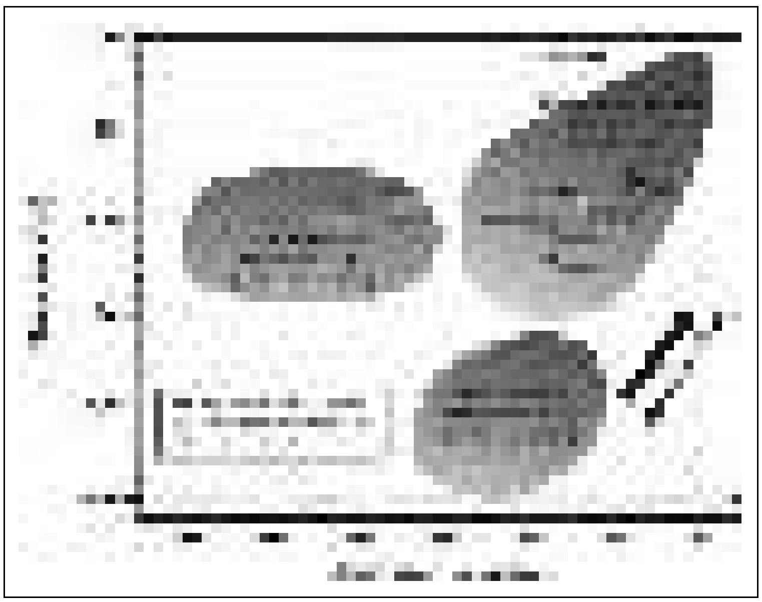


Figure 4-9 Methane Isotopic Results

Based on isotopic analysis of the dissolved methane in groundwater, it is thermogenic (from the oil-bearing formation) in origin, whereas detections in shallow soil gas are biogenic in origin (Figure 4-9). Therefore the methane in water and the methane in soil gas at the Inglewood Oil Field have different sources and are not in continuity. There are shallow occurrences of oil in the Investment Zone, within the Pico Formation, that are not commercially produced. Since these untapped zones are in closest proximity to the water-bearing zones, and the occurrence of methane is pervasive in the monitoring results, it does not appear to be related to oil and gas production activity. The occurrence is also not correlated to the locations of high-volume hydraulic fracturing. None of the levels detected are at concentrations that exceed levels considered safe, and none trigger further action under current regulations.

4.6.6 Methane Emissions and Climate Change

National Issue

Methane is the simplest hydrocarbon (alkane), consisting of one carbon and four hydrogen atoms. Methane is the main component of natural gas, typically over 90 percent by volume, and is one of the most abundant naturally-occurring organic compounds on earth. Pure methane is colorless, odorless, and nontoxic, but highly flammable, which makes it an attractive clean-burning fuel with a "carbon footprint" about 50 percent lower than coal when used to generate electricity. Since methane is 44 percent lighter than air, it dissipates upward when released. However, although nontoxic, methane is a simple asphyxiant and may displace oxygen in enclosed spaces and may also form explosive mixtures with air under certain conditions, thus presenting hazards. Methane is a greenhouse gas with an IPCC GWP coefficient of 21 relative to carbon dioxide. This means that methane has 21 times the averaged relative radiative forcing effect of CO_2 (USEPA 2011a, CCAR 2009).

A public concern related to hydraulic fracturing deals with methane gas that can escape into the atmosphere as a result of hydraulic fracturing operations and contribute to climate change. It is commonly recognized that methane gas can potentially escape as fugitive emissions during well completion. Large volumes of water are forced under pressure into the ground to fracture a rock formation and increase gas flow. A large portion of this water returns to the surface as flowback water within the first several days to weeks after injection. The flowback water is accompanied by quantities of methane that exceed the amount that can be dissolved within the flowback fluids. To assist in minimizing fugitive emissions at the Inglewood Oil Field, all flowback water accompanied by methane gas is piped to portable 500 bbl tanks connected to a South Coast Air Quality Management District (SCAQMD) permitted vapor control system. This is standard practice for oil operations throughout the Los Angeles Basin but differs from the air quality policies in shale gas producing states where in most cases, until recently, emissions from the flowback operations were largely unregulated. The rate of methane released with flowback fluid corresponds to the initial production rate and pressure of a well. Methane is also released during "drill-out," which is the stage of developing shale gases and oil in which the plugs that are set to separate fracturing stages are drilled out to release gas and/or oil for production. Fugitive methane emissions might result from equipment leaks and routine venting from pressure relief valves that are designed to purposefully vent gas (Howarth et al. 2011).

Untreated raw gas can contain some hydrogen sulfide (H_2S) which is highly odorous – the "rotten egg" smell – and the H_2S is toxic in high enough concentrations. Other natural sources of hydrogen sulfide include decaying organic matter under anaerobic (oxygen deprived) conditions. While hydrogen sulfide presents risks to oilfield workers, it is not considered a public safety risk due to safety zones between drilling activities and the general public which provide adequate distances for atmospheric dispersion in the event of leaks.

Relevance to Inglewood Oil Field

The Inglewood Oil Field operates in compliance with the requirements of the SCAQMD pursuant to Rules 463, 1148.1, 1173, and 1176 as applicable, which effectively control emissions of methane and other hydrocarbons into the atmosphere. In this regard emissions regulations relevant to the Inglewood Oil Field are significantly advanced compared to most shale gas producing states where, until recently, emissions from the flowback operations were predominantly unregulated. The Inglewood Oil Field is in the development stage rather than the exploration stage, so natural gas and produced water are contained in pipelines or enclosed tanks. Most concerns expressed regarding methane emissions to air and their effect on climate change are from natural gas fields in the exploration phase, where piping and gas treatment and sale facilities are not yet in place, as it is at Inglewood. As required by SCAQMD, there is an ongoing program of monitoring and repairing fugitive sources of hydrocarbon emissions. On a general comparative basis, air quality regulations in many oil and gas producing states are less comprehensive than the regulations adopted by the SCQAMD and apply to oil and gas activities at the Inglewood Oil Field.

The “carbon footprint” concern has been minimized at the Inglewood Oil Field as a result of its historic, stable long-term operation, and extensive local regulatory compliance framework for air quality including greenhouse gases. Unlike drilling operations in other less-regulated western states, current SCAQMD regulations prohibit uncontrolled venting of gas to the atmosphere which effectively mitigates the effects of hydraulic fracturing during well completion.

4.7 Other Emissions to Air

In compliance with SCAQMD Rules 463, 1148.1, 1173 and 1176 as applicable, hydrocarbon emissions at the Inglewood Oil Field are controlled and also monitored as described in an Air Monitoring Plan in accordance with Section E.2(d) of the Baldwin Hills CSD. This plan requires monitoring for hydrogen sulfide and total hydrocarbon vapors. It also requires that drilling or completion operations shut down if monitoring detects concentrations of hydrogen sulfide greater than 10 ppmv or hydrocarbon concentration of 1,000 ppmv or greater. Construction equipment and vehicles used for on-road and off-road purposes are also regulated by the CSD under Sections E.2(j) through E.2(n).

The following analyses focus on how the new USEPA hydraulic fracturing rules mesh with current SCAQMD rules, whether SCAQMD compliance comprises “de facto” USEPA compliance, or whether additional measures (activities, equipment) will be required to comply with USEPA notwithstanding SCAQMD. Applicable SCAQMD rules which prohibit uncontrolled emissions of VOC/GHG (e.g., raw untreated natural gas, tank headspace vapors, fugitive hydrocarbon leaks, etc.) are identified below:

- **Rule 463.** Organic Liquid Storage
- **Rule 1148.1.** Oil and Gas Production Wells
- **Rule 1173.** Control of Volatile Organic Compound Leaks and Releases from Components at Petroleum Facilities and Chemical Plants
- **Rule 1176.** VOC Emissions from Wastewater Systems

40 CFR Part 63 – New Source Performance Standards (NSPS) for New Hydraulically Fractured Wells (drilled after August 23, 2011)

To ensure that smog-forming volatile organic compounds (VOCs) are controlled without slowing natural gas production, USEPA’s final NSPS for VOCs establishes two phases for reducing VOCs during well completion. This approach will provide industry time to order and manufacture enough equipment to capture natural gas using a process called *green completions,* also known as "reduced emissions completions."

USEPA established the phased approach to address concerns raised in comments related to the availability of equipment and operators to conduct green completions in time to meet compliance dates in the proposed rule.

Phase 1

In the first phase (before January 1, 2015), industry must reduce VOC emissions either by flaring using a completion combustion device or by capturing the gas using green completions with a completion combustion device (unless combustion is a safety hazard or is prohibited by state or local regulations).

- A completion combustion device burns off the gas that would otherwise escape during the well-completion period (combustion generally would occur through pit flaring). Industry may use completion combustion devices to reduce VOC emissions until January 1, 2015, unless state or local requirements prohibit the practice or require more stringent controls (e.g., SCAQMD Rule 1148.1). USEPA encourages industry to begin using green completions during this time.

Phase 2

Beginning January 1, 2015, operators must capture the gas and make it available for use or sale, which they can do through the use of green completions.

- A completion device which captures the gas that would otherwise escape during the well-completion period will be required. Industry must use completion devices to reduce VOC emissions beginning January 1, 2015. Captured gas must be sent to economic use, either as fuel gas, sales gas, or reinjection.
- USEPA estimates that use of green completions for the three- to 10-day flowback period reduces VOC emissions from completions and recompletions of hydraulically fractured wells by 95 percent at each well (USEPA 2012d)
- Both combustion and green completions will reduce the VOCs that currently escape into the air during well completion. Capturing the gas through a green completion prevents a valuable resource from going to waste and does not generate NO_X, which is a byproduct of combustion.
- Methane, a potent greenhouse gas, and air toxics, which are linked to cancer and other serious health effects, also would be significantly reduced as a co-benefit of reducing VOCs.

Exceptions for New Wells

Green completions are not required for:

- New exploratory ("wildcat") wells or delineation wells (used to define the borders of a natural gas reservoir), because they are not near a pipeline to bring the gas to market.

- Hydraulically fractured low-pressure wells, where natural gas cannot be routed to the gathering line. Operators may use a simple formula based on well depth and well pressure to determine whether a well is a low-pressure well.

Owners/operators must reduce emissions from these wells using combustion during the well-completion process, unless combustion is a safety hazard or is prohibited by state or local regulations.

SCAQMD Rule 1148.1 – Oil and Gas Production Wells (relevant excerpts below, see entire rule for details)

"(a) The purpose of this rule is to reduce emissions of volatile organic compounds (VOCs) from the wellheads, the well cellars and the handling of produced gas at oil and gas production facilities.

(b) This rule applies to onshore oil producing wells, well cellars and produced gas handling activities at onshore facilities where petroleum and processed gas are produced, gathered, separated, processed and stored. Natural gas distribution, transmission and associated storage operations are not subject to the requirements of this rule.

(d)(6) Effective January 1, 2006, the operator of an oil and gas production facility shall not allow natural gas or produced gas to be vented into the atmosphere. The emissions of produced gas shall be collected and controlled using one of the following:

- *A system handling gas for fuel, sale, or underground injection; or*
- *A device, approved by the Executive Officer, with a VOC vapor removal efficiency demonstrated to be at least 95% by weight per test method of paragraph (g)(2) or by demonstrating an outlet VOC concentration of 50 ppm according to the test method in paragraph (g)(1). If the control device uses supplemental natural gas to control VOC, it shall be equipped with a device that automatically shuts off the flow of natural gas in the event of a flame-out or pilot failure.*

(d (7) Except as Rule 1173 applies to components of produced gas handling equipment located within 100 meters of a sensitive receptor, the operator shall repair any gaseous leaks of 250 ppm TOC or greater by the close of the business day following the leak discovery or take actions to prevent the release of TOC emissions to the atmosphere until repairs have been completed.

(d (8) Effective March 5, 2004, unless approved in writing by the Executive Officer, CARB, and USEPA as having no significant emissions impacts, no person shall:

- *Remove or otherwise render ineffective a well cellar at an oil and gas production well except for purposes of abandonment to be certified by the California Department of Conservation, Division of Oil, Gas and Geothermal Resources; or*
- *Drill a new oil and gas production well unless a well cellar is installed for containment of fluids."*

Analysis

Until December 31, 2014, compliance with SCAQMD Rule 1148.1 subparts (d)(6)(A) – capture, or (d)(6)(B) – incineration, constitutes compliance with Phase 1 of the new USEPA rule. Beginning January 1, 2015, only capture pursuant to subpart (d)(6)(A) can be used for so-called "green completions" under Phase 2; incineration pursuant to subpart (d)(6)(B) will no longer be allowed by USEPA. Thus, subpart (d)(6)(B) will be superseded by 40 CFR 63 on January 1, 2015. Since PXP presently complies with Rule 1148.1, PXP is also presently in basic compliance with 40 CFR 63.

Other Equipment - NSPS Requirements for New & Modified Pneumatic Controllers

Pneumatic controllers are automated instruments used for maintaining a condition such as liquid level, pressure, and temperature at wells and natural gas processing plants, among other locations in the oil and natural gas industry. These controllers often are powered by high-pressure natural gas and may release gas (including VOCs and methane) with every valve movement, or continuously in many cases as part of their normal operations.

The final rule affects high-bleed, gas-driven controllers (with a gas bleed rate greater than 6 standard cubic feet per hour) that are located between the wellhead and the point where gas enters the transmission pipeline.

- The rule sets limits for controllers based on location. For controllers used at the well site, the gas bleed limit is 6 cubic feet of gas per hour at an individual controller.
- The final rule phases in this requirement over one year, to give manufacturers of pneumatic controllers time to test and document that the gas bleed rate of their pneumatic controllers is below 6 cubic feet per hour.
- Low-bleed controllers used at well sites (with a gas bleed rate less than 6 standard cubic feet per hour) are not subject to this rule.

The final rule includes exceptions for applications requiring high-bleed controllers for certain purposes, including operational requirements and safety. The rule also includes requirements for initial performance testing, recordkeeping and annual reporting.

Analysis

Since PXP presently complies with SCAQMD Rule 1173 (*Control of Volatile Organic Compound Leaks and Releases from Components at Petroleum Facilities and Chemical Plants; and Rule 466.1 – Valves and Flanges*) PXP is also presently in basic compliance with this section of 40 CFR 63, notwithstanding particulars and details. This topic will also require coverage in the MRRP discussed above.

Other Equipment - NSPS Requirements for Storage Vessels at the Well Site

Storage tanks at natural gas well sites are commonly used to store condensate, crude oil and produced water. These tanks may be subject to two standards: the NSPS for VOCs and the major source air toxics standards (NESHAP) for Oil and Natural Gas Production.

NSPS Requirements

New storage tanks with VOC emissions of 6 tons a year or more must reduce VOC emissions by at least 95 percent. USEPA expects this will generally be accomplished by routing emissions to a combustion device.

- To ensure enough combustion devices are available, the final rule provides a one-year phase-in for this requirement.
- After one year, owners/operators of new storage tanks at sites with wells in production must comply. Owners/operators at sites with no wells in production will have 30 days to determine the emissions from a tank; and another 30 days to install controls.

Air Toxics Requirements

In response to public comments, USEPA did not finalize proposed air toxics standards for storage vessels *without* the potential for flash emissions, which currently are not regulated under the NESHAP for Oil and Natural Gas Production. The agency determined that it needs additional data in order to establish emission standards for this type of storage vessel. The previous standards for storage tanks *with* the potential for flash emissions remain in place.

- The final rule amends the definition of “associated equipment, “ meaning that emissions from all storage vessels now will be counted toward determining whether a facility is a major source under the NESHAP for Oil and Natural Gas Production.

Analysis

Since PXP presently complies with SCAQMD Rule 1176 (*VOC Emissions from Wastewater Systems*), Rule 463 (*Organic Liquid Storage*), and Rule 1178 (*Further Reductions of VOC Emissions From Storage Tanks at Petroleum Facilities*) PXP is also presently in basic compliance with this section of 40 CFR 63, notwithstanding particulars and details.

In particular, during the hydraulic fracturing process, any fluid flowback is captured in a closed system diverted to a portable 500 bbl. tank connected to a SCAQMD-permitted hydrocarbon vapor control system (activated carbon canisters). While PXP currently logs and reports the performance of this system to SCAQMD pursuant to the rule, this topic also requires coverage in the MRRP discussed above. Also, all aboveground stationary tanks are vapor tight and connected to existing vapor recovery systems as required by Rule.

Construction Emissions Estimation for Off-road Equipment and On-road Vehicles

Cardno ENTRIX has estimated mass emissions of criteria pollutants and greenhouse gases (GHG) for off-road equipment and on-road vehicles using emission factors published by the SCAQMD (SCAQMD 2008) and USEPA (USEPA 2011e, 2011f). The project schedule and equipment/vehicle list provided by PXP and Halliburton served as the basis for the analysis. The results of the analysis are presented in the emissions thresholds and summary Tables 4-2, 4-3, and 4-4 contained in this section (SCAQMD 2011). As shown in the tables, the estimated emissions are well below the daily limits set by the SCAQMD.

Table 4-2 Emissions Thresholds – South Coast AQMD

Criteria Pollutant	Temporary Construction lbs/day	Permanent Operation[1] lbs/day
Volatile Organic Compounds (VOC as CH_4)	75	55
Carbon Monoxide (CO)	550	550
Oxides of Nitrogen (NO_X as NO_2)	100	55
Sulfur Dioxide (SO_X as SO_2)	150	150
Respirable Particulates (PM_{10})	150	150
Fine Particulates ($PM_{2.5}$)	55	55

Source: SCAQMD 2011

[1] Does not apply to this project (not a permanent stationary source)

Table 4-3 Estimated Emissions of Criteria Pollutants

Criteria Pollutants	Maximum lbs/day	Threshold lbs/day	Total tons
Volatile Organic Compounds (VOC as CH_4)	2.0	75	0.007
Carbon Monoxide (CO)	13.8	550	0.048
Oxides of Nitrogen (NO_X as NO_2)	13.8	100	0.048
Sulfur Dioxide (SO_X as SO_2)	0.01	150	0.000
Combustion Particulates (C-PM_{10})	0.7	150	0.002
Combustion Particulates (C-$PM_{2.5}$)	0.6	55	0.002

Table 4-4 Estimated Emissions of Greenhouse Gases

Greenhouse Gases	Daily lbs/day	Total Project	
		Tons[1]	Tonnes[2]
Carbon Dioxide (GHG - CO_2)	1,320	4.62	4.19
Methane (GHG - CH_4)	0.03	0.0001	0.0001
Nitrous Oxide (GHG - N_2O)	0.08	0.0003	0.0002
Carbon Dioxide Equivalents (CO_2 eqv)	1,344	4.71	4.27

Sources: USEPA 2011e, 2011f, CCAR 2009

[1] short ton = 2,000 lbs

[2] metric tonne = 1,000 kg or 2,204.6 lbs

Methodology

For general engine exhaust emissions, the pre-processed SCAQMD factors are outputs from the California Air Resources Board (CARB) EMFAC and OFFROAD software applications and are the same conservative factors used in the official statewide URBEMIS and CalEEMod software applications for general land use planning in all 58 counties. For federal relevancy in all 50 states, the on-road and off-road factors are consistent with 40 CFR Parts 9, 69, 80, 86, 89, 94, 1039, 1048, 1051, 1065, and 1068 as applicable. For diesel off-road equipment with specified Tiers (1, 2, 3 or 4), engine exhaust emissions are based on applicable standards pursuant to 40 CFR 89.112, 13 CCR 2423, and 69 FR 38957-39273.

SCAQMD on-road and off-road factors were used for volatile organic compounds (VOC), carbon monoxide (CO), nitrogen oxides (NO_X), sulfur oxides (SO_X), respirable particulate matter (PM_{10}), carbon dioxide (CO_2), and methane (CH_4). USEPA factors were used for nitrous oxide (N2O), which are not included in the SCAQMD factors. For specified off-road Tiers, USEPA factors for VOC, CO, NO_X, SO_X, PM_{10}, CO_2, CH_4 and N_2O were used. For estimation purposes, fine particulate matter ($PM_{2.5}$) was quantified as 92 percent of PM_{10} for consistency with the EMFAC software (SCAQMD 2008). Where applicable, off-road and/or on-road fugitive dust emissions were estimated using USEPA algorithms contained in Chapters 11 and 13 of AP-42 (USEPA 2011a, 2011b).

Global Warming Potential (GWP) coefficients developed by the Intergovernmental Panel on Climate Change (IPCC) were used to quantify the globally averaged relative radiative forcing effects of a given GHG, using carbon dioxide as the reference gas. Accordingly, GWP coefficients of 1 for CO_2, 21 for CH_4, and 310 for N_2O were applied to aggregate GHGs as CO_2 equivalents (CO_2e) (USEPA 2011e, CCAR 2009).

4.8 Noise and Vibration

Noise attenuation and noise limits for activities occurring on the Inglewood Oil Field are addressed in Section E.5 of the CSD. This regulation sets hours for quiet drilling on the oil field (as outlined in the associated Quiet Mode Drilling Plan) and time limits for construction and deliveries to the oil field. Vibration levels are addressed in Section E.6 and must not exceed a velocity of 0.25 mm/second over a range of 1 to 100 hertz (Hz) in any developed area. The CSD requires that noise and vibration levels be monitored on the oil field to ensure that oil operations do not exceed the set thresholds. Table 4-5 lists noise levels for various types of sources for reference (70 dB, which is an annoyingly loud noise level to some individuals is used as an arbitrary base of comparison).

Table 4-5 Noise Sources and Their Effects

Noise Source	Decibel Level	Human Effects
Jet take-off (at 25 meters)	150	Eardrum rupture
Aircraft carrier deck	140	
Military jet aircraft take-off from aircraft carrier with afterburner at 50 feet (130 dB).	130	
Thunderclap, chain saw. Oxygen torch (121 dB).	120	Painful. 32 times as loud as 70 dB.
Steel mill, auto horn at 1 meter. Turbo-fan aircraft at takeoff power at 200 feet (118 dB). Riveting machine (110 dB); live rock music (108 - 114 dB).	110	Average human pain threshold. 16 times as loud as 70 dB.
Jet take-off (at 305 meters), use of outboard motor, power lawn mower, motorcycle, farm tractor, jackhammer, garbage truck. Boeing 707 or DC-8 aircraft at one nautical mile (6080 feet) before landing (106 dB); jet flyover at 1000 feet (103 dB); Bell J-2A helicopter at 100 feet (100 dB).	100	8 times as loud as 70 dB. Serious damage possible in 8-hr exposure
Boeing 737 or DC-9 aircraft at one nautical mile (6080 feet) before landing (97 dB); power mower (96 dB); motorcycle at 25 feet (90 dB). Newspaper press (97 dB).	90	4 times as loud as 70 dB. Likely damage 8-hr exposure
Garbage disposal, dishwasher, average factory, freight train (at 15 meters). Car wash at 20 feet (89 dB); propeller plane flyover at 1000 feet (88 dB); diesel truck 40 mph at 50 feet (84 dB); diesel train at 45 mph at 100 feet (83 dB). Food blender (88 dB); milling machine (85 dB); garbage disposal (80 dB).	80	2 times as loud as 70 dB. Possible damage in 8 hr exposure.
Passenger car at 65 mph at 25 feet (77 dB); freeway at 50 feet from pavement edge 10 a.m. (76 dB). Living room music (76 dB); radio or TV-audio, vacuum cleaner (70 dB).	70	Arbitrary base of comparison. Upper 70s are annoyingly loud to some people.
Conversation in restaurant, office, background music, Air conditioning unit at 100 feet	60	Half as loud as 70 dB. Fairly quiet
Quiet suburb, conversation at home. Large electrical transformers at 100 feet	50	One-fourth as loud as 70 dB.
Library, bird calls (44 dB); lowest limit of urban ambient sound	40	One-eighth as loud as 70 dB.
Quiet rural area	30	One-sixteenth as loud as 70 dB. Very Quiet
Whisper, rustling leaves	20	
Breathing	10	Barely audible

Sources: Federal Interagency Committee on Noise 1974; 1992 *Federal Agency Review of Selected Airport Noise Analysis Issues*, Federal Interagency Committee on Noise (August 1992).

To address concerns regarding perceptible vibration and noise during high-volume hydraulic fracturing operations, PXP commissioned Behrens and Associates, Inc., a firm specializing in noise and vibration studies, to measure produced vibration during the VIC1-330 and VIC1-635 high-volume hydraulic fractures and the TVIC-221 and TVIC-3254 high-rate gravel pack events. The ground-borne vibration survey for each event was completed while all equipment was operated under normal loads and conditions.

The high-volume hydraulic fracturing treatment on September 16, 2011, was completed on the VIC1-330 well, located in the northwestern portion of the field. Ground-borne vibration levels were measured in one direction (west) at 10-foot intervals from the high-volume hydraulic fracture operation (Figure 4-10A). Measured levels indicate that the maximum ground-borne vibration produced during the operation was 0.006 inch per second (0.1524 mm/second), as measured 40 feet from the operation. At 160 feet from the operation, measured vibration was 0.001 inch per second (0.0254 mm/second). As shown on the Figure 4-10 below, both of these levels are imperceptible to humans (Behrens and Associates, Inc. 2011). These measurements are also below the limit set by the CSD. No noise monitoring was conducted during the VIC1-330 treatment.

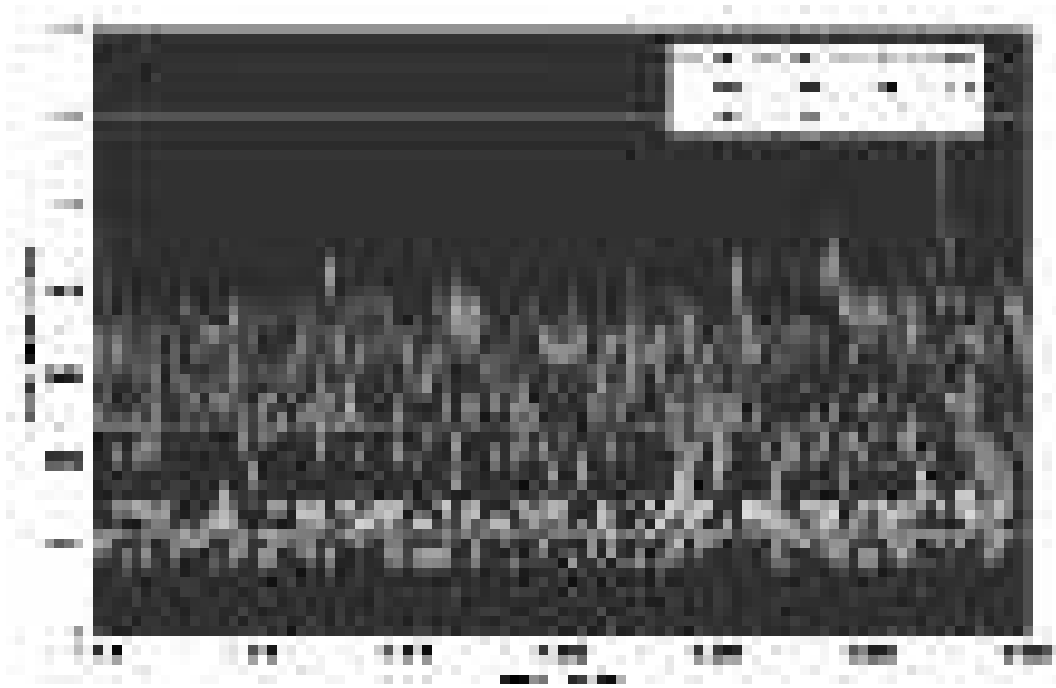


Figure 4-10A Ground Vibration Level Measurements from High-Volume Hydraulic Fracture at VIC1-330

The high-volume hydraulic fracture on January 6, 2012, was completed on the VIC1-635 well. Ground-borne vibration levels were measured to the south of the high-volume hydraulic fracture site at 50, 100, 150, and 200 feet from the well. Measured levels (Figure 4-10B) indicate that the maximum ground-borne vibration was 0.0012 inch per second (0.0305 mm/sec) as measured 50 feet from the operation. Similar to the measured level during the prior high-volume hydraulic fracture operation at VIC1-330, at 150 feet from the operation, measured vibration was 0.001 inch per second (0.0254 mm/second). These measured levels are imperceptible to humans.

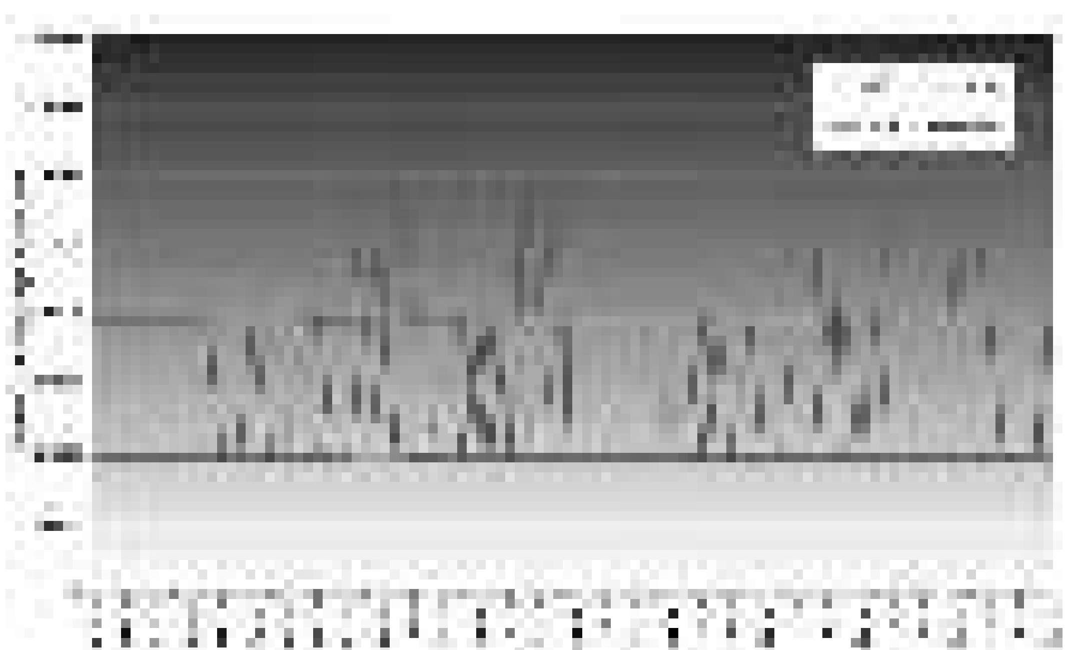


Figure 4-10B Ground Vibration Level Measurements from High-Volume Hydraulic Fracture at VIC1-635

In addition to ground-borne vibration measurements, Behrens and Associates, Inc. also took sound level measurements during the high-volume hydraulic fracturing operation at VIC1-635 using a calibrated sound level meter. The microphone was set at five feet above ground surface. The measured noise level at 100 and 200 feet from the operation was 68.9 and 68.4 decibels (dBA), respectively (Behrens and Associates, Inc. 2012a). These are within CSD limits.

The high-rate gravel pack treatments were completed on January 7 and 8, 2012, on the TVIC-221 and TVIC-3254 wells, which are located immediately adjacent to one another. The ground-borne vibration levels were measured during the high-rate gravel pack at TVIC-221 and TVIC-3254 at a distance of 50, 100, 200, and 300 feet to the east of the high-rate gravel pack operation site (Figure 4-10C).

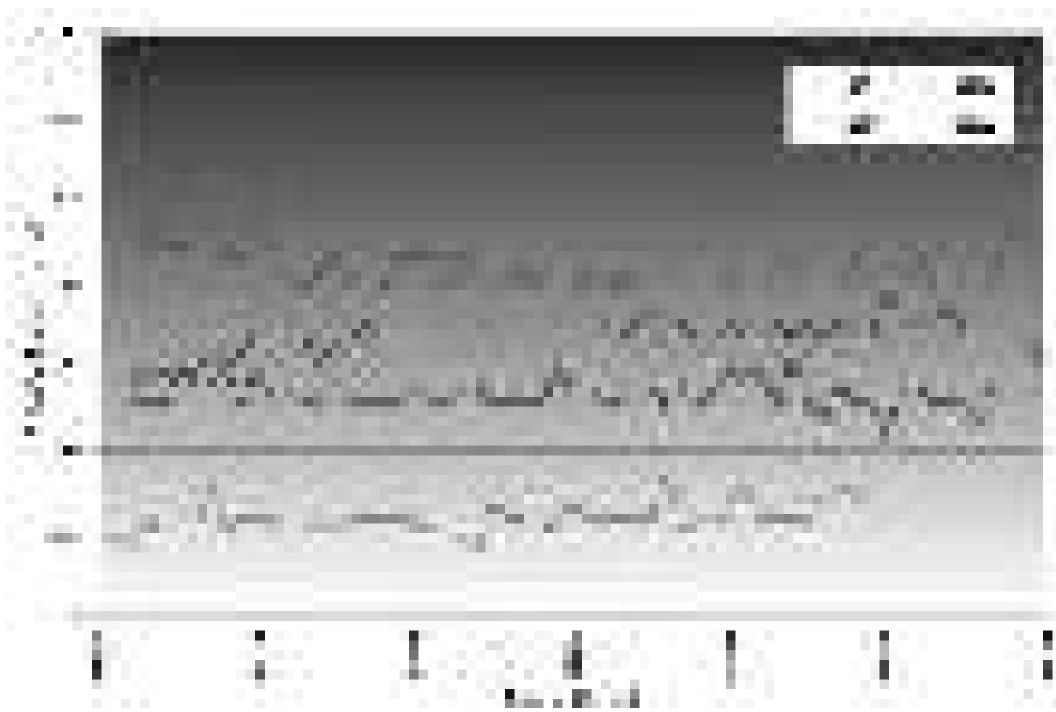


Figure 4-10C Ground Vibration Level Measurements from Gravel Pack Operations at TVIC-221 and TVIC-3254

Measured levels indicate that the maximum ground-borne vibration produced during the high-rate gravel pack operation was 0.012 inch per second (0.304 mm/second), as measured 50 feet from the operation. At 300 feet from the operation, measured vibration was less than 0.004 inch per second (0.102 mm/second). While 0.01 inch per second is the low threshold of vibration that may be perceptible to humans (see Figure 4-11), levels measured further from the site are imperceptible. Further, while the vibration level near the well is greater than the 0.25 mm/second CSD limit, the vibration is decreased to below the limit well away from any developed areas (Behrens and Associates 2012b).

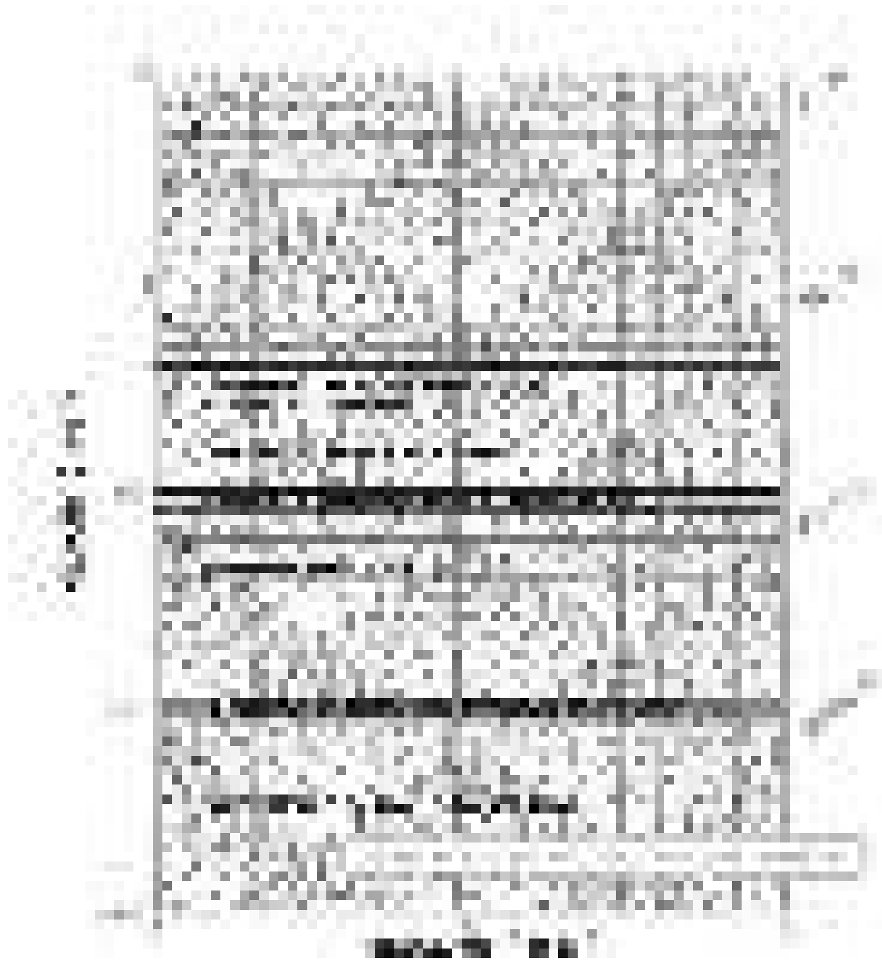


Figure 4-11 Vibration Sensitivity Chart

Sound level measurements were conducted at 100 feet and 200 feet from the TVIC-221 and TVIC-3254 high-rate gravel pack operations. The measured noise level at 100 feet was measured at 68.1 dBA and the noise level at 200 feet was 63.5 dBA (Behrens and Associates 2012b). These are within CSD limits.

4.9 Los Angeles County Department of Public Health Study

In response to health concerns expressed by residents in communities near the Inglewood Oil Field during the EIR process for the CSD, and at the request of the Second Supervisorial District, the Los Angeles County Department of Public Health (LAC DPH) conducted a community health assessment on the population living in communities surrounding the Inglewood Oil Field. The assessment was designed to determine if the health concerns reflect a higher than expected rate or an unusual pattern of disease in the concerned communities. The report was sent to three external peer reviewers who found it to be technically sound.

The conclusions of the health study indicate that there is not a detectable relationship between the activities at the Inglewood Oil Field and the health of the surrounding community. Five types of blood-related cancer (most common types of cancer associated with petroleum exposure) to determine if operations at the Inglewood Oil Field had any adverse impact on cancer rates in the surrounding community. The study found there was no conclusive evidence or link between Inglewood Oil Field activities and cancer rates in the community. The report acknowledges that the data cannot determine whether there is a small adverse health effect, nor can the data address the contribution of other, non-quantifiable health-related issues such as smoking, lack of exercise, and social determinants of health (LAC DPH 2011). As described in Chapter 7, conventional hydraulic fracturing and high-rate gravel pack operations have occurred at the field for several years along with other oil and gas development activity. Any prospective impact from these operations would have contributed to the assessment's baseline findings. The Health Study indicates that operations at the field have not had an adverse effect on the health of the local community.

The Health Assessment included five components, each of which is summarized below:

- An analysis of mortality (death) rates based on data reported on death certificates;
- An analysis of rates of low-birth-weight births based on data reported on birth certificates;
- An analysis of rates of birth defects based on data collected by the California Birth Defects Monitoring Program;
- An analysis of cancer rates based on data compiled by the University of Southern California (USC) Cancer Surveillance Program; and
- A community health survey of self-reported illness, including asthma and other health concerns.

The report of the Health Assessment (LAC DPH 2011) included conclusions for the first four components. The community health survey of self-reported illness was postponed to allow enough time to evaluate the effects of continuous drilling and released in April 2012.

4.9.1 Mortality

From 2000 to 2007, the mortality rate for all causes of death was 731.9 deaths per 100,000 persons in the Inglewood Oil Field communities and 751.7 deaths per 100,000 persons in Los Angeles County, after adjusting for age and the racial/ethnic distribution of the underlying populations. Although the mortality rate appears lower in the Inglewood Oil Field communities, there was no statistically significant difference in the mortality rates for all causes of death, after adjusting for age and race/ethnicity.

The differences in mortality rates for the leading causes of death and premature death do not appear to be related to the geographic location of the Inglewood Oil Field communities. Many of the differences observed within these communities are common in Los Angeles County and represent a significant public health challenge throughout the county. The disparities in mortality rates can best be addressed by targeting the underlying causes of these disparities.

4.9.2 Low Birth Weight

After adjusting for race/ethnicity, the rate of low-birth-weight births was 7.2 per 100 live births in the Inglewood Oil Field communities and 7.0 per 100 live births in Los Angeles County as a whole. There was no statistical difference in the rates of low-birth-weight births in the Inglewood Oil Field communities compared to Los Angeles County, after adjusting for race/ethnicity. There were differences in rates of low-birth-weight births among racial/ethnic groups with African

Americans having the highest rates of low-birth-weight births in the Inglewood Oil Field communities as well as in Los Angeles County. These disparities in low-birth-weight births represent another significant public health challenge throughout the county.

4.9.3 Birth Defects

For 28 of the 29 categories of birth defects, there was no statistically significant difference in the Inglewood Oil Field communities compared to Los Angeles County as a whole. Babies born in the Inglewood Oil Field communities between 1990 and 1997 were slightly more likely (1.2 times as likely) to be born with a limb defect compared to babies countywide. Limb defects are not known to be caused by exposure to petroleum products. Since multiple comparisons were made, the increase may be explained by statistical chance.

4.9.4 Cancer

The analysis found no evidence of elevated rates of acute myelogenous leukemia (AML), the type of cancer most definitively linked to petroleum products (benzene) or three of the other types of blood-related cancer for any of the race/ethnic groups examined. There was an excess risk of chronic myelogenous leukemia (CML) in non-Hispanic whites based on the occurrence of two cases above the expected number in 2000 through 2005. CML has not been consistently linked with exposure to petroleum products from oil fields or refineries. These two additional cases of CML may be explained by statistical chance, because the analysis examined multiple comparisons. Furthermore, in most of the studies examining this issue, occupational exposure to specific petroleum-based chemicals, such as benzene, was measured, rather than residential proximity to oil wells.

4.9.5 Community Survey

The community survey was developed to quantify self-reported illness and environmental concerns among residents living near the Inglewood Oil Field. The community was defined by a 1/5-mile buffer around the oil field and participants were randomly selected. Surveying was conducted by telephone in both English and Spanish. A total of 1,020 residents participated. The survey results were compared to a health survey conducted for all of Los Angeles County in 2007 to provide a comparison between those living in proximity to the Inglewood Oil Field and residents of the county as a whole. The results indicated that the prevalence of health conditions reported by respondents to the Inglewood Oil Field survey were similar to those in Los Angeles County, with the exception that more reported high blood pressure/hypertension in the area around the Inglewood Oil Field than in the Los Angeles County survey. The survey also found that the racial/ethnic disparities that exist in Los Angeles County were also reflected in the Inglewood Oil Field community (greater African Americans report hypertension and heart disease and the 46 percent of respondents in the Inglewood Oil Field community survey were African American compared to 9 percent in the Los Angeles County survey. Other issues addressed in the survey were smoking (13 percent of respondents reported smoking), eating fast food more than once per week (38 percent), and being obese or overweight (26 percent and 39 percent respectively).

With regard to the Inglewood Oil Field, participants were asked about the presence of offensive odors, illnesses caused by outdoor air pollution, and noise. Of the respondents, 86 percent did not notice any odors, and of those that did report odors, only 1.3 percent indicated concern that the odor was caused by the oil field. While 58 percent of respondents indicated concerns about air pollution, only 14.5 percent reported having an illness or symptom in the past year caused by pollution in the air outdoors. This percentage is slightly less than the percentage reported by Los Angeles County residents as a whole (17.5 percent). In regard to noise, participants were asked how much they were bothered by noise from six neighborhood sources: (1) cars and trucks, (2) airplanes, (3) garden equipment, (4) neighbors (including loud music, crying children, or barking dogs), (5) construction, and (6) oil field operations. Of the six sources, noise from the oil field was reported least frequently (LAC DPH 2012).

4.9.6 Health Assessment Limitations and Recommendations

The Health Assessment (LAC DPH 2011) noted limitations and a recommendation. The limitations were as follows:

- The analyses cannot confirm whether exposures to chemicals from oil drilling activities at the Inglewood Oil Field may be associated with a small increase in the risk of mortality, low-birth weight births, birth defects, or cancer among specific individuals living nearby, because epidemiological investigations of this type are more conclusive with larger sample sizes (more cases to analyze).
- The analyses do not take into account other important determinants of health such as behavioral risk factors (such as smoking and physical activity), social factors (such as community resilience, education, income, and access to health care) since these data were not available on the birth certificates, death records, or cancer registry records.
- The analysis cannot establish causal relationships between emissions from oil drilling activities and specific causes of death because of the lack of information on the individual levels of exposure to emissions that could establish dose-response curves and temporal relationships as well as the multitude of other risk factors that influence these disease outcomes. For example, a high-rate of mortality from asthma in the community adjacent to the Inglewood Oil Field would not prove that the oil field operations are causing asthma since there are many other potential causes, such as exposures to traffic-related air pollution, tobacco smoke, or adverse environmental conditions in the home. Alternatively, a normal or low rate of mortality from asthma would not prove that the Inglewood Oil Field is safe, again because of the many other factors that influence the rate. Thus, these results should be interpreted with caution.

All of the conclusions of the health study indicate that there is not a detectable relationship between the activities at the Inglewood Oil Field and the health of the surrounding community and that the occurrences of diseases of concern and mortality rates in the community are consistent with the rate of occurrences throughout the Los Angeles Basin. In other words, areas with no oil field operations were determined to have roughly the same mortality rate as the surveyed community around the Inglewood Oil Field. However, the report acknowledges that the data cannot determine whether there is a small effect, nor can the data address other health-related issues such as smoking, exercise, and social determinants of health. Because of these limitations, the Health Assessment recommends that local community health and safety would be more appropriately assessed by careful monitoring of the Inglewood Oil Field operations to ensure compliance with regulations and standards. In this regard, the CSD provides for Environmental Compliance Coordinator inspections and the annual Environmental Quality Assurance Program audit.

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Chapter 5

Regulatory Framework

5.1 Introduction: Local Source of Energy in the Context of Community Concerns

Since high-volume hydraulic fracturing was first used for shale gas development in the northeastern United States and tight sands development in the Intermountain West, there has been extensive coverage of controversies surrounding its use. Hydraulic fracturing has been called "the environmental issue of 2011" by Time magazine, was the subject of an HBO® movie ("Gasland"), and has been at the center of the debate regarding the pace at which the United States will move towards renewable sources of energy generation.

Although most of the news has been about the development of shale gas, tight sands and coalbed methane deposits, rather than the type of oil and natural gas development that occurs at the Inglewood Oil Field, community outreach conducted as part of this study (including one public meeting and open comment period) has indicated that many of the concerns surrounding shale gas development are shared by the local community. Questions and concerns submitted by the public to Los Angeles County for evaluation in this study were clearly influenced by media coverage of controversies in other parts of the country. As identified through the community outreach conducted by the County and other studies conducted on hydraulic fracturing operations in the U.S., the primary environmental and health issues of concern associated with hydraulic fracturing operations include:

- Potential for contamination of groundwater, including drinking water supplies;
- Potential for migration of gases and related explosion hazards;
- Environmental hazards associated with the chemical packages used during hydraulic fracturing operations;
- Potential for hydraulic fracturing operations to cause earthquakes;
- Issues related to well integrity; and,
- Air emissions and greenhouse gas emissions of hydraulic fracturing operations in comparison to regular oil field operations.

Many of these concerns are addressed by the existing regulatory framework. However, public concern has led to continuing efforts to expand the regulatory framework. This section summarizes the California regulatory framework as it pertains to oil and gas development, and hydraulic fracturing. The section then summarizes regulations and ongoing studies conducted by the Federal government, and many state governments. Finally, the regulatory framework specific to the Inglewood Oil Field, including the additional regulatory overlay of the CSD provisions, is considered in the context of state and federal regulations and guidelines.

5.2 Regulatory Framework and Government-Sponsored Reviews of Hydraulic Fracturing

The federal, state, and local laws, ordinances, regulations, and standards that govern oil field development throughout the United States mandate protection or mitigations against the potential environmental impacts of the entire development process. These protections include numerous provisions in the Clean Air Act, Clean Water Act, Safe Drinking Water Act, Endangered Species Act, and the Oil Pollution Act. Extensive California regulations and local provisions regulating nuisance also apply. The Inglewood Oil Field is unusual in that it has much greater regulation and oversight of its operations than most other onshore oil fields as a result of the Baldwin Hills CSD, which governs operations at the Inglewood Oil Field.

The widespread use of hydraulic fracturing since 1949 has been addressed through this extensive regulatory framework. Hydraulic fracturing is only one part of the entire oil and natural gas development process, and does not require, by itself, individual permits or approvals in California or most other oil and gas producing states. Instead, protections required for these resources during oil and gas development also apply to the use of hydraulic fracturing in general, as a completion technique.

Natural gas drilling activity brought hydraulic fracturing well completion techniques into public prominence. Shale gas production began and was first proven successful in the current oil and gas-producing states of Texas, Oklahoma, and the Intermountain region, and was viewed as a means to more securely achieve energy independence. USEPA reviews of high-volume hydraulic fracturing as used for coal-bed methane in 2004 found no justification for additional environmental controls (USEPA 2004). Significant national and public interest in the technique emerged however, when the shale gas production reached the Marcellus Shale in the northeast United States, especially Pennsylvania. The introduction and application of new technologies led to a dramatic and rapid increase in exploration activity. Communities largely unfamiliar with the oil and natural gas industry began seeing a large influx of drill rigs and production pumps, and construction of new well pads, access roads, and supporting infrastructure such as tanks and surface impoundments. This led to public concern that environmental issues were not being adequately addressed. That initial concern was primarily related to the policy of oilfield service companies to maintain confidentiality of the precise chemical names and concentrations used in hydraulic fracturing fluids. This information was considered a proprietary trade secret and oilfield service companies maintained that to reveal the information would put them at a competitive disadvantage. As a result, several states initiated independent reviews of the environmental impacts of hydraulic fracturing with an emphasis on water quality and chemical disclosure. The most comprehensive of these reviews was the Supplemental Generic Environmental Impact Statement (SGEIS) prepared by the State of New York in 2011, following release of a Generic Environmental Impact Statement which had not specifically regulated hydraulic fracturing operations as a specific action.

Growing public attention has also led the USEPA to allocate increased resources to studying the technique. In addition, the USEPA currently has two ongoing reviews, the first focused on the potential effects of hydraulic fracturing on drinking water supplies, and the second focused on the definition of "diesel fuel" as part of a review of the 2005 EPAct provisions. The 2005 EPAct recognizes hydraulic fracturing as a well completion process, and requires a UIC permit if the fluid used for hydraulic fracturing is diesel fuel. The USEPA is the federal agency tasked with implementing the underground injection control program; however, 42 states (including California) have primary enforcement and permitting responsibility under this program. In California, the Division of Oil, Gas, and Geothermal Resources (DOGGR) is the state agency that enforces the underground injection control program. The USEPA also recently released air quality rules relative to hydraulic fracturing.

Since the passage of the 2005 EPAct, many states have adopted regulations or passed legislation requiring operators to disclose the composition of the fluids used in the hydraulic fracturing process. In 2011, the U.S. Secretary of Energy convened a shale gas Production Subcommittee made up of university, agency, and NGO experts to address the expanded production of shale gas in a safe manner (www.shalegas.energy.gov). The committee concluded that shale gas can be developed in an environmentally responsible manner, and emphasized among other things the need for improved public information, improved coordination between shale gas developers and local, state, and federal government (including the STRONGER reviews described in this Section), as well as the kinds of protections that are in place at the Inglewood Oil Field through the CSD and other regulations and voluntary reporting.

The Inglewood Oil Field is not developing shale gas reserves, but primarily oil reserves with associated natural gas. As such, some specific regulations, studies, and concerns described in this section are not strictly applicable to the Inglewood Oil Field. However, they are described in this report because they give an important context to the questions and concerns that have been raised by the community.

This section addresses the current regulatory framework governing the use of hydraulic fracturing at the time of writing, and presents the results of various studies prepared by the federal government and by individual states.

5.3 California Regulations

5.3.1 DOGGR Regulations

DOGGR was formed in 1915 to regulate all oil and gas activities in the state of California with uniform laws and regulations. DOGGR supervises the drilling, operation, maintenance, and plugging and abandonment of onshore and offshore oil, gas, and geothermal wells. By regulating these activities DOGGR aims to prevent damage to: (1) life, health, property, and natural resources; (2) underground and surface waters suitable for irrigation or domestic use; and (3) oil, gas, and geothermal reservoirs.

DOGGR responsibilities are detailed in Section 3000 of the California Public Resources Code and Title 14, Chapter 4 of the California Code of Regulations. These regulations address issues such as well spacing, blow-out prevention devices, casing requirements, plugging and abandonment of wells, maintenance of facilities and safety systems, inspection frequency and reporting requirements. DOGGR programs also include: well permitting and testing; safety inspections; oversight of production and injection projects; environmental lease inspections; idle-well testing; inspecting oilfield tanks, pipelines, and sumps; hazardous and orphan well plugging and abandonment contracts; and subsidence monitoring.

California oil and gas regulations were reviewed in 1992 by the IOGCC and USEPA, at which time DOGGR made numerous changes to its program based on recommendations provided as part of the review. For example, DOGGR initiated the Idle-Well Management Program, which aims to reduce the number of long-term idle wells by encouraging operators to reactivate or plug and abandon their idle wells. In addition, DOGGR strengthened its requirements regarding well bonds and pipelines located in environmentally sensitive areas. The State Review of Oil and Natural Gas Environmental Regulations (STRONGER), a non-profit, multi-stakeholder organization that helps oil and natural gas producing states evaluate their environmental regulations associated with the exploration, development and production of crude oil and natural gas was formed in 1999, and reviewed California's regulations again in 2002. The STRONGER Review took note of California's stringent regulations on exploration and production waste management requirements and Underground Injection Control (UIC) programs. The Review included DOGGR's public participation and outreach, interagency coordination, abandoned well program, and data management proficiency. During the 2002 review the Stronger Review team did not address, nor offer recommendations for, hydraulic fracturing operations or regulations (STRONGER 2002).

While DOGGR's regulations do not include provisions specific to hydraulic fracturing, its broad authority over oil and gas operations gas and regulations encompasses the regulation of hydraulic fracturing in order to protect life, health, property, and natural resources including water supply (under Section 3106 of the Public Resources Code).

5.3.2 Baldwin Hills Community Standards District

The Baldwin Hills CSD, on Page 9, describes hydraulic fracturing (fracing) by including it in the definition of reworking, as follows:

> *"'Reworking' shall mean recompletion of an existing well and includes operations such as liner replacements, perforating, or fracing. Reworking also includes redrilling a well that is not deepened or sidetracked beyond the existing well bore."*

The CSD does not contain specific provisions which apply only to hydraulic fracturing. Rather, the CSD addresses all environmental aspects of oil field operation, and these aspects also apply to the potential environmental effects associated with hydraulic fracturing. These include analysis and provisions that address air quality, water quality, traffic, noise, and impacts to other environmental resource categories. The Baldwin Hills CSD also addresses seismic risk, contingency measures in the event of earthquakes including a requirement for an on-site accelerometer to measure effects of seismic activity and trigger contingency actions. The CSD also analyzes cumulative impacts, and environmental justice. The Baldwin Hills CSD, and the associated EIR, are incorporated by reference in to this Hydraulic Fracturing Study. The Hydraulic Fracturing Study does not identify a new impact not analyzed in the EIR, nor does it identify impacts greater in significance than those analyzed in the EIR.

5.3.3 Proposed California Regulations

As stated above, DOGGR does not currently regulate hydraulic fracturing specifically; it does not monitor hydraulic fracturing, nor are there reporting or permitting requirements. During legislative budget hearings held in Sacramento in March 2011, representatives from the California State Department of Conservation (DOC) testified that the agency would promulgate its own rulemaking process related to hydraulic fracturing. DOGGR hosted seven workshops between May and July 2012, to gather information as part of the rulemaking process. Two workshops were conducted in the Los Angeles Basin, one in the City of Culver City on June 12, 2012, and one in Long Beach on June 13, 2013. DOGGR plans to circulate the draft regulations in Fall 2012.

In addition to DOGGR's plans for rulemaking, two bills related to establishing new regulations for the practice of hydraulic fracturing were introduced in the California Legislature during the 2011–2012 legislative session, Assembly Bill 591 and Senate Bill 1054. The Legislature adjourned for the year without passing either measure.

California Assembly Bill 591, introduced in February 2011, would have required operators conducting hydraulic fracturing to disclose the chemical constituents of the fracturing fluid to DOGGR and the public, as well as the following additional information to DOGGR:

- the source and amount of water used in the exploration or production of the well;
- data on the use, recovery and disposal of any radiological components or tracers injected into the well; and
- if hydraulic fracturing is used, disclosure of the chemical information data described above.

California Senate Bill 1054 (Pavley), introduced in February 2012, would have required well owners or operators to notify surface property owners before commencing drilling operations and hydraulic fracturing operations near or below their property. The bill would have also required that notification be given to DOGGR, the appropriate RWQCB, water supplier, and municipal government. The bill would have also extended DOGGR's permit review time from the current 10 days to 15 days and required DOGGR to submit an annual report to the Legislature that includes the number of wells with notices, and an evaluation of compliance for the notification requirements.

5.4 Federal Regulations and Studies

5.4.1 Federal Regulations

Underground Injection Control Program

At the federal level, hydraulic fracturing is addressed under the Safe Drinking Water Act (SDWA), which was enacted in 1974. The SDWA gives USEPA's Office of Water the primary authority to protect drinking water. Under the UIC Program of the SDWA, USEPA is required to protect drinking water from contamination caused by underground injection of fluids. The UIC Program established six classes of injection wells that have purposes ranging from injection of hazardous materials and sewage, mining fluids, radioactive wastes, oil and gas fluids, and carbon dioxide (CO2) sequestration. Class II wells are associated with oil and natural gas production and include injection of:

- Fluids brought to the surface in connection with natural gas storage operations, or conventional oil or natural gas production (e.g., produced water);
- Fluids used for enhanced oil or natural gas recovery; and,

- Liquid hydrocarbons being stored, usually as part of the U.S. Strategic Petroleum Reserve. (USEPA 2011b).

As part of the 2005 EPAct, the U.S. Congress included hydraulic fracturing under the authority of the UIC Program when diesel fuels are used in the fracturing process (Paragraph 1 of Section 1421(d)). The EPAct did not provide a definition of diesel fuel. As of this writing, USEPA is conducting a review to develop a definition of diesel fuel. Interpretations of "diesel fuel" vary from only the use of 100 percent diesel fuel to the use of any diesel in a chemical package. The review process began in spring 2011 and draft guidance was issued in May 2012.

The Inglewood Oil Field does not use diesel fuel, in any amount, for hydraulic fracturing, and available records indicate that it was never used.

Chemical Disclosure

In 2009, the Fracturing Responsibility and Awareness of Chemicals Act was introduced in Congress. The legislation is commonly referred to as the FRAC Act. The FRAC Act proposes regulating hydraulic fracturing by requiring public disclosure of the chemical constituents used in hydraulic fracturing fluids. The FRAC Act states that proprietary information must be released in the event of a medical emergency. Congress did not take any action of the FRAC Act in the 111^{th} session of Congress, (2009 through 2011). The Act was re-introduced in the 112^{th} Congress in March 2011 (Lustgarten 2009). Since the FRAC Act, there have also been other bills discussed or introduced in Congress.

PXP posts the chemicals used in hydraulic fracturing on the public website FracFocus.org, as described below. This disclosure is consistent with the current regulations in the various states with disclosure laws.

EPA Regulation for VOC Reduction

On April 17, 2012, the USEPA released new regulations for reducing air pollution from hydraulic fracturing under the National Emissions Standard for Hazardous Air Pollutants (NESHAPS) for oil and natural gas production. The focus is on reduction of volatile organic compounds (VOCs) that are smog precursor compounds. The new regulations will take effect in two phases:

- **Phase 1:** Before January 1, 2014, use a combustion device (flare) or gas capture to reduce VOC emissions.
- **Phase 2:** Before January 1, 2015, capture all natural gas for sale. Exceptions are provided for exploratory wells or delineated wells used to determine or define the area of a natural gas reservoir, because exploratory wells are not near a pipeline and unable to bring gas to market, and for low pressure wells that cannot supply a gathering line.

The Inglewood Oil Field already exceeds the requirements of this new regulation through compliance with SCAQMD provisions as discussed in greater detail in Chapter 4.

5.4.2 Federal Studies

USEPA's Review of the Impacts of Hydraulic Fracturing on Drinking Water - 2004

In 2004, the USEPA conducted a study that analyzed the potential for contamination of underground sources of drinking water (USDW) caused by hydraulic fracturing of coalbed methane (CBM) natural gas wells. Like shale gas, CBM is an unconventional source of natural gas. Natural gas extraction wells are drilled into coal seams, the coal seam is dewatered by pumping, and natural gas then can desorb from the coal and be brought to the surface in the well. While not all CBM wells are completed by hydraulic fracturing, a portion of the wells do require the utilization of the technique. CBM resources tend to be at shallower depths than shale gas, and accordingly have a greater potential for affecting groundwater supplies if wells are not installed and abandoned according to current standards. The USEPA conducted this study in response to public concern that completing CBM wells by hydraulic fracturing had impacted the quality of groundwater, as well as by congressional need for additional data in the development of the 2005 EPAct. The USEPA released the results of the study in a report titled *Evaluation of Impacts to Underground Sources of Drinking Water by Hydraulic Fracturing of Coalbed Methane Reserves*.

The USEPA's 2004 study was two-fold. The first part was an extensive review of existing literature on the impacts of hydraulic fracturing on USDWs. The USEPA reviewed more than 200 peer reviewed publications and interviewed more than 50 employees in the natural gas industry, representatives of state and local agencies, and 40 concerned citizens and groups. The research focused on water quality incidents potentially associated with CBM hydraulic fracturing.

The second part of the study included a review of incidents of drinking water contamination thought to be associated with CBM hydraulic fracturing operations. The USEPA reviewed studies and investigations performed by state agencies in response to citizen complaints. Complaints investigated included: (a) drinking water with unpleasant taste and odor, (b) impacts to wildlife and vegetation, and (c) loss of water in wells and aquifers. After reviewing the data and incidents, the USEPA concluded that there were no conclusive links between water quality degradation in USDWs and hydraulic fracturing in nearby CBM wells, even though thousands of CBM wells annually were being hydraulically fractured.

The USEPA did determine that in some instances, the coal beds being produced were located within drinking water sources; that is, the coal beds were shallow enough to be within fresh water aquifers. In these cases, fluids and chemicals (including diesel fuels) used for hydraulic fracturing were introduced directly into drinking water sources, because the coal beds were located in drinking water sources. As a result of this finding, the USEPA entered into a Memorandum of Agreement in 2003 with three major service companies, which cumulatively perform 95 percent of the United States' hydraulic fracturing projects, to eliminate diesel fuel from the fracturing fluids that are injected directly into USDWs.

The 2004 USEPA study concluded that hydraulic fracturing fluids in CBM wells do not threaten USDWs. Based on this conclusion, the USEPA recommended against a Phase II study (USEPA 2004).

The Inglewood Oil Field does not use diesel for hydraulic fracturing or for high-rate gravel packs.

USEPA's Additional Review of Impacts of Hydraulic Fracturing – 2011

Continued technological advancements in the field of hydraulic fracturing and the application of the technology to tight sand and shale reservoirs has made the practice more prevalent since USEPA released its 2004 report. Public interest and concerns about the impact of hydraulic fracturing on human health and the environment have grown in direct proportion with increased media and internet attention to the practice. Concerns intensified when hydraulic fracturing was introduced in the Marcellus Shale in the northeastern states in approximately 2005. As a result of increased public interest, in fiscal year 2010 the U.S. Congress' Appropriation Conference Committee directed USEPA to conduct research to study the relationship between hydraulic fracturing and drinking water resources. The purpose of the study was to answer two overarching questions: (1) Can hydraulic fracturing impact drinking water resources, and, if so, (2) what conditions intensify these impacts?

In February 2011, the USEPA published a *Draft Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources*, with the objective of identifying the factors that have the potential to affect sources of drinking water. The study began with input from an External Science Advisory Board, which recommended that the study include:

- Use of lifecycle framework to identify important research questions;
- Direct initial research to sources and pathways of potential impacts of hydraulic fracturing on water resources, especially drinking water;
- Analysis of five to ten in-depth case studies at locations representing the full range of regional variability across the nation; and,
- Stakeholder engagement throughout the research process.

As the study focuses almost exclusively on water resources, USEPA examined how water was used during each stage of hydraulic fracturing operations and developed related fundamental research questions (Table 5-1).

To answer these questions, the USEPA study will use a combination of:

- retrospective case studies focusing on studying potential impacts where hydraulic fracturing has already occurred;
- prospective case studies focusing on sites where hydraulic fracturing will occur after research has begun so that site conditions can utilize monitoring before, during, and after hydraulic fracturing operations; and
- general scenario evaluations which will explore hypothetical situations related to hydraulic fracturing.

Table 5-1 Examination of Water Use During Hydraulic Fracturing

Water Use in Hydraulic Fracturing Stage	Research Questions
Water acquisition	How might large volume water withdrawals from ground and surface water resources impact drinking water resources?
Chemical mixing/site management	What are the possible impacts of releases of hydraulic fracturing fluids on drinking water resources?
Well construction and injection of fracturing fluids	What are the possible impacts from the injection and fracturing process on drinking water resources?
Flowback and produced water generation	What are the possible impacts of releases of flowback and produced water on drinking water resources?
Water treatment and waste disposal	What are the possible impacts of inadequate treatment or hydraulic fracturing wastewater on drinking water resources?

Source: USEPA 2011d

In each case, the research approach includes literature reviews, gathering and analyzing existing data, analytical methods, modeling/scenario evaluations, toxicity assessments, and stakeholder-suggested case studies. In addition, the USEPA will summarize the available data on chemical, physical, and toxicological properties of hydraulic fracturing fluid additives to better understand their effects and identify data gaps. The chemicals will also be compared to naturally occurring substances.

The USEPA's November 2011 Final Study Plan states that they have conducted an initial literature review, requested and received information from industry on chemicals and practices used in hydraulic fracturing, discussed initial plans for case studies with landowners and industry representatives, and conducted baseline sampling for retrospective case studies. An interim report is expected by the end of 2012 and is expected to contain a synthesis of results from the retroactive case studies and initial results from prospective case studies. A final report will be released in 2014, which will include results from the long-term prospective studies (USEPA 2011d).

The Inglewood Oil Field has very limited groundwater, no aquifers or water supplies, and is not located within or near an underground source of drinking water. The water supply for the nearby communities is derived primarily from the Colorado River and from Northern California with supplemental groundwater sources all located more than 1.5 miles from the oil field. As such, the results of the ongoing USEPA Study on the effects of hydraulic fracturing to drinking water supplies are not anticipated to produce results that are relevant to operations on the Inglewood Oil Field.

5.5 State Regulations and Studies

5.5.1 State-Specific Regulations

Hydraulic fracturing that includes diesel fuel is subject to the federal UIC program; this program is implemented by the California DOGGR, consistent with most states that have an oil and gas industry. The Inglewood Oil Field does not use diesel fuel for hydraulic fracturing.

Oil and gas producing states have regulated the practice by focusing on regulations specific to well bore integrity, well drilling and casing requirements, waste disposal, setback and operating requirements, and other conditions. The USDOE has stated that regulation at the state and local level allow laws to be tailored to the local environment, and state regulatory agencies tend to have the best information, knowledge and experience of the local conditions (USDOE et al. 2009).

State Chemical Disclosure Regulations

While regulations requiring the disclosure of the chemical constituents used in hydraulic fracturing fluids are under consideration at the federal level, several states have enacted regulations requiring chemical disclosure. Between 2010 and 2011, eight states passed chemical disclosure regulations. Wyoming was first to pass a regulation in September 2010, followed by Arkansas, Pennsylvania, Michigan, Texas, Montana, Colorado, Louisiana, and West Virginia. Ohio also amended existing laws in July 2012, to include disclosure of chemicals used in hydraulic fracturing. California, Illinois, and New Mexico have proposed rules, and several additional states appear to be considering rules.

Each state's regulations are generally consistent, although each differs slightly. For example, some regulations follow the federal proposal that companies can assert that information is proprietary. Other states require disclosure of propriety information to regulatory agencies but not the public. Table 5-2 summarizes the key provisions of the chemical disclosure regulations of the eight states that have enacted chemical disclosure laws.

Legislation was introduced in the 2011–2012 California legislative session that would have required operators conducting hydraulic fracturing to disclose the chemical constituents of the fracturing fluid to DOGGR and the public. In addition, a rulemaking process that will likely require disclosure of hydraulic fracturing chemicals is currently (as of the writing of this study) underway by DOGGR.

The Inglewood Oil Field voluntarily meets the requirements of most state chemical disclosure laws by posting the information on the publically-available website FracFocus.org, described below. As such, PXP would likely be in compliance with any reasonably contemplated chemical disclosure laws at either the State or Federal level.

Table 5-2 Summary of State Chemical Disclosure Regulations

State	Date Enacted	Enforced By	Reporting Required	Volume or Concentration Reporting Required	Proprietary Chemical Disclosure
Wyoming[a]	Sep-10	Wyoming Oil and Gas Conservation Commission	All chemicals used in hydraulic fracturing	Volume and concentration of the products are disclosed, but not of individual ingredients in chemical mixtures	Disclosed to regulators; undisclosed to the public.
Arkansas[a]	Jan-11	Arkansas Department of Environmental Quality	All chemicals used in hydraulic fracturing	None	Disclosure not required
Pennsylvania[a]	Feb-11	Bureau of Oil and Gas Management	All fracturing additives and chemicals All hazardous chemicals (as defined by OSHA) used on well-by-well basis	For hazardous chemicals only	Disclosure not required

Table 5-2 Summary of State Chemical Disclosure Regulations

State	Date Enacted	Enforced By	Reporting Required	Volume or Concentration Reporting Required	Proprietary Chemical Disclosure
Michigan[a]	Jun-11	Michigan Department of Environmental Quality	All hazardous chemicals (as defined by OSHA)	For hazardous chemicals only	Disclosure not required
Texas[a]	Dec-11	Texas Railroad Commission	All chemicals used in hydraulic fracturing.	For hazardous chemicals only	Disclosure not required
Montana[b]	Aug-11	Montana Board of Oil and Gas Conservation	All chemicals used on well-by-well basis	Concentrations of additives	Disclosure not required
Colorado[c]	Dec-11	Colorado Oil and Gas Conservation Commission	All chemicals used in hydraulic fracturing	Concentration of all chemicals	Requires disclosure of chemical family only
West Virginia[e]	Dec-11	West Virginia Department of Environmental Protection	All chemical used in hydraulic fracturing	None	Disclosure not required
Louisiana[d]	Oct-11	Louisiana Department of Natural Resources	All chemicals used in hydraulic fracturing	Concentration of all chemicals	Requires disclosure of chemical family only
Ohio[e]	Jul-12	Ohio Department of Natural Resources	All chemicals used in hydraulic fracturing	Volume and concentration of products	Disclosure not required

Sources: [a] Kusnetz 2011, [b] Falstad 2011, [c] Watson 2011, [d] Hall 2011, [e]NRDC 2012b

Self-Regulation by Industry

Disclosure of the chemical compounds used in hydraulic fracturing has been one of the primary issues that aroused public skepticism regarding the safety of hydraulic fracturing for development of shale gas in New York. As state and federal chemical disclosure laws were in development, the Groundwater Protection Council and Interstate Oil and Gas Compact Commission (IOGCC) in collaboration with the oil and natural gas industry, began examining methods to promote self-reporting and self-regulation to fill the gap and respond to public interest. This collaboration led to the development and launch of FracFocus (www.fracfocus.org) in 2011, a national hydraulic fracturing chemical registry.

FracFocus is managed by the Ground Water Protection Council and IOGCC. The mission of these organizations is conservation and environmental protection. The site was created to provide public access to reported chemicals used for hydraulic fracturing within an area of interest in a user-friendly interface. To help the public put the information into perspective, the site also provides objective information on hydraulic fracturing, the chemicals used, and the purposes they serve and the means by which groundwater is protected.

The high-volume hydraulic fracturing operations completed for this study were reported by PXP on FracFocus and are provided in Appendix B. PXP uses FracFocus on a company-wide basis for all shale oil and natural gas related hydraulic fracturing completions.

New York Supplemental Generic EIS

In 1992, New York State published a Generic Environmental Impact Statement on the Oil, Gas and Solution Mining Regulatory Program. In general, a Generic EIS (GEIS) is similar to a Programmatic Environmental Impact Report prepared pursuant to the California Environmental

Quality Act (CEQA). The New York GEIS analyzes the environmental impacts of the development of oil, natural gas, and solution mining resources, and provides options for mitigation. In this way, common or generic impacts of these activities can be considered "pre-evaluated," and if implemented in conformance with the mitigation conditions no further review is needed. Where site-specific actions differ from those evaluated in the GEIS, or where alternate mitigation is proposed, supplemental environmental review is conducted on those site-specific features.

New York's GEIS did not evaluate the effects of high-volume hydraulic fracturing as currently applied in the Marcellus Shale. As a result, the New York Department of Environmental Quality prepared a Supplemental Generic EIS (SGEIS) to study new technology and techniques related to hydraulic fracturing and to identify potential adverse impacts associated with the new technologies. The Draft SGEIS was released in September 2009 and in response to public comment, a Revised Draft was released in September 2011. The public comment period for the revised draft ended in January 2012, no further iterations have been released.

The draft SGEIS and the GEIS noted potential impacts, including water withdrawals, stormwater runoff, leaks and spills, and waste disposal. However, no adverse impacts to water resources were determined with regard to disposal of waste fluids, except that the disposal of flowback water could cause adverse impacts if not treated before disposal.

The draft SGEIS also finds that there is no significant impact to water resources likely to occur as a result of underground vertical migration of fracturing fluids through the shale formations, primarily the Marcellus Shale. The shale formations are vertically separated from the potential freshwater aquifers by at least 1,000 feet of sandstones and shales with low permeability. Furthermore, a supporting study for the draft SGEIS determined that it is highly unlikely that groundwater contamination would occur by fluids escaping from wellbores. The study notes that regulatory officials from 15 states recently testified that groundwater contamination as a result of high-volume hydraulic fracturing has not occurred (NYSDEC 2011).

In 2009, the New York City Department of Environmental Protection (NYCDEP) expanded upon the draft SGEIS, assessing potential impacts specific to the New York City water supply resulting from natural gas development in the Catskill and Delaware watersheds. Although, the Marcellus Shale in these areas has high gas production potential, these two watersheds provide 90 percent of New York City's water supply. The assessment concluded that there were potential cumulative impacts from a conceivable large-scale high-volume fracturing program in the Catskill and Delaware watersheds, which could substantially increase the risk to the New York City water supply (NYCDEP 2009). The overall recommendation of the NYCDEP study is that hydraulic fracturing not proceed, although it provides recommended mitigation measures if hydraulic fracturing does occur. However, in comparison to current and future activities to the Inglewood Oil Field, the NYCDEP assessment assumes a high density fracturing effort, which is much less than what has been done so far, and is proposed to occur, at the Inglewood Oil Field.

There are several relevant comparisons to be made between the conditions evaluated in the SGEIS, the NYCDEP study, and the Inglewood Oil Field. First, New York City's water supply, as well as much of the state, is derived from surface waters in relatively pristine areas and transported in pipes to New York City, and as such the water supply does not require filtration. The Inglewood Oil Field, however, has no surface water supplies within 10 miles, no groundwater supplies within

1.5 miles and no water transportation infrastructure. Ballona Creek is north and west of the field, but is not used for water supply. As is discussed in detail in Chapter 4, beneath the field itself, there is very limited groundwater, no aquifers or water supplies. The water supply for the nearby communities is derived primarily from the Colorado River and from Northern California; with supplemental groundwater sources all located more than 1.5 miles from the oil field (West Basin Municipal Water District 2011, Culver City 2010, City of Inglewood 2010, Golden State Water Company 2011, California American Water 2011b, see Figure 4-3b). There are no sources of groundwater supplying the City of Culver City system (Culver City 2010). The distance from water supply systems minimizes any potential risk of water quality contamination from fracturing. Note that even with this consideration, the SGEIS protections are still met by the Inglewood Oil Field, despite the much lower potential for water quality impacts.

Second, the Inglewood Oil Field has recently been the subject of a site-specific EIR, prompted by the proposal of the CSD by the County. Table 5-3 below provides a summary of the findings of the SGEIS and a comparison of its' suggested mitigation measures with the Baldwin Hills CSD regulations. In part because the CSD is site-specific, there is a greater amount and type of protections required compared to the SGEIS.

Table 5-3 Summary of Findings of the SGEIS and Comparison with Baldwin Hills CSD

Resource	SGEIS Impact	SGEIS Mitigation	Consistency with Inglewood Oil Field CSD
Water Resources	Depletion of water supply in streams	Passby Flow Requirements	No nearby streams for water supply, Cal Water indicates there is sufficient water supply to meet field needs
	Damage to groundwater resources	Pump testing and site-specific evaluation	Site specific monitoring and reporting; no local groundwater use and limited isolated occurrence of groundwater
	Water Contamination from stormwater runoff	State Pollution Discharge Elimination System (SPDES) permit with all associated requirements	National Pollutant Discharge Elimination System (NPDES permits and compliance with a site specific Construction Stormwater Pollution Prevention Plan are required
	Water Contamination from spills or hydraulic fracturing fluids in wellbores	Onsite reserve pits, blow-out preventers, secondary containment (dikes, pads, liners, sumps)	Secondary containment units, Catch Basins, containment berms, SPCC, NPDES permit
	Aquifer/Groundwater Contamination from Hydraulic Fracturing		Site specific monitoring and reporting; no local groundwater use and limited isolated occurrence of groundwater
	Contamination of soil/water from improper disposal or transportation of waste solids and fluids		BMPs required as part of SWPPP. BMPs include: operating procedures, and practices to control site runoff, spills/leaks, waste disposal
	Contamination of NYC unfiltered water supply	No hydraulic fracturing within 4,000 feet of these watersheds	Nothing comparable in California to NY unfiltered supply. No fracturing in Inglewood occurs closer than 1.5 miles (7,920 feet) from any water resource.

Table 5-3 Summary of Findings of the SGEIS and Comparison with Baldwin Hills CSD

Resource	SGEIS Impact	SGEIS Mitigation	Consistency with Inglewood Oil Field CSD
Freshwater Wetlands	Contamination from accidental release	Site-specific SEQRA review and required permits. Mandatory setbacks.	A system of catch basins around the field addresses potential for accidental releases. Regulated by NPDES program.
Ecosystems and Wildlife	Habitat fragmentation	BMPs to reduce habitat impacts, surveys and restrictions during mating migratory/mating seasons	Site specific special status species habitat protection plan and reporting Site specific Stormwater Pollution Prevention Plan
	Harm to populations due to habitat loss	Well pad reclamation	Site specific special status species habitat protection plan and reporting
	Invasive species	BMPs to reduce invasive species	Site specific special status species habitat protection plan and reporting Monitoring; restoration focused on removal of invasive species
Air Quality	Degradation of air quality	Technology standards, restrictions on sulfur content and BTEX class compounds, and public reporting	Implementation of Air Monitoring Plan. H_2S provisions. Extensive regulatory framework monitored and enforced by SCAQMD that exceeds NY requirements
GHG emissions	Increased GHG emissions due to drilling and production	GHG impacts mitigation plan	Compliance with AB 32 California Climate Change Act, GHG inventories Site specific Air Monitoring Plan
Naturally Occurring Radioactive Material	Exposure of workers and the public to harmful levels of radiation	Monitoring and testing requirements	CSD does not address because Naturally Occurring Radioactive Material does not occur onsite.
Visual	Temporary visual impacts from new structures	Visual impacts mitigation plan and design siting requirements	Landscaping Plan, Visual Site plan (removal of abandoned/unused equipment, licorice paint used on equipment)Screening landscaping, painting
Noise	Temporary Impacts from drilling and fracturing operations and traffic	Noise impacts mitigation plan	CSD Section E.5: Noise Attenuation (noise limits, back-up alarm restrictions, limited delivery hours, construction time limits, construction equipment requirements): Quiet Mode Drilling Plan including sound walls for drilling within 300 feet of residences.
Transportation	Increased traffic, damaged roadways	Road use agreements, transportation plan, road condition assessment	There is no anticipated impact/increase to traffic associated with future oil development; addressed in CSD EIR Inglewood Oil Field is an already developed, active oil field, and drilling is at existing pads to the extent feasible. Different from New York case where much of the development footprint is new.

Source: NYSDEC 2011, County of Los Angeles 2008

STRONGER Reviews

In 2007 there were 33 states with either oil or natural gas production, 27 of which cumulatively produce more than 99.9 percent of the United States' oil and natural gas (GWPC 2009). The State Review of Oil and Natural Gas Environmental Regulations (STRONGER) was formed in 1999, and is a non-profit, multi-stakeholder organization that helps oil and natural gas producing states evaluate their environmental regulations associated with the exploration, development and production of crude oil and natural gas. Prior to 1999, reviews were jointly conducted by the IOGCC and USEPA. The USEPA, USDOE and the America Petroleum Institute, among others, have provided funding to support the STRONGER reviews of the State regulatory review processes. In 2009, a Hydraulic Fracturing Workgroup was formed within STRONGER to address regulatory issues specific to hydraulic fracturing as a well completion technology. STRONGER reviews in 2011 have tended to focus on oil and gas regulations as they apply to hydraulic fracturing operations. STRONGER developed guidelines for hydraulic fracturing in 2010, and reviews of this process in different states generally follow these guidelines. The guidelines are not detailed, but set forth general guidelines such as:

- Wells should be properly designed and constructed.
- Water, wastewater, and waste management should be planned and conducted carefully.
- Information on the chemicals used in hydraulic fracturing operations should be disclosed and reported.

Four states were reviewed in 2011: Oklahoma, Ohio, Louisiana, and Colorado. Pennsylvania was reviewed in 2010 and Arkansas was reviewed in 2012. There was a STRONGER review for California in 2002, but the review did not address hydraulic fracturing. The STRONGER state review process is a non-regulatory program and relies on states to volunteer for reviews. Summaries of the findings for California and the states reviewed from 2010 to 2012 follow below.

California

California regulations were initially reviewed by IOGCC and USEPA in 1992, at which time DOGGR made numerous beneficial changes to its program. For example, DOGGR initiated the Idle-Well Management Program, which aims to reduce the number of long-term idle wells by encouraging operators to reactivate or plug and abandon their idle wells. In addition, DOGGR has strengthened its requirements regarding well bonds and pipelines located in environmentally sensitive areas. STRONGER reviewed California's regulations in 2002. The STRONGER Review Team commended DOGGR's changes since the initial IOGCC/USEPA review and also took note of California's stringent regulations on exploration and production waste management requirements and UIC programs. The Review included DOGGR's public participation and outreach, interagency coordination, abandoned well program, and data management proficiency. During the 2002 review the Stronger Review team did not address, nor offer recommendations for, hydraulic fracturing operations or regulations (STRONGER 2002). This 2002 Review indicated that as of 2000, California had approximately 207 operating oil fields and produced 840,000 barrels per day, ranking fourth in the United States for oil production. Today, California produces approximately 550,200 barrels per day with 209 active oil fields (DOGGR 2011).

Oklahoma

The Oklahoma Corporation Commission and the Oil and Gas Conservation Division (OGCD) regulate oil and gas operations in the state. Hydraulic fracturing has been conducted in Oklahoma for over sixty years. More than 100,000 wells have been hydraulically fractured during this time and state regulatory agencies have never identified an instance where hydraulic fracturing has adversely affected groundwater resources. Specifically, the OGCD mapped the base of fresh water throughout the state and determined that the oil and gas producing zones were all much deeper than the base of fresh water in nearly all well locations. Consequently, the OGCD determined that the risk of drinking water contamination resulting from hydraulic fracturing operations is limited. Nonetheless, in 2010, the OGCD recently incorporated new regulatory measures, such as well completion reports, and stricter requirements for the storage and recycling of flowback water, to address this limited risk. The state also initiated a five-year plan to help manage hydraulic fracturing (STRONGER 2011a).

Pennsylvania

The STRONGER review for Pennsylvania was conducted in 2010. Hydraulic fracturing is regulated by the Department of Environmental Protection (DEP) and the Bureau of Oil and Gas Management (BOGM). Hydraulic fracturing has been used in Pennsylvania since the 1950s and nearly all of the wells drilled since the 1980s have been hydraulically fractured. According to the DEP there are no verified instances of groundwater contamination resulting from hydraulic fracturing. Pennsylvania has more comprehensive regulations than Oklahoma, likely due to the increased interest in development of the Marcellus Shale. (STRONGER 2010) For example, in 2008, BOGM began requiring plans to manage water withdrawal and protect water quality standards.

As a result of the development of the Marcellus Shale, there have been several surface and subsurface water control issues, such as water withdrawal and wastewater management. Plans must indicate how much water will be withdrawn and from where, so that the DEP can ensure that excessive water withdrawals will not impact water quantity. Produced water recycling is encouraged, and there are strong regulations for fracturing waste generation, transportation and disposal. The 2010 film “Gasland” highlighted drinking water concerns in the town of Dimock, Pennsylvania. This topic is addressed in Chapter 4, including the results of recent USEPA sampling of local supply wells in Dimock that did not detect contaminants of concern.

Ohio

Oil and natural gas operations are overseen by the Ohio Department of Natural Resources (ODNR) and the Divisions of Mineral Resource Management (DMRM). Hydraulic fracturing has been performed since the 1950s and most wells drilled today are hydraulically fractured. The STRONGER review did not identify any instances of groundwater contamination as a result of hydraulic fracturing. After an independent review of its oil and gas program, the DMRM updated its regulations pertaining to hydraulic fracturing in June 2010.

STRONGER’s 2011 review indicated that Ohio has strong reporting requirements and enforcement tools. They also praised Ohio for thoroughly reviewing potential pathways for groundwater contamination and increasing staffing levels in the DMRM. In addition, STRONGER commended Ohio for thoroughly reviewing and revising oil and gas regulations in

2010. STRONGER provided several recommendations on how to strengthen existing regulations including ensuring that sufficient information is available with regard to the chemical constituents of fracturing fluids and thoroughly evaluating the need for and availability of surface and groundwater for hydraulic fracturing operations in the context of competing water uses. The overall finding was that the regulations protect water resources. More than 800,000 wells have been fractured in Ohio without any verified instances of groundwater contamination (STRONGER 2011b).

Louisiana

Hydraulic fracturing has been occurring in Louisiana since the 1960s, and is regulated by the Louisiana Office of Conservation. Currently, the Haynesville Shale is the primary area of interest for hydraulic fracturing operations in the state of Louisiana. This formation must be fractured to be commercially productive. Work permits are required prior to well construction operations, including hydraulic fracturing. Production in the Haynesville Shale began in 1910 and there are 1,586 Class II injection wells in this area. There have been no cases of contamination of underground sources of drinking water in this area. There are also rules related to the exploration and production of gas in the Haynesville Shale, including setback, noise, vibration, odor, lighting, venting and flaring requirements. Regulations for pressure testing of casing and cementing, as well as requirements for fracturing fluid flowback storage and disposal, also exist (STRONGER 2011c).

Colorado

The Colorado Oil and Gas Conservation Commission regulates oil and gas operations, including hydraulic fracturing, which has been occurring in Colorado since 1947. Nearly all active wells in Colorado have been hydraulically fractured, and no instances of ground water contamination have been confirmed. In 2007, Colorado comprehensively updated its oil and gas regulations, resulting in several new requirements related to hydraulic fracturing including but not limited to: (a) chemical inventories at well sites are required, (b) wells must be cased with steel pipes and surrounded by cement to prevent fluid and gas leakage, (c) surface casing to a specified minimum depth is required for well control and to protect shallow aquifers, (d) setbacks, baseline water quality sampling and other improved environmental protections, (e) baseline water well sampling is required, and (f) operators developing coal bed methane (CBM) wells must inspect local plugged and abandoned wells within one-quarter mile, sample adjacent water supply wells, and meet other requirements to minimize gas or water leakage. These new regulatory provisions were all commended by STRONGER (STRONGER 2011d).

Arkansas

STRONGER issued its review of Arkansas' oil and gas regulatory program in 2012. The Arkansas Oil and Gas Commission regulates the industry, and the review concluded that the Arkansas program is well managed and generally meets the 2010 Hydraulic Fracturing Guidelines. Program strengths included an update of oil and gas rules in 2004, in response to the increased activity in the Fayetteville Shale. In particular, Arkansas was among the first states in the nation to establish a system for the public disclosure of chemicals used in hydraulic fracturing operations. The program also has an effective water well complaint protocol, and a web site with information on hydraulic fracturing, including the areas of the Fayetteville Shale with active hydraulic fracturing of wells.

The Fayetteville Shale currently has 4,000 active oil and gas wells, with plans for development of over 14,000 more natural gas wells. Both the Arkansas Oil and Gas Commission and the Arkansas Department of Environmental Quality have responded to complaints of water well contamination within the Fayetteville Shale development area. To date, neither agency has found any evidence of water contamination from hydraulic fracturing in any of the water wells tested. In addition, the United States Geological Survey office in Little Rock has recently completed a water well testing program in Van Buren County, one of the most heavily drilled counties where hydraulic fracturing operations have occurred. No evidence of contamination from hydraulic fracturing has been found in the water wells tested.

STRONGER recommended that Arkansas require agency notification prior to commencing hydraulic fracturing operations. They also recommended increased funding and staffing of the Arkansas Oil and Gas Commission to allow for inspections (STRONGER 2012).

Summary

The STRONGER reviews have focused on regulatory programs, but in each state they also evaluated records with respect to contamination of underground sources of drinking water by hydraulic fracturing activity, hence the relevance to this study. The states reviewed in 2011 and 2012 have thousands of wells that had been hydraulically fractured. No evidence for groundwater contamination was found in any of these cases. Reviews had been conducted by state agencies, federal agencies, and the U.S. Geological Survey.

Table 5-4 summarizes the findings of the STRONGER reviews as they relate to hydraulic fracturing. The first column includes strengths of the regulatory program, and the second column includes recommendations. In general the comments refer to the regulatory programs themselves.

Although the recommendations apply more to regulatory agencies than to specific oil fields, where the recommendations are specific to the process of hydraulic fracturing, the Inglewood Oil Field would either meet or exceed these recommendations. In part this is because the Inglewood Oil Field operates under enhanced environmental controls that require notification, setbacks, and other provisions as required by existing regulations, primarily the CSD. In addition, PXP is voluntarily following chemical disclosure policies that meet those recommended by STRONGER.

Table 5-4 Summary of State STRONGER Reviews of Hydraulic Fracturing

Existing Regulation Strengths as summarized by STRONGER	Recommendations from STRONGER
Oklahoma	
▪ Comprehensive regulatory standards for hydraulic fracturing have been developed which: (see following cells)	▪ Reporting requirements should include volumes of hydraulic fracturing fluids and proppants used, pressures recorded, and hydraulic fracture materials used
▪ Prohibits pollution of a fresh water from well completion activities	▪ Recycling of flowback water and use of alternate, lower quality water should be encouraged
▪ Provides minimum casing and cementing standards	▪ More stringent regulations with regard to notification to the Oklahoma Corporation Commission prior to fracturing operations should be required

Table 5-4 Summary of State STRONGER Reviews of Hydraulic Fracturing

Existing Regulation Strengths as summarized by STRONGER	Recommendations from STRONGER
▪ Provides strong regulations related to the construction and maintenance of flowback water storage tanks ▪ Requires sampling of hydraulic fracturing waste materials or flowback water to monitor chemicals of concern, primarily salts and TDS	▪ The state should procure additional funding to ensure a staffing needs are met based on expected needs in the future
Pennsylvania	
▪ Comprehensive water planning process to ensure that demands on water resources related to hydraulic fracturing are managed through a planning process	▪ Stronger casing and cementing requirements have been proposed but have not been adopted into law
▪ Regulations encourage baseline groundwater quality sampling plans	▪ Encourage more comprehensive baseline studies in situations where there are increased risk factors
▪ Potential risks must be identified in a preparedness plan, which requires operators to list chemical additives used and wastes generated	▪ Require operators to identify potential conduits for fluid migration
▪ Waste characterization is required, including generation, transportation and disposal tracking	▪ Require notification prior to hydraulic fracturing. Currently this information is only transmitted via well completion reports and DEP does not have the opportunity to inspect
▪ Strong waste storage tank/pit requirements	▪ Secondary containment requirements for tanks used in hydraulic fracturing regulations
Ohio	
▪ Comprehensive well completion reporting is required and must include type and volume of fluid used for stimulation, reservoir breakdown pressure, recovered fluid containment methods, etc.	▪ Chemical disclosure regulations should be more comprehensive than currently exist
▪ Casing and cementing plans are required during the permitting process	▪ The state should evaluate the impact of hydraulic fracturing on surface and groundwater availability
▪ Notification is required before hydraulic fracturing occurs ▪ Well permits require a comprehensive review of potential pathways for groundwater contamination ▪ Pit placement and construction guidelines are implemented through permit conditions ▪ Strong enforcement tools	▪ Stricter spill notification regulations
Louisiana	
▪ The use of alternative water sources and the recycling of waste fluids are encouraged and promoted by recent legal amendments	▪ The minimum depth of surface casing is based on the total depth of the well. To protect groundwater, the depth to the USDW and depths of productive zones should also be considered
▪ Permitting of commercial waste fluid treatment and reclamation for hydraulic fracturing water supply purposes has been streamlined to make the process easier	▪ There are no cementing requirements for well construction or for casing weights or grades. Standards should be developed to meet anticipated pressures
▪ Increase in water source and volume reporting requirements, coupled with recycling provisions has significantly decreased water demand	▪ Reporting should include materials used, volumes of fracturing fluids, proppants used, and fracture pressures
▪ Surface water has been sufficiently analyzed and there is adequate water available for anticipated hydraulic fracturing needs	▪ Spill Prevention and Control Plans are currently required, but additional contingency plans are recommended

Table 5-4 Summary of State STRONGER Reviews of Hydraulic Fracturing

Existing Regulation Strengths as summarized by STRONGER	Recommendations from STRONGER
Colorado	
▪ Operators are required to keep chemical inventories at all well sites, which must be provided to agencies and health care providers upon request	▪ To help protect water resources from contamination, standards should be developed for minimum and maximum surface casing depths. All past problems related to surface casing in a hydraulically fractured well should be considered when developing this standard.
▪ Bradenhead annulus pressure during hydraulic fracturing operations must be measured and reported in an effort to help protect groundwater	▪ Materials used, aggregate volumes of fracturing fluids, proppants used and fracture pressures should be recorded
▪ Identification of potential pathways for fluid migration is required in certain circumstance	▪ An evaluation of naturally occurring radioactive material in hydraulic fracturing wastes should be required ▪ The availability of water resources for fracturing operations should be evaluated, as water supply is a significant issue in this arid region. Plans should be implemented to maximize water reuse and recycling if it is determine that water supply is an issue ▪ Requires operators to study and address potential pathways for fluid migration in more detail ▪ Stricter regulations related to providing notification and receiving approval prior to hydraulic fracturing
Arkansas	
▪ Since 2006, AOGC reviewed and revised numerous rules concerning environmental and production related concerns associated with hydraulic fracturing	▪ Notification prior to hydraulic fracturing so field inspectors can better monitor operations and related activities
▪ Developed water well complaint protocol, guiding staff towards efficient review and response to water well complaints and identification of laboratory analysis parameters	▪ Funding to continue support of Arkansas Department of Environmental Quality and seek resources to better Department
▪ AOGC's user friendly website informs public of hydraulic fracturing operations and other pertinent information regarding hydraulic fracturing	▪ Funding to increase AOGC Staffing Levels to ensure Commission inspection goals are met

Source: STRONGER 2010, 2011a-d, 2012.

NRDC's Evaluation of Hydraulic Fracturing Wastewater and Disclosure Regulations

In May 2012, the Natural Resources Defense Council (NRDC) published a report analyzing regulations related to wastewater generated from hydraulic fracturing. The report focuses on wastewater disposal methods and regulations in Pennsylvania but notes that the issues raised are relevant everywhere hydraulic fracturing occurs.

The report states that the most common management options for shale gas wastewater are recycling for continued use during oil and gas operations, treatment and discharge to surface waters, storage in impoundments and tanks, and applying it to the land (e.g. dust suppression). NRDC highlights environmental concerns associated with each disposal method, such as accidental spills when wastewater is temporarily stored in tanks or ponds on-site, inadequate treatment at publicly owned treatment facilities, or chemicals washing off roadways as a result of the land application method. Subsequently, NRDC recommends the following policy changes to strengthen regulations: regulate discharges from treatment plants more strictly; regulate hydraulic fracturing wastewater as a hazardous waste, either under RCRA or state regulations; only allow injecting of wastewater with hazardous characteristics into Class I hazardous waste wells, and strictly regulate Class II disposal wells in the interim; and prohibit land application and temporary storage in impoundments and tanks.

At the Inglewood Oilfield, produced water is transported by pipeline to the field's water treatment plant where it is mixed with other produced water generated at the field, treated, and reinjected into the oil and gas producing formations. This is in accordance with CSD Condition E.2.(i), which requires that all produced water is contained within closed systems at all time. NRDC notes that on-site recycling can have significant cost and environmental benefits by reducing freshwater consumption as well as the amount of wastewater requiring disposal. NRDC also notes that disposal by underground injection requires less treatment than other methods and creates the least risk of contaminating the environment. NRDC notes that this method can create risks of earthquakes and can require transportation over long distances, though in the case of the Inglewood Oilfield it is transported within the field boundary via pipeline and the existence of the waterflood operation significantly reduces concerns of induced seismicity because it injects water in to the depressurized zones of oil extraction.

NRDC encourages on-site wastewater recycling, the method used at the Inglewood Oilfield for beneficial reuse of its treated produced water, and does not identify any related policy recommendations directly pertaining to wastewater reuse other than noting that the benefits of reuse can sometimes be offset by the energy use and generation of concentrated residuals (NRDC 2012a).

As noted above, the focus of the NRDC report is primarily hydraulic fracturing in the Marcellus Shale and Pennsylvania regulations. In response to the report, in July 2012, the Secretary of Pennsylvania Department of Environmental Protection (DEP) issued a letter stating that "the Report is incorrect and inapplicable to Pennsylvania in many respects." The letter asserts that the report incorrectly characterizes wastewater disposal methods currently used in Pennsylvania and the associated regulations. The letter also mentions that report underestimates the quantity of wastewater that is recycled and indicates that the NRDC is biased against the industry (Pennsylvania DEP 2012). In turn, the NRDC issued a response letter defending the report and continuing to urge Pennsylvania DEP to strengthen their regulations.

In addition to assessing wastewater regulations, in a separate article published in July 2012, NRDC conducted a comparison of disclosure regulations for hydraulic fracturing between states related to advance public notice requirements prior to hydraulic fracturing; disclosure of information concerning the geological and environmental context of the wells, comprehensive disclosure about the hydraulic fracturing "treatment" (i.e. pressures, volume and type of base fluids, depths, etc.); and disclosure about the volume of wastewater created as well as its storage, treatment and/or disposal. The article points to lack of public access to disclosed information even when disclosure regulations do exist, and poor compliance with and enforcement of regulation (NRDC 2012b).

5.6 Inglewood Oil Field in State and National Regulatory Perspective

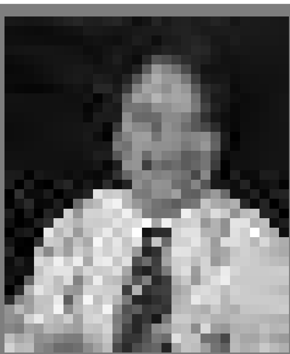
The federal, state, and local laws, ordinances, regulations, and standards that govern oil field development throughout the United States require protections against the potential environmental impacts of the entire development process. These protections range from provisions in the Clean Air Act, Clean Water Act, Safe Drinking Water Act, Endangered Species Act, and through extensive California regulation addressing air quality, water resources, biological resources, and cultural resources, and at the local level. The Inglewood Oil Field is unusual in that it has much greater regulation and oversight of its operations than most other onshore oil fields as a result of the County of Los Angeles CSD.

The Baldwin Hills CSD, and the associated EIR, together address most of the issues that are part of a hydraulic fracturing operation, such as truck traffic, water use, community compatibility (noise, light and glare, etc.), air quality, and other environmental resource categories. In addition, the EIR evaluates cumulative impacts, and environmental justice. These two documents are incorporated by reference into this Hydraulic Fracturing Study, which evaluates the effects measured and monitored during the high-volume hydraulic hydraulic fracturing and high rate gravel packing operations conducted in 2011 and 2012, as well as past activities of this type. The Hydraulic Fracturing Study did not identify a new impact not analyzed in the EIR, nor did it identify impacts greater in significance than those analyzed in the EIR.

Exacting protective measures and close monitoring are required by the Baldwin Hills CSD and by county, regional and federal agencies. These field-specific reviews and public and agency interactions compel PXP to enforce real-time compliance with all environmental standards in the Inglewood Oil Field. The long history of oil production in the area provides operators with an excellent understanding of the local subsurface conditions and reduces standard risks and uncertainties that would be present in new operations.

Chapter 6

Qualifications of Preparers



Current Position
Vice President,
Senior Principal

Discipline Areas
- Geology
- Geochemistry
- Hydrogeology
- Remediation Technology
- Fate and Transport Analysis
- CEQA/NEPA Environmental Impact Analysis
- Sediment Transport Analysis
- Civil Engineering

Education
- Ph.D., Geology and Geochemistry, Massachusetts Institute of Technology, 1989
- B.S., Civil Engineering and Geology, Stanford University, Stanford, 1983

Professional Registrations
- California Registered Geologist No. 5927

Appointed Positions
- National Academy of Sciences appointed Scientific Advisory Board – Giant Sequoia National Monument
- Natural Resources Management Department, former Executive in Residence – California Polytechnic University, San Luis Obispo
- UNESCO World Heritage Site Designation Advisory Council
- Volcanologist, RED Nacional de Emergencia, Chile

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Daniel R. Tormey, Ph.D., P.G.

Hydrogeologist, Geochemist, & Civil Engineer

Dr. Tormey is an expert in water and energy. He works with the environmental aspects of all types of energy and energy development, as well as water supply, water quality, hydrology, sediment transport, and groundwater-surfacewater interaction. He has well-developed skills in framing and analyzing environmental issues, and in communicating complex ideas to a wide range of audiences. Noted for the creativity of his approaches, he has conducted numerous studies related to the development of unconventional sources of natural gas, including high-volume hydraulic fracturing of oil and gas shales, and coal bed methane development.

Dr. Tormey has managed projects on behalf of government regulatory agencies (US Bureau of Land Management, US Forest Service, Federal Energy Regulatory Commission, US Bureau of Reclamation, US Army Corps of Engineers, US Office of Surface Mining, California State Lands Commission, California Public Utilities Commission, RWQCB-SD, local agencies) and for project proponents (PG&E, SCE, oil and gas companies, water agencies, among others). He has testified in the Federal Court of Claims on water rights and water takings issues. Dr. Tormey has managed the preparation of 15 Proponents Environmental Assessments for submission to the California Public Utilities Commission for projects including transmission lines and pipelines, and power plants fired by natural gas, oil, diesel, coal, and nuclear.

Dr. Tormey has evaluated environmental aspects and risks of oil and gas development and transport on many oil and gas fields in California and elsewhere in America, and more than 15 fields worldwide. He has led studies of the environmental impacts of hydraulic fracturing, water injection, and other oil and gas practices. He has evaluated beneficial reuse of produced water for agriculture, industrial, and restoration of habitat. He has studied carbon capture and storage using depleted oilfields as the storage reservoir, as well as the use of CO_2 for enhanced oil recovery. He has prepared environmental reports for pipelines carrying oil, natural gas, hydrogen, refined products, and biosolids. He has managed or been technical lead on offshore oil and gas projects, including licensing of eight liquefied natural gas (LNG) import terminals, marine terminals, and platforms (operation, abandonment, and reuse).   He has extensive experience in the preparation of environmental reviews supporting acquisition or divestiture of oil and gas producing facilities and related infrastructure.

He has worked on assessment, remediation, and restoration on many oilfields throughout the world, and is an expert on benchmarking and applying sound environmental solutions in that arena. He has pioneered bioremediation of oil-impacted soils, and has designed over 15 acres of such treatment cells. He has also considered the overall environmental setting (biological and cultural resources) of the oilfield in determining appropriate remediation responses.

Dr. Tormey has been technical lead for over two hundred projects requiring fate and transport analysis of chemicals in the environment, including modeling of chemicals in groundwater and surfacewater, study of linked groundwater-surfacewater systems, sediment transport analysis, quantification of adsorption/desorption kinetics, air dispersion modeling, among others. His work with contaminants also includes site assessment, forensic geochemistry, risk assessment, feasibility study, and site remediation. Dr. Tormey has served as a technical expert in fate and transport issues supporting either litigation and testimony in State Court, and agency testimony involving petroleum, solvents, metals, pesticides, and plastic components.

Dr. Tormey actively pursues volcanology research around the world, with a focus on interactions between geophysical variables that affect risk assessment, risk preparedness, and contingency planning.

Megan Schwartz
Senior Project Scientist

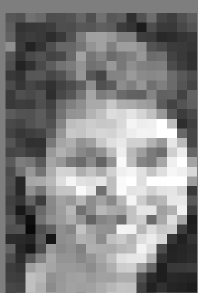
Current Position
Senior Project Scientist

Discipline Areas
> NEPA CEQA Planning
> Permitting
> Oil and Gas
> Renewable Energy
> Litigation Support
> Environmental Site Assessments

Education
> M.E.S.M. Environmental Science & Management, UC Santa Barbara, 2004
> B.A. Biological Anthropology, UC San Diego, 2002

Ms. Schwartz is a senior environmental planner and project manager with a Masters in Environmental Science and Management. She has addressed many controversial issues related to energy development in the southwestern United States and globally. Ms. Schwartz has addressed the potential impacts of proposed projects under both the California Environmental Quality Act (CEQA) and National Environmental Policy Act (NEPA). She also excels at regulatory compliance, including permitting and mitigation planning and implementation. She works with electrical local utilities and the oil and gas industry, as well as the regulatory agencies with jurisdiction over energy production.

Ms. Schwartz has worked on a variety of energy issues including hydroelectric power including dam removal on the Klamath River, oil and gas development around the world with a focus on water quality and community compatibility. She has also evaluated environmental effects of submarine cable installation, natural gas storage, transmission lines in the southwest and connecting to wind power in Mexico, and coal-fired power plants being either retired or repowered. Ms. Schwartz has been a technical lead on studies examining potential contamination of local groundwater supplies and beneficial reuse options of produced water from oil and gas fields. She has also evaluated the composition of additives used for hydraulic fracturing.



Molly Middaugh
Senior Staff Scientist

Current Position
Senior Staff Scientist

Discipline Areas
> Environmental Management, Permitting, and Compliance
> NEPA/CEQA
> Permitting
> Litigation Support
> Natural Resources Damages
> Environmental Communications
> Energy and Climate Change Policy

Education
> B.A. Environmental Analysis, Pomona College (Magna Cum Laude), 2010

Ms. Middaugh is a senior staff environmental scientist with a background in environmental science, economics, and policy. Her experience includes employment in the public, private and NGO sectors. Ms. Middaugh has worked on significant projects related to energy policy, with an emphasis on climate change, carbon offsets and international deforestation policy. She has a strong understanding of federal environmental regulations and policy that first developed when she worked for a member of the U.S. House of Representatives on Capitol Hill, as well as for the Energy and Climate Change program of the Center for Strategic and International Studies, also in Washington D.C.

Ms. Middaugh has evaluated the environmental impacts of energy and mining-related projects under the NEPA and CEQA process, as well as obtaining California and federal permits for these actions. She assists in analyzing and prioritizing environmental risks of oil and gas development. Ms. Middaugh also characterizes the chemical properties of and assesses beneficial reuses of produced water. She is currently engaged in the analysis of the chemical composition of fluids used as additives in hydraulic fracturing. She has also analyzed the environmental economic consequences in Natural Resource Damage claims.

Ms. Middaugh also works as part of the environmental communications team to translate technical scientific documents into more simple and understandable language.

Chapter 7

Supporting Material and References

7.1 Supporting Material

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Appendix A

Peer Reviewer Comment Letter

Summary of Peer Reviewer Comments for the Hydraulic Fracture Study: PXP Inglewood Oil Field

Prepared for

Rena Kambara
Regional Planning Assistant
Los Angeles County
Department of Regional Planning
320 West Temple Street, 13th floor,
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Prepared by

John P. Martin, Ph.D.
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Philadelphia, PA

October 3, 2012

FINAL REPORT

Context

In May, 2012, the authors of this report were selected by the County of Los Angeles (the "County") and Plains Exploration & Production Company ("PXP") as peer reviewers for the *Hydraulic Fracturing Study: PXP Inglewood Oil Field* prepared by Cardno-ENTRIX (the "consultant"). This report and peer review resulted from the Baldwin Hills CSD litigation Settlement Agreement.

Based on the direction provided in Term 13 of the Settlement Agreement, the objective of the peer review is to review the work of the independent consultant and provide advice to the consultant and a final evaluation to the County and PXP in order to provide an accurate and verifiable study:

> *The peer reviewer will be provided with access to all the data and materials provided to the independent expert. The peer reviewer shall agree to keep all proprietary information confidential. If the peer reviewer determines that the study is materially inadequate, incomplete or inaccurate, it shall so advise PXP's consultant who will complete the study as reasonably recommended by the peer reviewer and provide the revised study to the peer reviewer within 90 days. Upon acceptance by the peer reviewer, the study and all supporting material, including comments by the peer reviewer, shall be forwarded to the County, DOGGR, the Regional Water Quality Control Board ("RWQCB"), CAP and Petitioners and be available to the public, with any proprietary information redacted.*

In June, 2012, we began the process of reviewing the data used to develop the study, evaluating reports drafts and conducting a two day visit the site to get a perspective on the site context. What follows is a summary of the iterative process of review, advice and evaluation that led to the completion of the final study. Upon completion of the review, we both feel, based on information provided us and our own experience, that the report is adequate, complete and accurate and reflected thoughtful consideration for our comments and suggestions.

Process

We received the report draft of the study on July 14th, 2012 in compliance with the Settlement Agreement. We were also provided access to all data, research and reports used to assemble this draft at this time. We completed an initial review of the draft and the background material and developed comment and advice. These comments were communicated over a period of weeks rather than as one single response as we worked independently and then in coordination with each other and the contractor. This process allowed a number of key points to be refined based on effective criticism. The consultant responded by providing a revised draft for our review and advice. From this draft, we provided further advice and comment. The final study therefore reflects this iterative peer review process rather than a single review and response that typify journal peer review. Ultimately, the final report is more responsive to our input than may have been otherwise.

Throughout this process, we strived to offer thoughtful and timely input into the evaluation and provide advice to the consultant on ways to improve the study and ultimately to consider whether the report was materially adequate, complete and accurate. In this memo, we tried to be cognizant of this charge and summarize the process, advice and evaluation.

Evaluation of the Draft Reports

Overall Impression: For the first draft of the report, both of us concluded that the report's organization was overly complex which might have made it difficult for a general audience to understand. We found ourselves losing some key concepts due to the flow. We suggested that certain chapters be combined and that the flow reflect a time relationship. This request led the consultant to produce a second draft reflecting the reorganization. The number of chapters went from 11 to 7 in total. With this consolidation and reorganization, the second draft was far more understandable for the non-technical reader.

The major consolidations involved creating a chapter called "Environmental Effects Monitored in Conjunction with Hydraulic Fracturing Tests" by combining two chapters that split the environmental effects into two parts. Also, the overall discussion of hydraulic fracturing was taken from two chapters to one covering "past, present and future." Finally, a single regulatory chapter captured both the regulatory framework of various jurisdictions and public concerns with operations that were originally two separate chapters. This new report structure carries to the final version.

The study benefits from the tremendous amount of available data. Given the field's long history of production, there is a significant amount of data available to assess the geological and operating conditions at the field. In addition to the historic data, the current operator conducted two fully-monitored hydraulic fracturing operations and two high-rate gravel packs. Other available data includes a number of reports on geological characterization, environmental evaluations, potential community impacts, and regulations. The draft report did a reasonable job presenting a summary of this vast database. We offered some suggestions of ways to streamline the material presented to enhance readability.

Completeness: Both the first and revised draft were quite comprehensive and for the most part complete. A few areas found to be lacking in the first draft included the following:

1. Given the relevance of fluid migration to this topic, we felt that the consultant needed to expand the discussion of hydraulic connectivity given in the first draft. In the revised draft, the discussion was combined into one clear section discussing this important topic.
2. The first draft was lacking a truly representative geological cross section of the field along with a geologic map. A sequence of cross sections providing a better visualization and a geologic map of the field appeared in the revised draft.
3. Being that our experience is primarily in the northeastern USA and Canada, we thought that a better discussion was needed on how this field compares to those outside of California where many of these issues have arisen.
4. Many figures lacked scales, adequate figure captions, and legends. In the revised draft, these items were improved. Annotation was added to the remaining figures for the final report.
5. We suggested a revised discussion of induced seismicity to include the relevance of the 1983 Coalinga earthquake to potential hydraulic fracturing in the, deeper thrust-faulted, pre-Pliocene units of the Inglewood field in light of similar geologic settings and regional stress regimes. This was included in section 4.5.5.

Also, there were a few key resources were not included in the draft that, if included, could help the reader understand the issues involved with hydraulic fracturing including Frolich 2012, Myers 2012, NRC earthquake study 2012, USGS 2012, and Warner 2012. Most of our suggested references were incorporated into the revised draft and many of these references were used to respond to our concerns above.

Adequacy: As mentioned, the draft report was extremely comprehensive. There were a few areas that we did not feel were completely adequate:

1. The diagrams showing the geologic structure at the field are difficult to understand and interpret. We requested that these should be redone or the text should be expanded to explain the visual. The final draft accomplishes this.
2. The oil fields of California and this field particularly are not similar to the fields of either Texas or the northeast. This distinction needed to be adequately explained to put the discussion of environmental issues in context. The revised draft did a much better job of this.
3. The regulatory section initially did not focus on the key elements of California in a way that made it clear to us how operations are regulated. The revised draft

combined sections and brought California front and center making this much easier to follow.

Accuracy: We did not find any major inaccuracies in either draft though there were some specific statements that were either inaccurate or contradictory and needed revision. Also, there were a few statements that lacked supporting evidence and could be questioned for accuracy. We requested that these statements be corrected or supporting evidence be provided. In response to our comments the author corrected statement in revised draft or provided a rationale for leaving the text as is that satisfied our inquiry.

Topic-Specific Report Comments

In addition to the numerous typographical edits and suggestions, there were some topic-specific comments that we spent considerable time reviewing. These topics include geology, induced fractures, seismicity, environmental issues, and regulation.

Geology: We had some issues with the readability of the diagrams for non-technical audiences. The first draft included one diagram which did not convey the complexity of the field as described by the work of Elliot and others (2009). The final version includes all three of their cross sections and the description. This helps to explain the geological conditions in the field much better and helps put the 3D visualizations in context. Blade-like features in the subsurface of several of the diagrams were unexplained and easily misinterpreted as fracture orientation. These features actually represent the distribution of proppant for the hydraulic fracturing stages. The final draft clarifies this with better legends.

Induced Hydraulic Fractures: We had some trouble understanding the discussion of the height and orientation of the induced hydraulic fractures from the two Nodular Shale tests. We suggested clarification of height of induced fractures to read height of zone of induced fractures. This change was made in the final draft.

Seismicity: In addition to the comment on page three, we requested some discussion of how the operator designs the surface infrastructure for hydraulic fracturing to mitigate the impact of a seismic event similar to what occurred at Coalinga. The final draft includes a description of the required standards for structures and other surface equipment.

Environmental Issues: The report covered the environmental issues typically identified with hydraulic fracturing but these issues were spread among a few chapters. We suggested that they be combined into one chapter.

One issue that we commented on was the potential for fluid migration. The first draft did approach this topic but we felt that a clearer description needed to be included regarding why the geology limited fluid movement. In the revised report, the information presented and the flow of the discussion better explains why the lack of hydraulic connectivity minimizes the potential for fluid migration off of the site.

Since air emissions releases are of concern, the air quality section needs to be as comprehensive as possible. The draft report covered this topic well in three different chapters. We suggested that this might be more effective if most of this information could be condensed into the environmental effects chapter. This was accomplished in the reorganization of the revised draft.

There are a number of issues with hydraulic fracturing that are actually common to any oilfield operation regardless of the completion method. This includes many of the community impacts such as traffic and noise. The Baldwin Hills CSD EIR covers many of these common issues. Though we did not suggest that this study repeat the contents of the EIR, we felt that some reference should refer the reader to the appropriate EIR documents.

Regulation: The first draft covered regulations but we felt that the section should be reorganized so California regulation was covered first and that comparisons with other jurisdictions be made to California. Upon revision, the new regulation section accomplishes this effectively. The comparison table with the New York SGEIS is particularly useful to identify specific issues such as spill containment that were identified as important issues in the New York process.

Concluding Comments and Final Report Evaluation

Through the iterative review process, our comments, questions and criticisms were integrated into study in ways that, we feel, improved the final product. As peer reviewers, it is not our charge to become coauthors but to offer suggestions for improvement based on our expertise. In the end, the work remains that of the authors.

As the endpoint of the peer review process, the County and PXP has asked us to make a determination as specified in the Settlement Agreement:

> "If the peer reviewer determines that the study is materially inadequate, incomplete or inaccurate, it shall so advise PXP's consultant"

On September 30, 2012 we received from the consultant the final report for review and acceptance. Upon review, we both feel, based on information provided us and our own experience, that the report is adequate, complete and accurate and reflected thoughtful consideration for our comments and suggestions. This document serves as our final advice to the consultant, the County and PXP.

The Reviewers:

John P. Martin, Ph.D.

John is the founder of JPMartin Energy Strategy LLC which provides strategic planning, resource evaluation, project management and government/public relations services to the energy industry, academic institutions and governments. Prior to forming JPMartin Energy Strategy LLC in 2011, John spent 17 years working on energy research and policy issues at the New York State Energy Research and Development Authority and developed a series of projects targeting oil and gas resources, renewable energy development and environmental mitigation. He currently serves on the USDOE's Unconventional Resources Technical Advisory Committee. While at NYSERDA, he co-directed the Governor's Carbon Capture and Sequestration (CCS) Working Group, an interagency committee organized in 2007 to address CCS issues and served as point person on a series of technical studies looking at all aspects of hydraulic fracturing and multiwell pad development. John regularly lectures and publishes on such diverse topics as the shale resources development, carbon capture and sequestration, compressed-air energy storage, renewable energy resource development, and research policy. Prior to joining NYSERDA, he worked in academia, consulting and regional planning. He holds a Ph.D. in Urban and Environmental Studies, an M.S. in Economics and a B.S. in Geology, all from Rensselaer Polytechnic Institute. He also holds an M.B.A. from Miami University and completed graduate work in mineral economics at West Virginia University.

Peter D. Muller, Ph.D., C.P.G.

Independent consulting geologist specializing in structural geology, geologic mapping, and geologic data analysis. Presently researching subsurface migration of fluids in the northern Appalachian Basin and the relationship to hydraulic fracturing. Senior Geologist with Alpha Geoscience (2010-2012) concentrating on shale gas development in NY and PA. Professor of geology at the State University of New York at Oneonta (1983-2009; Chair 1999-2003) teaching courses in structural geology, map and field geology, engineering geology, mineral resources, waste management, physical geology, and environmental science. Worked as a staff geologist for Dames and Moore Consultants (1973-1975) and the Maryland Geological Survey (1980-1982). Peter received his BS in geology from Bucknell University (1971) and PhD in geology from Binghamton University (1980). He has extensive geological field experience in a wide range of settings, both domestic and international, and has published peer-reviewed research (articles and maps) on the structure, petrology, and tectonics of the Maryland Piedmont, the Adirondack Mountains of New York, and the Ruby and Blacktail ranges of southwest Montana.

Appendix B

Chemical Additives Used and FracFocus Reports

Hydraulic Fracturing Fluid Product Component Information Disclosure

Fracture Date	9/15/2011
State:	California
County:	Los Angeles
API Number:	0403726720
Operator Name:	Plains Exploration & Production
Well Name and Number:	VIC 1-330
Longitude:	-118.379139976
Latitude:	34.006457093
Long/Lat Projection:	NAD83
Production Type:	Oil
True Vertical Depth (TVD):	8,030
Total Water Volume (gal)*:	168,210

Hydraulic Fracturing Fluid Composition:

Trade Name	Supplier	Purpose	Ingredients	Chemical Abstract Service Number (CAS #)	Maximum Ingredient Concentration in Additive (% by mass)**	Maximum Ingredient Concentration in HF Fluid (% by mass)**	Comments
7% KCL Water	Operator				100.00%	86.77644%	Density = 8.700
SAND - PREMIUM WHITE	Halliburton	Proppant	Crystalline Silica, Quartz	14808-60-7	100.00%	3.70605%	
PRC SAND	Halliburton	Proppant	Crystalline Silica, Quartz	14808-60-7	100.00%	8.59803%	
			Hexamethylenetetramine	1009-7-0	2.00%	0.17196%	
			Phenol / Formaldehyde Resin	900303-35-4	5.00%	0.42990%	
SSA-2	Halliburton	Sand	Crystalline Silica, Quartz	14808-60-7	100.00%	0.35578%	
FR-66	Halliburton	Friction Reducer	Hydrotreated Light Petroleum Distillate	64742-47-8	30.00%	0.01335%	
LOSURF-300M™	Halliburton	Surfactant	1,2,4 Trimethylbenzene	95-63-6	1.00%	0.00079%	
			Ethanol	64-17-5	60.00%	0.04763%	
			Heavy Aromatic Petroleum Naphtha	64742-94-5	30.00%	0.02382%	
			Naphthalene	91-20-3	1.00%	0.00079%	
			Poly(oxy-1,2-Ethanediyl), Alpha-(4-Nonylphenyl)-Omega-Hydroxy -,Branched	127087-87-0	10.00%	0.00794%	
CL-28M CROSSLINKER	Halliburton	Crosslinker	Crystalline Silica, Quartz	14808-60-7	5.00%	0.00249%	
			Borate Salts	Proprietary	60.00%	0.02989%	
MO-67	Halliburton	Buffer	Sodium Hydroxide	1310-73-2	30.00%	0.00283%	
BA-40L BUFFERING AGENT	Halliburton	Buffer	Potassium Carbonate	584-08-7	60.00%	0.03990%	
FE-1A ACIDIZING COMPOSITION	Halliburton	Misc Additive	Acetic Acid	64-19-7	60.00%	0.00255%	
			Acetic Anhydride	108-24-7	100.00%	0.00425%	
K-38	Halliburton	Crosslinker	Disodium Octaborate Tetrahydrate	12008-41-2	100.00%	0.02099%	
LGC-36 UC	Halliburton	Gelling Agent	Guar Gum	9000-30-0	60.00%	0.16582%	
			Naphtha, Hydrotreated Heavy	64742-48-9	60.00%	0.16582%	
BE-3S BACTERICIDE	Halliburton	Biocide	2,2 Dibromo-3-Nitrilopropionamide	10222-01-2	100.00%	0.00119%	
			2-Monobromo-3-Nitrilopropionamide	1113-55-9	5.00%	0.00006%	
OPTIFLO-III DELAYED RELEASE BREAKER	Halliburton	Breaker	Ammonium Persulfate	7727-54-0	100.00%	0.00889%	
			Crystalline Silica, Quartz	14808-60-7	30.00%	0.00267%	
SP BREAKER	Halliburton	Breaker	Sodium Persulfate	7775-27-1	100.00%	0.00237%	

* Total Water Volume sources may include fresh water, produced water, and/or recycled water

** Information is based on the maximum potential for concentration and thus the total may be over 100%

All component information listed was obtained from the supplier's Material Safety Data Sheets (MSDS). As such, the Operator is not responsible for inaccurate and/or incomplete information. Any questions regarding the content of the MSDS should be directed to the supplier who provided it. The Occupational Safety and Health Administration's (OSHA) regulations govern the criteria for the disclosure of this information. Please note that Federal Law protects "proprietary", "trade secret", and "confidential business information" and the criteria for how this information is reported on an MSDS is subject to 29 CFR 1910.1200(i) and Appendix D.

Hydraulic Fracturing Fluid Product Component Information Disclosure

Fracture Date	1/5/2012
State:	California
County:	Los Angeles
API Number:	0403726421
Operator Name:	Plains Exploration & Production
Well Name and Number:	VIC1 635
Longitude:	-118.3771225
Latitude:	34.00234951
Long/Lat Projection:	NAD83
Production Type:	Oil
True Vertical Depth (TVD):	8,430
Total Water Volume (gal)*:	125,248

Hydraulic Fracturing Fluid Composition:

Trade Name	Supplier	Purpose	Ingredients	Chemical Abstract Service Number (CAS #)	Maximum Ingredient Concentration in Additive (% by mass)**	Maximum Ingredient Concentration in HF Fluid (% by mass)**	Comments
7% KCL Water	Operator				100.00%	81.77644%	Density = 8.700
SAND - COMMON WHITE	Halliburton	Proppant	Crystalline silica, quartz	14808-60-7	100.00%	1.42615%	
SAND - PREMIUM WHITE	Halliburton	Proppant	Crystalline silica, quartz	14808-60-7	100.00%	4.30405%	
CRC SAND	Halliburton	Proppant	Crystalline silica, quartz	14808-60-7	100.00%	9.98805%	
			Hexamethylenetetramine	1009-7-0	2.00%	0.19976%	
			Phenol / formaldehyde resin	900303-35-4	5.00%	0.49940%	
LOSURF-300M™	Halliburton	Surfactant	1,2,4 Trimethylbenzene	95-63-6	1.00%	0.00079%	
			Ethanol	64-17-5	60.00%	0.04763%	
			Heavy aromatic petroleum naphtha	64742-94-5	30.00%	0.02382%	
			Naphthalene	91-20-3	1.00%	0.00079%	
			Poly(oxy-1,2-ethanediyl), alpha-(4-nonylphenyl)-omega-hydroxy-, branched	127087-87-0	10.00%	0.00794%	
K-38	Halliburton	Crosslinker	Disodium octaborate tetrahydrate	12008-41-2	100.00%	0.02099%	
FR-66	Halliburton	Friction Reducer	Hydrotreated light petroleum distillate	64742-47-8	30.00%	0.01335%	
SandWedge® NT	Halliburton	Conductivity Enhancer	Dipropylene glycol monomethyl ether	34590-94-8	60.00%	0.29738%	
			Heavy aromatic petroleum naphtha	64742-94-5	10.00%	0.04956%	
BA-40L BUFFERING AGENT	Halliburton	Buffer	Potassium carbonate	584-08-7	60.00%	0.03990%	
CL-28M CROSSLINKER	Halliburton	Crosslinker	Crystalline silica, quartz	14808-60-7	5.00%	0.00250%	
			Borate salts	Confidential Business Information	60.00%	0.02995%	
FE-1A ACIDIZING COMPOSITION	Halliburton	Misc Additive	Acetic acid	64-19-7	60.00%	0.01278%	
			Acetic anhydride	108-24-7	100.00%	0.02130%	

LGC-36 UC	Halliburton	Gelling Agent	Guar gum	9000-30-0	60.00%	0.16582%	
			Naphtha, hydrotreated heavy	64742-48-9	60.00%	0.16582%	
MO-67	Halliburton	Buffer	Sodium hydroxide	1310-73-2	30.00%	0.00283%	
BE-3S BACTERICIDE	Halliburton	Biocide	2,2 Dibromo-3-nitrilopropionamide	10222-01-2	100.00%	0.00285%	
			2-Monobromo-3-nitrilopropionamide	1113-55-9	5.00%	0.00014%	
K-38	Halliburton	Crosslinker	Disodium octaborate tetrahydrate	12008-41-2	100.00%	0.01902%	
SP BREAKER	Halliburton	Breaker	Sodium persulfate	7775-27-1	100.00%	0.01949%	

* Total Water Volume sources may include fresh water, produced water, and/or recycled water

** Information is based on the maximum potential for concentration and thus the total may be over 100%

Ingredient information for chemicals subject to 29 CFR 1910.1200(i) and Appendix D are obtained from suppliers Material Safety Data Sheets (MSDS)

PAVILLION, WYOMING AREA DOMESTIC WATER WELLS DRAFT FINAL REPORT AND PALATABILITY STUDY

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and

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APPENDICES

A [illegible]

B Sampling and Analysis Plan (SAP)

C Quality Assurance Project Plan (QAPP)

D Health and Safety Plan (HASP)

E [illegible]

F [illegible]

G [illegible]

H [illegible]

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J [illegible]

K [illegible]

L [illegible]

M Laboratory Analytical Reports and Chain of Custody Records

N [illegible]

O Data Validation Reports

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TABLES

EXECUTIVE SUMMARY

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1 INTRODUCTION AND SCOPE OF WORK

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1.1 Rationale for Conducting Groundwater Investigation

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1.2 Summary of Previous Investigations and Response Actions

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[illegible] Objectives

The objectives of this Study were to:

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1.4 Scope of Work

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The objectives of the [illegible] were to [illegible] factors that could potentially affect water quality, including but not limited to the following:

- Well construction and integrity
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[illegible] Appendix [illegible]

1.4.4 [illegible]

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1.4.5 Review of Other Documents

[illegible]

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[illegible] Appendix [illegible]

1. Geology and Gas Resources
2. Groundwater Resources
3. Previous Groundwater Investigations

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2 BACKGROUND

2.1 Site Location and Description

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2.2 Land Use

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2.3 Geology

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2. 2014 GROUNDWATER INVESTIGATION METHODOLOGY

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2.1 Water Supply Well Sampling

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2.1.1 Selection of Sampling Locations

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3.1.3 [illegible]

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3.1.4 [illegible]

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3.1.4 Field Parameter Measurements

[illegible]

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3.1.5 Field Quality Control Sampling

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3.1.6 Sample Analysis, Preservation and Handling

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3.1.7 Data Validation and Reporting

The samples were analyzed by [illegible] Laboratories in [illegible], California; [illegible] Laboratories, Inc., in [illegible]; and [illegible] Laboratories in [illegible], Wyoming. [illegible]

- [illegible] Laboratories, Inc., [illegible], California
- [illegible] Laboratories, [illegible], South Carolina
- [illegible] Laboratories, Inc., [illegible], Ohio
- [illegible], Inc., [illegible], Wyoming
- [illegible] Laboratories, Inc., City of Industry, California

The reports from all [illegible] laboratories were reviewed for [illegible]. Data validation review was performed by [illegible], California, [illegible]. [illegible]

The quality control [illegible]. [illegible] assigned qualifier flags to the data as follows:

- J - The analyte was positively identified, but the reported numerical value is an approximate concentration of the analyte in the sample.
- [illegible] - [illegible] additional information.
- U - The analyte was analyzed for, but was not detected at or above the reported sample quantitation limit.
- UJ - The analyte was not detected at or above the reported sample quantitation limit and the reported quantitation limit is approximate and may or may not represent the actual limit [illegible] the analyte in the sample.

- [illegible]

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3.2 Downhole Video Surveys and Well Owner Questionnaires

3.2.1 Downhole Video Surveys

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[illegible] RESULTS OF [illegible] GROUNDWATER INVESTIGATION WITH SUMMARY OF PRIOR SAMPLING RESULTS

[illegible] Water-Supply Well Construction and Use

[illegible] Water-Supply Well Construction and Condition

[illegible]

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[illegible] Surrounding Land Use

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4.1.1 Water Supply Well Sample Representativeness

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4.2 Water Supply Well Sampling

4.2.1 June and August 2012 Groundwater Sampling

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- [illegible] (U.S. EPA, 2012).

- U.S. EPA Drinking Water Advisory [illegible] (U.S. EPA, [illegible]).

In the tables of laboratory analytical results, when the value of an analytical result is less than the method detection limit (MDL) or the [illegible] limit (RL), [illegible]. The MDL and RL are defined below:

- MDL: The minimum concentration of a substance that can be measured and reported with 99 percent confidence that the analyte concentration is greater than zero; it is determined from analysis of a sample [illegible] containing the analyte.

- RL: The RL is greater than the MDL. It is laboratory specific and is [illegible]. The RL is [illegible] in the quantification of the sample concentration.

When the concentration of an analyte is between the MDL and the RL, there is confidence that the analyte is present but the actual concentration is [illegible]. These concentrations reported between the MDL and RL are [illegible] a "J" flag.

In the tables of [illegible] of field water-quality parameters and laboratory analytical results, when a reported concentration is greater than a drinking water standard or comparison value, the result value is highlighted [illegible]. In accordance with notes in the tables, [illegible] result is also highlighted.

4.3.2 Historical Groundwater Sampling Events

This section includes [illegible] from previous sampling events. These include historical sampling results from 1979 to 2008, [illegible] from 2009 through 2011 [illegible].

Previous sampling events (from 1979 to 2008) include the following:

- LGRC for the well owners following installation in 1996

- [illegible], on behalf of [illegible], from September 2006 through October 2008. The October 2008 sampling was performed [illegible]

[illegible]

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- Phase I: March 2009
- Phase II: January 2010
- Phase III: October 2010
- Phase IV: April 2011

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4.2.3 Groundwater Elevations

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4.2.3 Trace Metals

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[illegible] Dissolved Gases

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[illegible] Organic Analytes, Summary of Detections

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4.4 Results from Prior Sampling Events, Review of Data Quality and Report Completeness

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5. EVALUATION AND DISCUSSION

5.1 Gas Well Integrity

[illegible]

5.1.1 Potential for Movement of Gas or Other Fluids from Hydraulically Fractured Production Intervals

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5.1.2 Potential for Interaction with Adjacent Wellbores

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[illegible] Gas Well Pits as Potential Sources of Water Quality Impacts

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[illegible] Chemical Characteristics of Local Groundwater

[illegible] Chemical Characteristics of Groundwater [illegible]

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[illegible] Groundwater Quality Variation

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[illegible] Groundwater Quality Temporal Trends

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3.5 Exceedances of Water-Quality Standards and Comparison Values in Water-Supply Wells

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[illegible] Other Organic Chemicals Reported at Concentrations Greater Than Method Detection Limits

[illegible] Petroleum Hydrocarbons

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6.7 Geochemical Changes from Biodegradation of Dissolved Organic Compounds and Effects on Water Quality and Potability

6.7.1 Microbial Biodegradation of Organic Compounds and Geochemical Evolution

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[illegible] Water Quality and Palatability Effects

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[illegible] Dissolved Petroleum Hydrocarbons and Microbial Biodegradation

Biodegradation of petroleum hydrocarbons in groundwater [illegible]

[illegible] Treatment Options for Water Quality Issues

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5.8 Data Limitations/Evaluation

5.8.1 Evaluation of Data from MW01 and MW02 Installed by the US EPA

[illegible]

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5.8.2 Other Potential Sources of Petroleum Hydrocarbons

[illegible]

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5.8.3 Use of Water Supply Wells in Hydrogeologic Investigations

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6. CONCLUSIONS AND CONSIDERATIONS FOR ADDITIONAL WORK

6.1 Conclusions

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Pre-drilling water-quality data of groundwater prior to shale gas drilling in the Appalachian Basin: Analysis of the Chesapeake Energy Corporation dataset

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ABSTRACT

Unconventional natural gas production in the Marcellus and Utica formations of the Northeastern United States raises concerns about potential impacts to shallow groundwater. We examined and interpreted 13,040 analyses from pre-drilling groundwater samples from domestic water wells in northeastern (NE) Pennsylvania and 8004 samples from water wells in the "Western Area" which includes southwest Pennsylvania, eastern Ohio, and north-central West Virginia. These samples were acquired on behalf of Chesapeake Energy Corporation as part of its local pre-drilling water supply monitoring program. We evaluated concentrations of major ions and metals relative to federal drinking-water-quality standards upon which regulatory decisions are often based. Chesapeake's dataset, the most comprehensive for these areas, shows that exceedance of at least one water-quality standard occurs in 63% of water well samples in NE Pennsylvania and 87% in the Western Area. In NE Pennsylvania, 10% of the samples exceeded one or more of the United States Environmental Protection Agency's (USEPA) primary maximum contaminant levels (MCLs) for drinking-water supplies, 46.1% of the samples exceeded one or more of USEPA secondary maximum contaminant levels (SMCLs), and another 7% exceeded one or more of USEPA health advisory or regional screening levels for tap water.

In the Western Area 8% of samples exceeded one or more MCLs, 65% exceeded one or more SMCLs, and 15% exceeded one or more health advisory or regional screening levels for tap water. Chesapeake's dataset, orders of magnitude larger than any in previously published literature, shows that water-quality exceedances relate to factors such: as where the sample occurs within the groundwater flow system, the natural groundwater chemical type (hydrochemical facies), the geologic unit producing the water, and/or the topographic position (valley versus upland). Our comparison of these results to historical groundwater data from NE Pennsylvania, which pre-dates most unconventional shale gas development, shows that the recent pre-drilling geochemical data is similar to historical data. We see no broad changes in variability of chemical quality in this large dataset to suggest any unusual salinization caused by possible release of produced waters from oil and gas operations, even after thousands of gas wells have been drilled among tens of thousands of domestic wells within the two areas studied. Our evaluation also agrees with early researchers such as Piper (1933) and Lohman (1939, 1937) who found that the saline waters in both areas underlie fresher groundwater. The saline water is naturally-occurring connate brine or salt water which has not been flushed by circulating meteoric water; rather than vertical migration of salt water from deep strata such as the Marcellus shale as suggested by Warner et al. (2012). Elevated metals concentrations, particularly iron and manganese, partly relate to sample turbidity; dissolved metals would provide a more accurate measurement of metals in shallow groundwater than does the total metals analysis typically required by regulations.

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1. Introduction

The recent (post-2005) drilling and production of "unconventional" natural gas from the Marcellus and Utica Shale formations (Devonian and Ordovician age, Appalachian Basin, eastern North American) sparked broad public concern that shallow groundwater may be affected by recent drilling or production activities (e.g. Vidic et al., 2013; Kramer, 2011). Natural gas and oil produced from tight shale, coal beds, and tight sands using horizontal drilling and/or hydraulic fracturing techniques are termed "Unconventional". State regulatory agencies responded to this concern by generating regulatory frameworks designed to assess pre-drilling (baseline) water supply quality to identify possible changes in solute concentrations related to hydrocarbon extraction. Conventional oil and gas development, coal mining, agricultural development, septic effluent discharges, road salting, commercial development, and industrial development all have occurred in Appalachia for over 100 years; therefore this region cannot be viewed as "pristine" in the strictest sense. Without a proper understanding of pre-existing groundwater geochemical variability, investigators may incorrectly conclude that unconventional oil and gas development has altered shallow groundwater quality when it has not (i.e. a false positive).

In this paper, we present interpretation of the inorganic chemical composition of over 21,000 samples of groundwater collected by third party contractors from individual domestic or stock water-supply wells before Chesapeake Energy Corporation (Chesapeake) drilled nearby Marcellus and Utica shale oil/gas wells in Pennsylvania, West Virginia, and Ohio. In our analysis, we compare this pre-drilling data to the combination of federal water-quality standards and guidelines, and evaluate whether geology, topographic position, groundwater circulation patterns, groundwater chemical types (hydrochemical facies), and sample turbidity relate to water quality.

Chesapeake's dataset is unparalleled in size and is orders of magnitude larger than prior studies (e.g. Molofsky et al., 2013, 2011; Warner et al., 2012; Boyer et al., 2012). In northeast (NE) Pennsylvania we interpreted 13,040 groundwater sample analyses from 12,844 domestic/stock water wells and in southwest (SW) Pennsylvania, eastern (E) Ohio, and north-central (NC) West Virginia (collectively termed the "Western Area"), we interpreted 8004 sample analyses from 7983 water wells. Combined, these datasets provide the most comprehensive information on general chemistry and trace elemental composition of shallow groundwater in the Appalachian Basin to date.

2. Background

Unconventional and conventional oil and natural gas production in the Appalachian Basin commonly extract naturally occurring saline waters from oil and gas source beds and these produced saline waters usually exceed drinking-water standards for trace metals, major solutes, and dissolved solids (e.g. Chapman et al., 2012; Siegel et al., 1991). However, trace metals, major solutes, and dissolved solids also naturally exceed regulatory concentrations in many shallow groundwater aquifers in the Appalachian Basin (Ayotte et al., 2011; DeSimone, 2008). Some of these exceedances occur because shale and other rocks in the Appalachian Basin naturally contain potentially soluble minerals incorporating trace metals (e.g. iron, manganese, strontium, and barium) that are common in deep formation fluids (e.g. Siegel et al., 1987). Road de-icing and septic-system effluent can also contribute salinity to potable groundwater. Saline connate water occurs naturally at the base of all shallow freshwater flow systems in both areas (e.g. Williams et al., 1998; Feth, 1970, 1981; Poth, 1963; Lohman, 1939, 1937; Piper, 1933). Exceedances of drinking-water standards in water wells of the Appalachian region have occurred long before unconventional oil and gas development began (e.g. Gross and Low, 2012; Chambers et al., 2012; Low and Galeone, 2006; Williams et al., 1998; Stoner et al., 1987; Taylor, 1984; Matisoff et al., 1982).

Several recent studies in the Appalachian Basin addressed possible changes in potable groundwater chemical conditions prior to and after unconventional shale gas development. Warner et al. (2012) concluded, from variations in salinity, that deep Marcellus shale water may have naturally migrated upwards in the geologic past to shallow aquifers in NE Pennsylvania; a finding challenged by Engelder (2012). In contrast, Boyer et al. (2012) and Molofsky et al. (2013, 2011) found no evidence that methane or dissolved ions are systematically related to oil and gas operations.

Historical studies (e.g. Razem and Sedam, 1985; Poth, 1973a,b; Poth, 1973c, 1963; Carswell and Bennett, 1963; Lohman, 1939, 1937; Piper, 1933) provide detailed discussions on the natural occurrence of groundwater chemistry and connate brines found at shallow depths throughout Appalachia (including NE Pennsylvania and the Western Area). For example, Lohman (1939) noted that in north-central Pennsylvania: "*Most of the waters that contained more than 100 parts of sodium were from the Chemung formation* [now referred to as the Lock Haven Formation] *or from formations or deposits overlying the Chemung, and probably represent diluted connate water, that is, sea water that became trapped in the marine sediments at the time of deposition and subsequently diluted by meteoric water*." Fresh groundwater of meteoric origin flushes the salt water from the upper-most rocks, but only partially flushed it from deeper rocks, or from strata that have little groundwater circulation (such as in valleys) which Poth (1963) noted is where deep groundwater moves upward.

Eighty years ago, Piper (1933) reported circulating groundwater in bedrock of SW Pennsylvania evolves from young calcium-magnesium bicarbonate (Ca-Mg-HCO_3) water type in the uppermost groundwater, to older sodium-bicarbonate (Na-HCO_3) type at intermediate depths, and still older water deeper (and often in valleys) mixing with residual connate sodium-chloride (Na-Cl) brines. Piper (1933) also found that the presence of coal and carbonate minerals in rocks of Appalachia plays an important role in the occurrence of iron, sulfate, and hardness in groundwater.

However, there has not yet been a groundwater study that determines if chemical quality in groundwater prior to unconventional gas exploitation has been systematically maintained. To this end, we interpret the largest and densest dataset on groundwater quality available for Appalachia: 21,044 analyses from 20,827 domestic groundwater wells in Pennsylvania, Ohio, and West Virginia, orders of magnitude larger than prior studies. These samples were collected as part of Chesapeake's pre-drilling sampling program. Because many wells were nearby and in place prior to any sampling for a new unconventional gas well, this pre-drilling database also serves as a means to explore if prior drilling has modified natural groundwater quality.

Our study and many of the more recently referenced studies lack well-log data for the water wells sampled. This inhibits certain evaluations, but the overall size of the dataset provides significant information about general water-quality patterns. Reliance on previous work (e.g. Razem and Sedam, 1985; Poth, 1973a,b,c, 1963; Carswell and Bennett, 1963; Lohman, 1939, 1937; Piper, 1933) that described groundwater flow system conceptualization helped put these data into proper context. The United States Geological Survey (USGS) is currently conducting a groundwater availability study of the Appalachian Plateaus (USGS, 2014) where Pennsylvanian- and Mississippian-age groundwater flow systems are being studied.

3. Study area and general hydrogeology

Chesapeake's contractors collected and analyzed pre-drilling samples in NE Pennsylvania (Fig. 1) and in a Western Area, covering SW Pennsylvania, NC West Virginia, and E Ohio (Fig. 2). We evaluated these two geographic areas separately since the geology, physiography, and hydrogeology of the two regions differ. Unconventional gas in the Marcellus Formation is being developed in NE Pennsylvania, whereas unconventional gas and hydrocarbon liquids are being developed in the Western Area from both the Utica and Marcellus shales.

In NE Pennsylvania (Fig. 1) the sampling density averages 1 sample per 1.5 square kilometers (km^2) and approximately 98% of samples were collected from Bradford, Sullivan, Susquehanna, and Wyoming Counties; in the Western Area (Fig. 2) sampling density averages approximately 1 sample per 4.7 km^2.

A generalized stratigraphic column for NE Pennsylvania and the Western Area is provided in Fig. 3. The general hydrogeologic settings of both study areas (Razem and Sedam, 1985; Poth, 1973a,b,c, 1963; Carswell and Bennett, 1963; Lohman, 1939, 1937; Piper, 1933) have been known for many years as well as the general geochemical variability of shallow groundwater. Potable groundwater in NE Pennsylvania generally occurs in the upper 250 meters (m) of the subsurface, and overlies a freshwater-saline interface approximately 250 m deep under uplands and approximately 60 m deep beneath valleys (Williams, 2010). In NE Pennsylvania, relief is greater and regional flow systems are mostly found in valley bottoms where groundwater flow is controlled by regional dip of formations, topography, and hydrogeological boundaries coincident with large river valleys (Carswell and Bennett, 1963). In NE Pennsylvania, potable groundwater comes from fractured sandstones, siltstones, and shale of the Catskill and Lock Haven formations of Upper Devonian age (Figs. 1 and 3), alluvial deposits, and glacial outwash. Approximately 1500–3000 m of underlying rock separates the Marcellus and Utica shales producing commercial gas from these drinking-water aquifers in our study areas.

The Catskill and underlying Lock Haven formations consist of deltaic and marine shale, siltstone, sandstone and conglomerate interbedded with carbonaceous sediments and sometimes coal (e.g. Wilson, 2014; Castle, 2000; Taylor, 1984; Piotrowski and Harper, 1979). Precipitation recharges shallow groundwater systems through vertical shallow fractures and unconsolidated materials, and groundwater naturally discharges to streams in intervening valleys or as side-slope springs (e.g. Williams et al., 1998; Seaber et al., 1988).

In the Western Area, potable groundwater comes from Mississippian- and Pennsylvanian-age fractured rocks of the Allegheny-Pottsville formations and the Conemaugh and Dunkard groups (Fig. 2). These fractured rocks consist mostly of fluvial-deltaic non-marine shale, siltstone, sandstone, and limestone interbedded with commercially minable coal (Stoner et al., 1987; Razem and Sedam, 1985). In the Western Area, where relief is lower than in NE Pennsylvania, there is less driving head necessary to form deep flow systems. Here, groundwater flow is characterized by short flow paths typically extending no more than a few tens of kilometers (encompassing local and intermediate flow systems) with fresh water extending a few tens of meters below land surface before salt water is encountered (USGS, 2014). Permeable coal units above streams can act as drains and can be regional potable water aquifers (e.g. Carswell and Bennett, 1963).

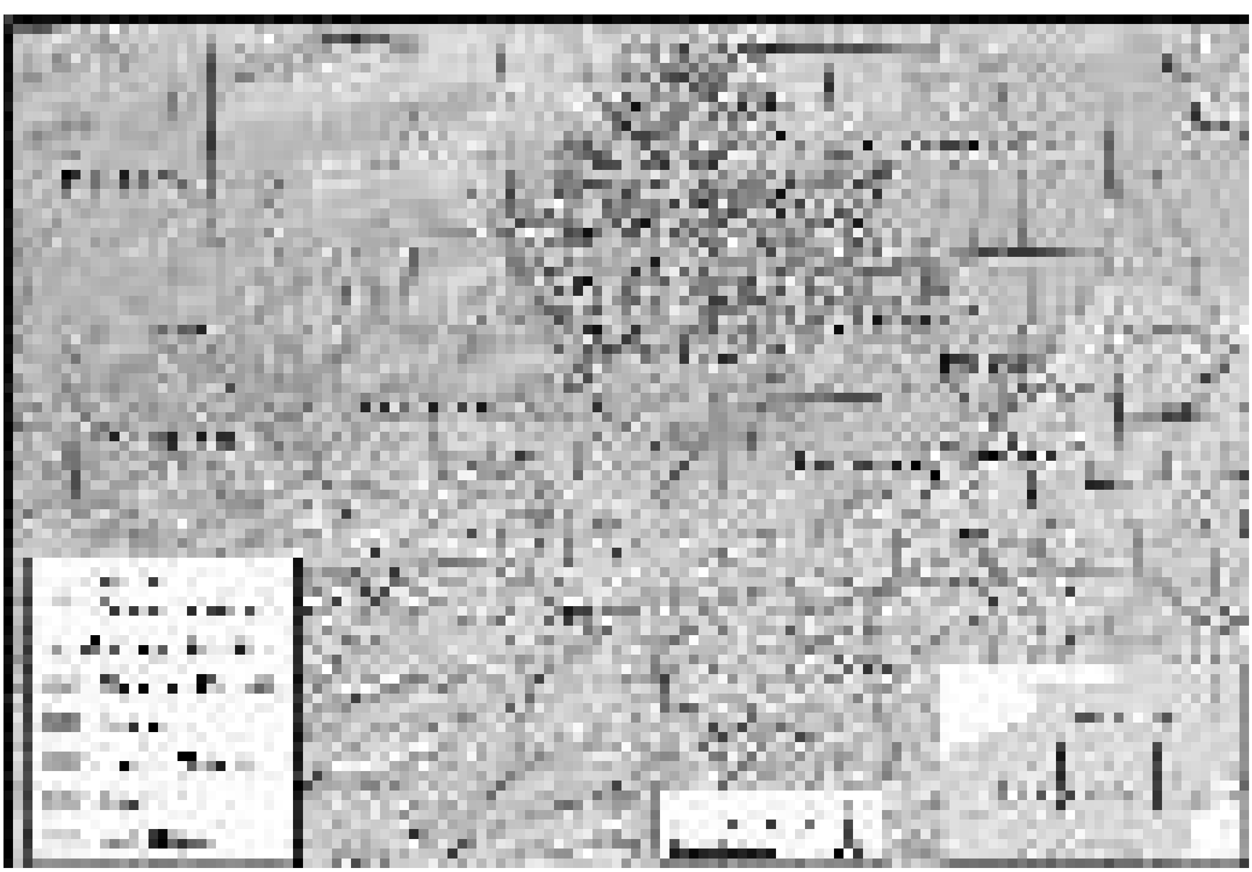

Fig. 1. Chesapeake's pre-drilling sample sites and key stratigraphic units in Northeast Pennsylvania. Samples that fall outside of the key stratigraphic units are categorized in the "Other" formation category. Approximately 98% of pre-drilling samples were collected from these 4 counties in NE PA: Bradford, Sullivan, Susquehanna, and Wyoming.

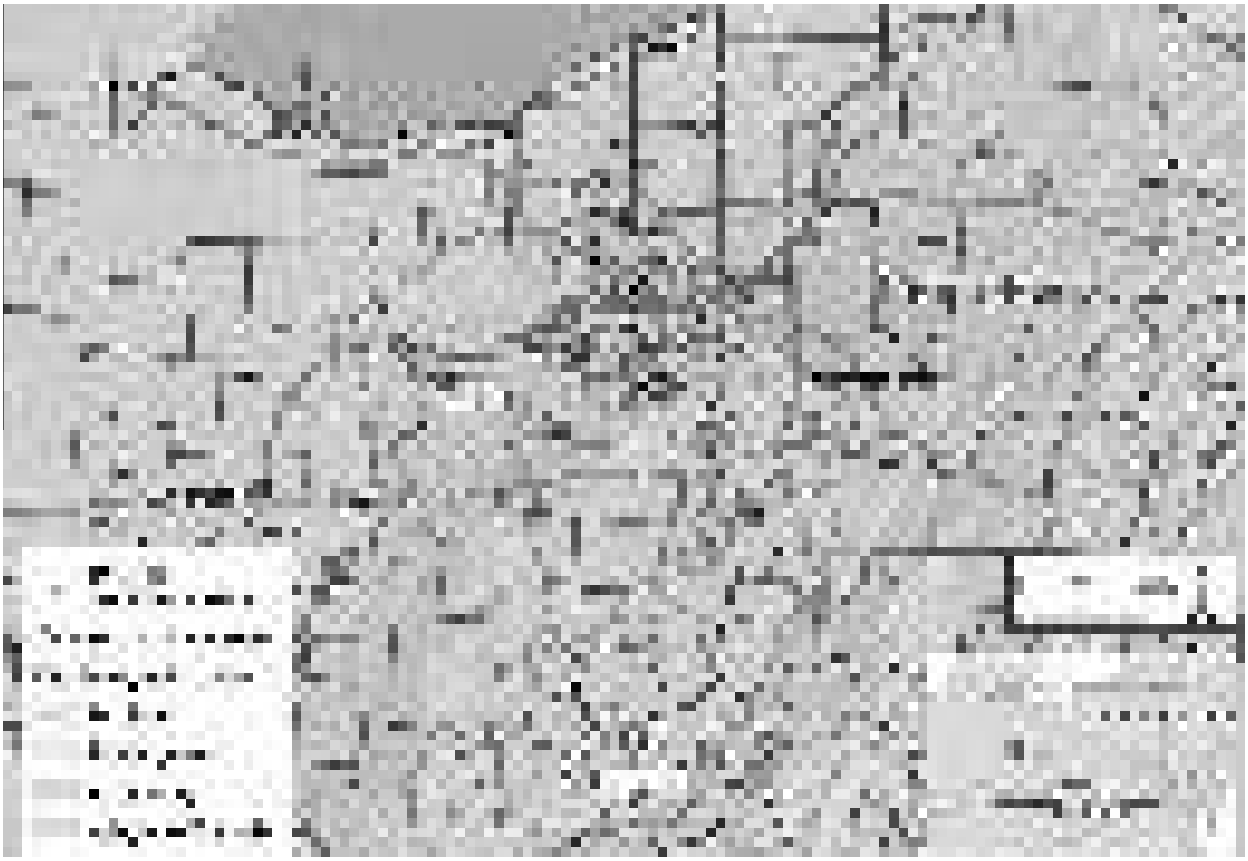

Fig. 2. Chesapeake's pre-drilling sample sites and key stratigraphic units in the "Western Area". Samples that fall outside of the key stratigraphic units are categorized in the "Other" formation category.

In both areas, shallow wells typically yield fresh water whereas deeper wells generally find salt water (in valleys, shallow wells also commonly find salt water). According to Piper (1933), Lohman (1937, 1939), and Poth (1963) the salt water is connate water (salt water present during marine deposition) but has been modified by various physical and chemical processes. In the upper-most rocks, connate salt water has been flushed by recharging meteoric water in contrast to deeper water only being partially flushed or not flushed at all by circulating meteoric waters.

In NE Pennsylvania, groundwater typically evolves along flow paths from a Ca-HCO_3 hydrochemical facies to a Na-HCO_3 facies and finally sometimes to Na-Cl facies groundwater in valley discharge areas (e.g. Molofsky et al., 2013; Williams et al., 1998; Callaghan et al., 1998). Domestic water wells can also intersect isolated coal lamina within the Catskill and Lock Haven formations (e.g. Wilson, 2014; Molofsky et al., 2011). Where the coal occurs, particularly in the Western Area, oxygenated recharge water leads to groundwater with high concentrations of iron and sulfate from pyrite oxidation.

Most private domestic water wells completed in consolidated bedrock in both the Western Area and NE Pennsylvania consist of open-rock wells, with only a short surface casing driven through the shallow, unconsolidated sediments or unstable bedrock. As a result, there is no filter pack to filter out clay- and silt-sized sediment and other particulates since the well bore is open to the formation. In contrast, water wells completed in alluvium and glacial deposits in stream valleys are typically open-ended completions (casing with an open end). Screened completions are rare in domestic wells (Williams et al., 1998).

4. Methods

4.1. Regulatory framework

The geochemical dataset we describe consists of data compiled from Chesapeake's groundwater quality survey programs designed to sample domestic/stock water wells within a radius of 762–1219 m from proposed unconventional oil and gas well sites prior to drilling operations. The spacing of sampling was done according to accepted state regulatory programs, but Chesapeake often went beyond the requirements of these programs (for example the required pre-drill sampling distance per Pennsylvania Department of Environmental Protection (PADEP) regulations in Pennsylvania is 762 m, but Chesapeake often extended the sampling distance to 1219 m). Chesapeake's contractors collected 13,040 pre-drilling samples from domestic or stock water-supply wells in NE Pennsylvania between June of 2009 and January 2012. In the Western Area 8004 samples were collected by contractors between September of 2009 and June 2012. Pre-drilling sampling continues in both areas and all data is released to relevant state agencies and individuals whose groundwater was sampled.

Although these samples were collected to provide *pre-drilling* water quality reference points at the local scale, conventional oil and gas drilling had occurred in both regions prior to 2007. In the four primary counties of NE Pennsylvania (Bradford, Sullivan, Susquehanna, and Wyoming), where 98% of these pre-drilling sampling data were collected, approximately 78 conventional wells had been drilled from 1900 through 2012, compared to about

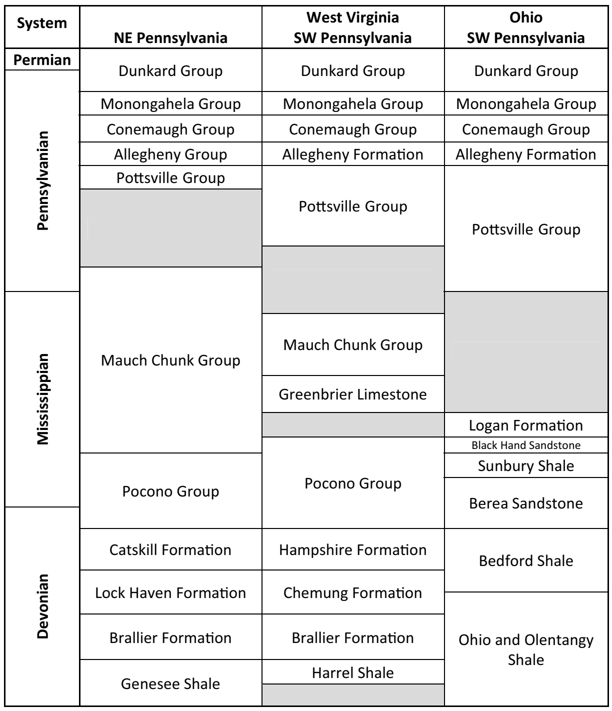

Fig. 3. Generalized stratigraphic columns for Paleozoic rocks in Pennsylvania, West Virginia, and Ohio that occur at/near the surface or are part of bedrock aquifers in the study areas (modified from Lloyd and William (1995), Trapp and Horn (1997), Ohio Geological Survey (1998), and USGS (2012)). The Dunkard Group includes the Greene, Washington and Waynesburg formations in Pennsylvania, West Virginia, and Ohio. The Pocono Formation is also known as the Burgoon Formation or Burgoon Sandstone in Pennsylvania. The lower portion of the Pocono Formation is also known as the Huntley Mountain Formation in Pennsylvania. The stratigraphy for SW Pa is most similar to West Virginia or Ohio.

1934 unconventional wells from 2007 through 2012 according to PADEP records (PADEP, 2015). Prior to 2007, approximately 5 unconventional wells had been drilled in NE Pennsylvania, and slightly over 30 in the Western Area.

Therefore, while the dataset represents pre-drill samples for specific future unconventional oil/gas wells, the same samples also reflect *post-drilling* samples for oil/gas wells than may have been in place at the time of sampling. Siegel et al. (2015) recently reported from the same dataset in NE Pennsylvania that there is no systematic increase in dissolved methane proximate to existing gas and oil wells. In the Western Area there are a much larger number of pre-existing conventional wells dating back to the mid-1800s.

4.2. Water well sampling protocols

Chesapeake's contractors collected groundwater samples from actively used domestic or stock water wells (from water outlets before treatment systems or pressure tanks, if possible) three to six months before natural gas drilling began in accordance with state regulatory programs. No samples were collected from public water systems. Sampling adhered to individual state regulatory protocols, and samples were analyzed in accordance with established US Environmental Protection Agency (USEPA) test methods and internal laboratory standard operating procedures (SOPs) using best practices accredited by the National Environmental Laboratory Accreditation Program (NELAP) and/or state regulatory authority. Chesapeake used three national contract laboratories for the analytical testing and four national contract environmental engineering firms to collect and process the groundwater samples. Chesapeake personnel did not collect, handle, or transport any pre-drilling samples.

In some instances more than one sample was collected from the same well. In almost all of these instances, only one additional sample was collected. In NE Pennsylvania, approximately 196 additional samples were collected, and in the Western Area approximately 21 additional samples were collected. These additional samples constitute a very minor portion of the dataset, and almost all were still collected as pre-drilling samples. Where these additional samples exist, the water-quality evaluation only used the highest concentration for a particular parameter.

As per protocols, unfiltered water samples were collected after allowing water to run for approximately 15 min to clear water lines and pressure tanks. Field measurements for water temperature, turbidity, pH, and specific conductance were made using standard field instruments (e.g. handheld YSI 556MPS multi-parameter meter and Lamotte 2020 field turbidity meter).

Once stability in field measurements were obtained, the sample aliquot was field-preserved with nitric acid to pH < 2 standard units (S.U.) for metals analyses and one or more unpreserved aliquots were collected for general chemistry parameters. Unpreserved aliquots for dissolved light gases were also collected and those results will be discussed in another paper. All samples were shipped to analytical laboratories overnight under chain-of-custody control and maintained at 6 °C or less.

The standard pre-drilling analytical parameter lists for both NE Pennsylvania and the Western Area are shown in Table S1 in Supplemental Information. This parameter list included 38 different laboratory analytical parameters as well as 4 field parameters. Figs. S1 (NE Pennsylvania) and S2 (Western Area) in the Supplemental Information present box and whisker plots showing percentiles for each parameter. Tables S2 (NE Pennsylvania) and S3 (Western Area) in the Supplemental Information summarizes the key statistics (such as mean, median, minimum value, maximum value, percentiles, and standard deviation) for each parameter. Analytical detection limits were equal to or below applicable water-quality standards noted in Table S1 (Supplemental Information). All analyses were done according to USEPA protocols and quality assurance, with appropriate instrumentation; the reader is referred to USEPA for details on these standard laboratory approaches (USEPA, 2014).

The analytical data was reviewed for Quality Control and Quality Assurance by third party contractors who uploaded the data into an electronic database maintained by Chesapeake.

4.3. Drinking-water standards and benchmarks

We compared the pre-drilling groundwater data to USEPA drinking-water standards (USEPA, 2015a) and guidelines: primary maximum contaminant levels (MCLs), secondary maximum contaminant levels (SMCLs), health advisory levels (HALs), or regional screening levels (RSLs) for tap water (USEPA, 2015b). The USEPA primary drinking water MCLs are enforceable health-based standards applied to public-water systems whereas SMCLs are non-enforceable standards for contaminants that may cause cosmetic or esthetic effects to the water. The USEPA HALs provided information on constituents (such as sodium) that may affect human-health. Finally, USEPA RSLs are risk-based screening levels considered protective for human health. Although we recognize that these standards are locally enforced differently, they have in practice been considered the same in the public debate over unconventional gas development (ATSDR, 2011).

If there was no MCL for a parameter, we used the SMCL. If no MCL or SMCL was available, we used a HAL or RSL for tap water. All regulatory drinking-water standards or guidelines used in this evaluation are provided in Table S1 (Supplemental Information).

4.4. Water well topographic position

Heisig and Scott (2013), Williams et al. (1998), and Stoner et al. (1987) identified possible relationships between water chemistry and topographic position in the Appalachian Basin and we similarly assessed this by classifying water chemistry in wells in "upland" or "valley" topographic settings. We include hilltops and intervening slope areas in the upland category; valleys include valley floors.

Topographic position can be broadly used as a proxy for groundwater residence time, or age based on subsurface flow paths through aquifers. Fig. 4 shows a generalized conceptual hydrochemical flow model for the Appalachian Plateau. Typically, older groundwater becomes more mineralized due to water–rock interactions along flow paths and mixing with connate brines. Streams in valley settings of the Appalachian Plateau serve as local- to intermediate-scale groundwater discharge zones, along with side-slope springs. Groundwater in major river valleys, the terminus of local, intermediate and regional groundwater flow paths, can be more mineralized than groundwater in upland settings. The left-hand side of the Fig. 4 reflects conditions more representative of the Western Area where interlayered coal seams containing pyrite lead to more sulfate in shallow groundwater than found in NE Pennsylvania.

We evaluated topographic position in NE Pennsylvania and the Western Area by two automated approaches. The first, the Topographic Position Index (TPI) method (Jenness et al., 2013; Weiss, 2001) compares (LiDAR) elevation at a particular location to the mean elevation in a neighborhood around the location. The second method uses average valley widths (after Molofsky et al. (2011)) and then applies buffers of corresponding size to stream centerlines from the National Hydrography Dataset (NHD) to validate the locations of valleys. We used the same buffers used by Molofsky et al. (2011); a 305 m buffer for the larger streams and a 152 m buffer for the smaller streams. In our analysis, we only used those wells where both methods agree with respect to occurrence of water type relative to topographic position.

4.5. Water well completion depth and geology

We estimated the probable formation that each well was set in by overlaying the well locations onto GIS shape files of USGS or state geologic survey bedrock maps (Nicholson et al., 2007; Pennsylvania Bureau of Topographic and Geologic Survey, 2001).

Information about well depth was provided by the well owner. Williams et al. (1998) noted that there are three water well types in common use in NE Pennsylvania: an open-hole well, an open-end well, and a screened well. Williams et al. (1998) noted that screened wells are mostly used in non-domestic wells completed in unconsolidated deposits, while the first two are used in most domestic completions.

We did an extensive review to match water-well logs in the Pennsylvania Groundwater Information System (Pennsylvania Department of Conservation and Natural Resources, 2014) database with the domestic/stock water wells from which the pre-drill samples were obtained, with limited success. Taylor (1984) published well completion data from 890 wells in Bradford, Susquehanna, Sullivan, and Wyoming counties of NE Pennsylvania wherein he showed completion information for wells based upon topographic setting (i.e. valleys, side slopes, or hilltops). Almost all wells completed on hilltops and side slopes were open-hole completions in the Catskill and Lock Haven formations, and in the valleys over 71% of the domestic wells after penetrating alluvium were still open-hole completions in the bedrock. Water wells in valley settings therefore cannot be assumed to be completed only in unconsolidated materials. Most high capacity wells for public, irrigation, industrial or commercial needs are completed in unconsolidated materials, but this is not the case with domestic water wells, which make up the entirety of the pre-drilling dataset.

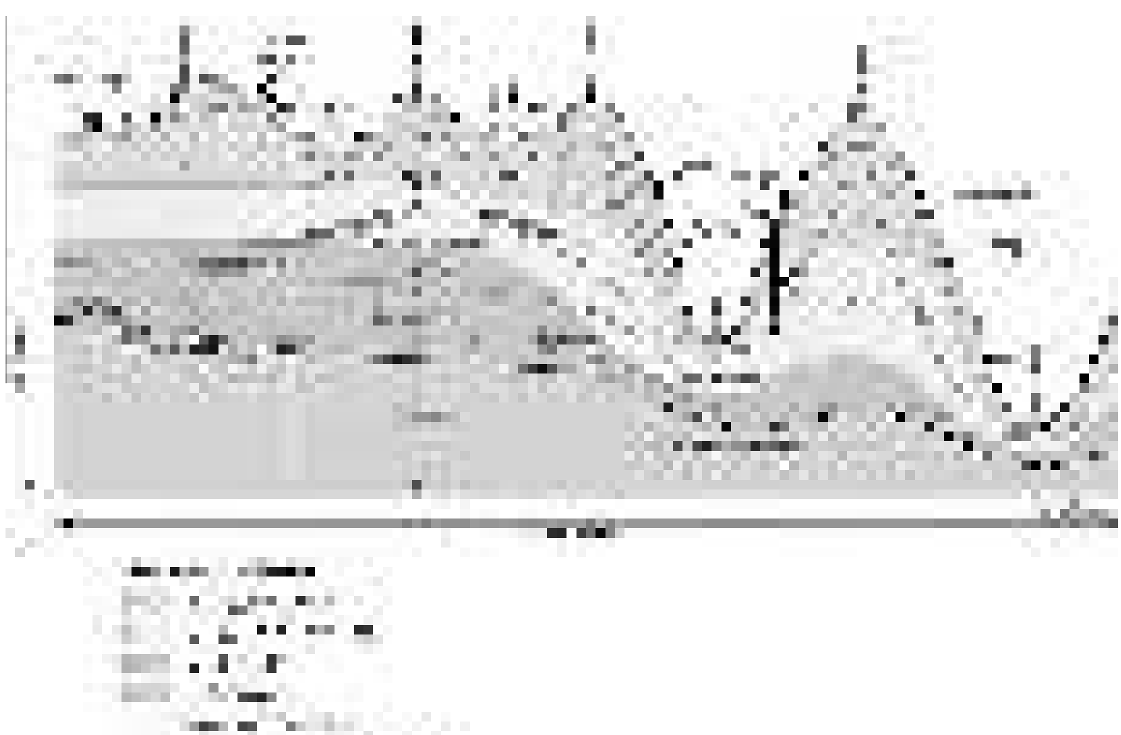

Fig. 4. Conceptual diagram showing groundwater flow zones in the Appalachian Plateau for different hydrogeologic settings and relationship with hydrochemical facies (modified after Fleeger (1999) and Wunsch (1995)). In the "Western Area", flow systems are shallower with waters in both regions discharging to streams possibly having variable chemistries based on the nature of the local convergence of flow systems at different scales—local, intermediate, and regional.

For our analysis we assumed that domestic water wells on side and upland areas overwhelmingly consisted of open-hole wells obtaining water from fractured bedrock. We knew the mapped bedrock formation underlying valleys and assigned valley wells per that bedrock type, knowing that there will be some unknown percentage of wells that would actually be completed only in the overlying alluvial or glacial deposits. However, our review showed that the common practice in the four-county area of NE Pennsylvania is to drill through unconsolidated sediments, set steel surface casing, and complete an open-hole well in the bedrock formations.

We also used well depth estimates to evaluate completion intervals in the valleys. Information provided by Taylor (1984) showed that the median depth for water wells completed within unconsolidated materials in valleys was 18.3 m (75th percentile was 28.3 m) for the four-county area representing the majority of the pre-drilling samples. When we evaluate the water well depth database for NE Pennsylvania, approximately 75% of wells completed in valleys have depths that exceed the median value of 18.3 m for unconsolidated deposit wells.

For geological evaluations involving these data, we have removed those wells that are more likely to be completed in unconsolidated deposits in valley settings which have reported well depths less than 28.3 m (approximately 3000 wells removed). Removing these shallow valley wells removes most of the potential influence of the unconsolidated deposits. After completing this process, we found essentially no difference in median parameter values of concentrations measured between the two datasets, showing that wells in valleys are mostly completed in bedrock formations, or that unconsolidated deposits have nearly identical water quality.

Table S4 (Supplemental Information) provides our estimate of the number of samples from each geological group or formation in NE Pennsylvania and in the Western Area. Table S5 (Supplemental Information) provides estimates of the number of samples from each geological formation in a particular topographic position (upland or valley setting).

We assumed that the bedrock unit supplying the groundwater to the water well was the same as the bedrock unit mapped at the ground surface or sub-cropping beneath the valley sediments. This is a reasonable assumption for the majority of wells because of the near horizontal attitude of the formations and overall thicknesses. For example, the Catskill and Lock Haven formations can be 1707 m (Inners, 1981) to 1219 m (Taylor, 1984) thick, respectively, compared to a median well depth in this area of 50.3 m (maximum reported well depth of 518 m). In NE Pennsylvania, 94% of the pre-drilling samples were from locations overlying the Catskill or Lock Haven formations.

For simplification in the Western Area, we evaluated bedrock at the Group level rather than the individual formation (see stratigraphic column in Fig. 3). For example, we combined the Greene, Washington, and Waynesburg formations into the Dunkard Group. In the Western Area, 90.5% of the pre-drilling samples were from the Allegheny/Pottsville formations and the Conemaugh Group. Our broad approach would lead to misidentification of the geology only where wells are extremely deep (and could intersect a different unit) or near the contact between formations; a very small fraction of the overall dataset. In areas where alluvial or glacial deposits were present, most domestic water wells were still completed into bedrock, and steel casing driven through the unconsolidated deposits.

4.6. Analysis of data

We used Microsoft Access and Excel, and Stata® software, version 11.2 (StataCorp., 2009) to calculate summary statistics. We provide a summary of the pre-drilling analytical data including key statistics as well as boxplots for analytes in the Supplemental Information.

4.7. Geochemical software and ion balance

We used the water-quality analysis program AquaChem® (Schlumberger, 2011) to generate Piper trilinear diagrams, calculate ion-charge balances, and determine primary ion water types. Piper diagrams visually characterize the broad geochemical evolution of shallow groundwater and can distinguish different groundwater types, mixtures of waters, and in some cases, major geochemical processes along groundwater flow paths (e.g. Hounslow, 1995).

We classified groundwater type or hydrochemical facies by selecting the single cation and anion present in each sample with highest percentage of cation or anion concentration (as milliequivalents per liter, meq/L). For convenience and the purpose of this broad synthesis, we did not address in detail mixed-water types, although the Piper diagrams show that geochemical mixing widely occurs. For example, mixed water types such as calcium-sodium bicarbonate (Ca-Na-HCO_3) or Ca-Mg-HCO_3 do occur within our study areas. It is possible that subdividing our broad regions into smaller areas will lead to better understanding of specific geochemical processes conditional with natural water chemistry. We include results only with a charge balance within ±7% in all groundwater hydrochemical facies evaluations. However, the full dataset, regardless of charge balance, was used for comparisons to relevant drinking-water standards.

4.8. Water treatment by water well owners

As part of our study, we determined whether water samples were collected prior to or after a home water-treatment system (a bag filter for sediment removal, water softener, and far less commonly, a home/tap reverse-osmosis unit). In cases where the water-treatment status was unknown, it was assumed that he water was sampled before any treatment in accordance with the sampling program protocols.

Table S6 (Supplemental Information) shows the breakdown of sample numbers that were treated, untreated, or unknown for each region. We excluded all treated and unknown samples, regardless of treatment system type, in hydrochemical facies classification by Piper diagrams. Although our approach excluded many samples for this specific evaluation, over 2600 untreated samples remained in NE Pennsylvania and over 3400 samples in the Western Area within a charge balance of ±7%. These are still large datasets by any measure and are sufficiently representative of the total population.

In our interpretation of arsenic, barium, cadmium, chromium, iron, lead, lithium, manganese, mercury, silver, sodium, and strontium water-quality patterns, we used all sample results regardless of charge balance or whether the water sample was collected after a treatment system. We used total results (unfiltered samples) since state regulatory agencies base their assessments on concentrations of total metals in water-supply systems, not dissolved solutes even if particulates occur in the water. We understand that from a regulatory standpoint, turbid water can lead to an inference of contamination. In turbid water, metals bound in the crystal lattices of clay-sized particulates become incorporated in the chemical analysis. These particulates are natural, and regulators may interpret these results as contamination presumed in the dissolved state when, in fact, it is not present. We wanted to assess how prevalent such false positives could be.

5. Results

5.1. Quality of analyses

In addition to evaluating ion charge balances for individual samples, the mean of all charge balances can provide an assessment of overall quality of a database (Fritz, 1994) and should be close to zero. The mean raw ion charge balance (with 1 standard deviation) was 0.30 ± 7.82% and 1.77 ± 6.07% for NE Pennsylvania and the Western Area, respectively. The overall mean of the absolute ion charge balance is 4.53 ± 6.37% and 3.86 ± 5.01%, for NE Pennsylvania and the Western Area, respectively.

If we only consider those data with ion balances within ±7%, then for NE Pennsylvania the overall mean ion balance is 0.49 ± 3.17% and the absolute mean balance is 2.63 ± 1.84%. Similarly for the Western Area, the overall mean ion balance is 1.41 ± 2.97%, and the absolute mean balance is 2.74 ± 1.82%. These mean ion balances are considered excellent for both groups of samples. For NE Pennsylvania, 83% (9361) of the samples with analyzed major ions balanced within our ±7% criterion and within the Western Area, 88% (7042) of the samples were within ±7%. We present the summary statistics for the ion balances in Table S7 in the Supplemental Information for reference.

5.2. Pre-drilling groundwater-quality exceedances

5.2.1. Northeast Pennsylvania

Table 1 shows the percent of exceedances of USEPA water-quality standards from the pre-drilling groundwater samples in NE Pennsylvania. Fig. 5a–k shows the spatial distribution of pre-drilling samples exceeding standards for arsenic, barium, iron, lead, manganese, sulfate, chloride, lithium, sodium, total dissolved solids (TDS), and turbidity, respectively.

Fig. S3 in the Supplemental Information shows the spatial distribution of all samples exceeding an MCL, SMCL, HAL, or RSL (excluding turbidity) for NE Pennsylvania. In NE Pennsylvania, 10.2% of the pre-drilling samples exceeded one or more MCLs (excluding turbidity). If turbidity is included then 22.1% of all pre-drilling samples exceed a MCL. Secondary MCLs were exceeded in 46.1% of the samples. Fig. 6 shows the percentage of water-quality exceedances per guideline or standard category for NE Pennsylvania.

The most common pre-drilling MCL exceedances were for turbidity (15.6%), arsenic (4.2%), lead (3.6%), and barium (3.3%). The most common SMCL exceedances were for manganese (33.9%) and iron (23.8%). Sodium concentrations exceeded the USEPA HAL of 20 mg/L in 38.5% of pre-drilling samples. Lithium exceeded the USEPA RSL for tap water in 19.1% of the well pre-drilling samples. Other metals also occasionally exceeded recommended standards. If all the MCL, SMCL, RSL, and HAL standards or guidelines are considered collectively (excluding turbidity), then approximately 63.0% of all pre-drilling samples collected from NE Pennsylvania exceeded at least one drinking-water standard, 34.3% exceeded two or more drinking-water standards, and approximately 15.4% exceed three or more standards. Including turbidity, 63.1% of pre-drilling samples exceeded one or more drinking-water standards.

Our results agree with historical water-quality studies in NE Pennsylvania (e.g. Ayotte et al., 2011; DeSimone, 2008; Williams et al., 1998; Taylor, 1984) that relied on much smaller datasets and before unconventional gas drilling began. For example, Williams et al. (1998) reported about 50% of 223 water wells sampled in a three county area within the pre-drilling sampling area of NE Pennsylvania had iron and manganese concentrations in groundwater exceeding SMCLs. Williams et al. (1998) also showed that chloride, arsenic, barium, cadmium, and lead concentrations

Table 1
Pre-drilling sample results exceeding drinking water guidelines – Northeastern Pennsylvania.

Parameter[a]	Number of samples analyzed[d]	Drinking water guideline or standard (mg/L)	Drinking water guideline or standard type[b]	Number of Samples exceeding guideline or standard[d]	Percent of Samples exceeding guideline or standard[d]
Arsenic	11,034	0.010	MCL	462	4.2
Barium	11,074	2.0	MCL	362	3.3
Benzene	11,075	0.005	MCL	1	<0.1
Cadmium	11,034	0.005	MCL	4	<0.1
Chloride	11,073	250	SMCL	221	2.0
Chromium	11,034	0.1	MCL	0	0.0
Ethylbenzene	11,075	0.7	MCL	0	0.0
Iron	11,075	0.3	SMCL	2647	23.8
Lithium	1729	0.04	RSL	331	19.1
Lead	11,032	0.015	TTL	405	3.6
Manganese	11,074	0.05	SMCL	3752	33.9
Mercury	11,034	0.002	MCL	0	0.0
Surfactants[c]	11,071	0.5	SMCL	28	0.3
pH	11,073	6.5–8.5 S.U.	SMCL	645	5.8
Selenium	11,032	0.05	MCL	17	0.2
Silver	11,034	0.1	SMCL	0	0.0
Sodium	11,074	20	Advisory	4263	38.5
Strontium	3423	12	RSL	10	0.3
Sulfate	11,075	250	SMCL	82	0.7
TDS	11,075	500	SMCL	554	5.0
Toluene	11,075	1	MCL	0	0.0
Turbidity	11,076	5 NTU	MCL	1738	15.7
Xylenes	11,075	10	MCL	0	0.0

TDS: Total Dissolved Solids.
[a] Samples analyzed for metals were not filtered during sampling or analysis. Some of the results may represent samples collected after a treatment system (e.g. particulate filter, water softener, etc.) within the water distribution system.
[b] Guideline or Standard types (see Table 1): MCL –USEPA primary Maximum Contaminant Level for public drinking water supplies; SMCL –USEPA Secondary MCL; RSL – USEPA Regional Screening Level for tap water (January 2015); TTL – treatment technology action level for lead defined by USEPA for public drinking water supplies; Advisory –USEPA drinking water health advisory for individuals on a restricted sodium diet.
[c] Surfactants analyzed as methylene blue active substances (MBAS).
[d] For duplicate samples only sample with highest value used for each parameter.

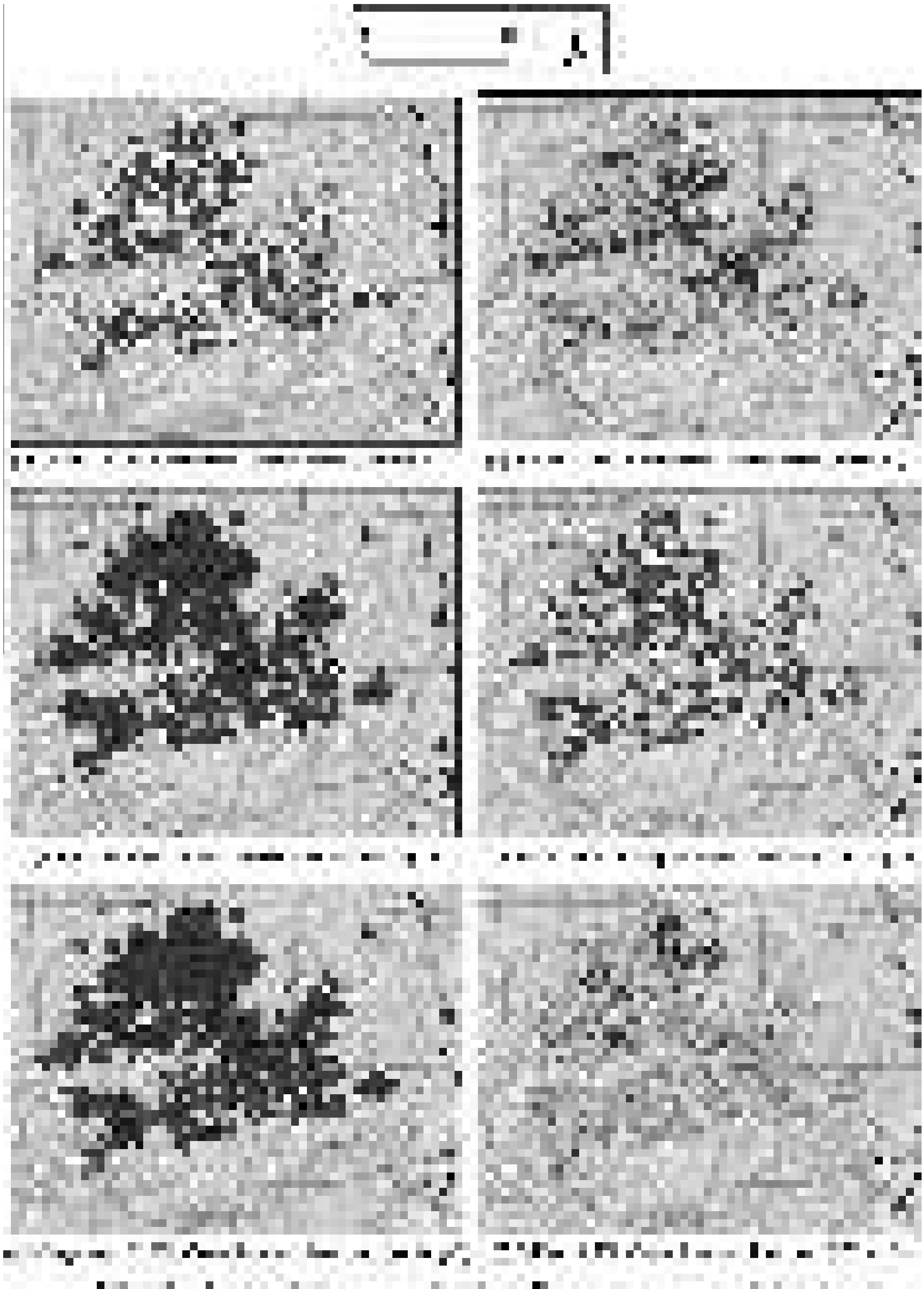

Fig. 5. Distribution of trace metals and other constituents in pre-drilling groundwater samples in Northeastern Pennsylvania. Samples exceeding the applicable MCL or other drinking water standard are shown with a red symbol; such elevated concentrations occur commonly in pre-drilling samples across the study area. The applicable standard and percent of samples exceeding that standard are listed beneath each map. For wells sampled more than once, the sample with the highest value is shown. Layers are vertically stacked in order shown in legend and some "no exceeds" may be covered.

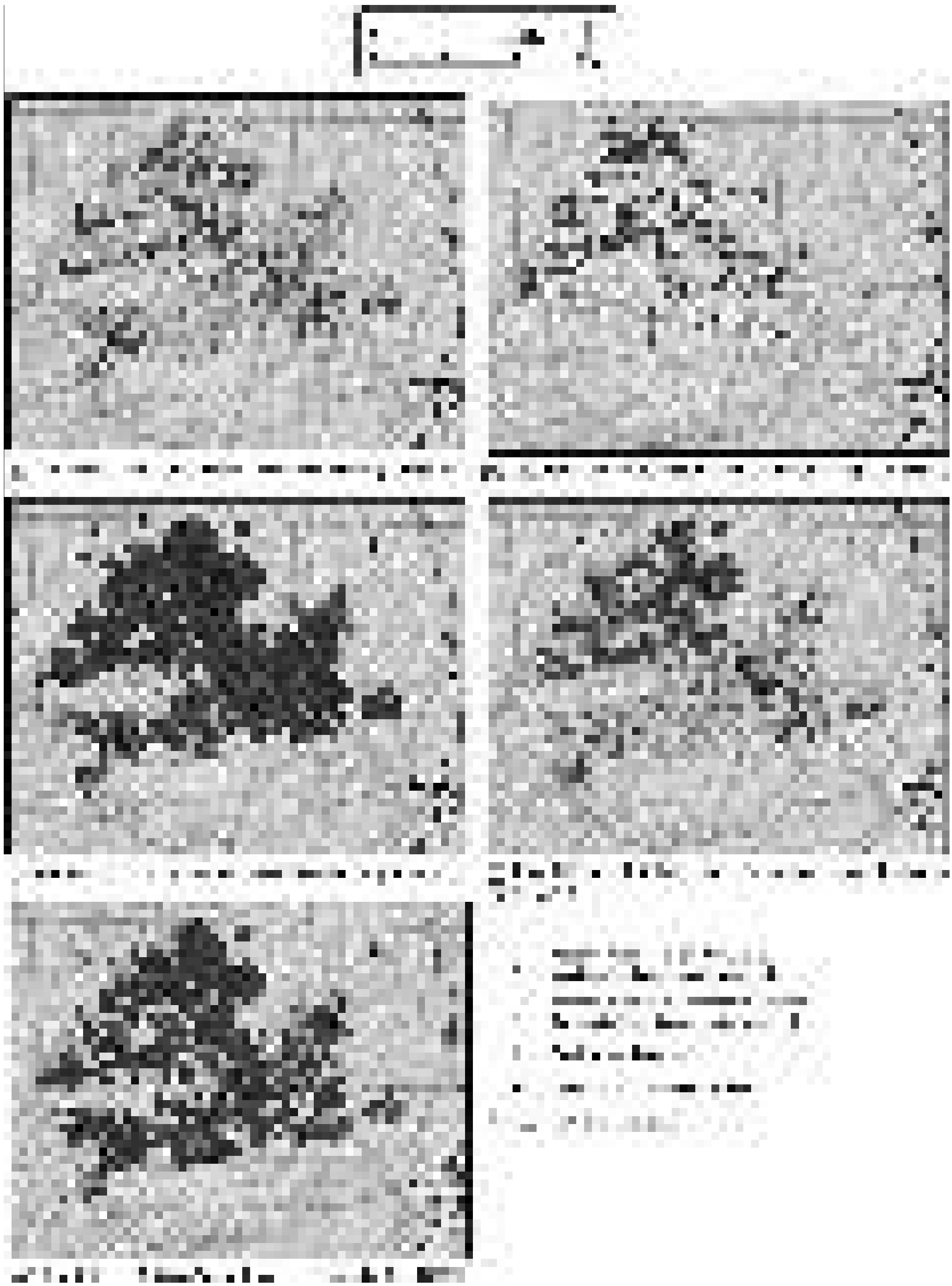

Fig. 5 (*continued*)

in groundwater can also exceed MCLs/SMCLs, especially in restricted-flow zones containing shallow naturally-saline groundwater, and about 40% of sodium concentrations in that study exceeded the HAL.

Table 2 compares the key water-quality parameters from the historical data provided in Williams et al. (1998) and Taylor (1984) to the 2009–2012 pre-drilling data for the Catskill and Lock Haven formations. The median chloride, TDS, and sodium concentrations in the pre-drilling data, shown in Table 2, have similar ranges as the historical data (pre-2007) from the Catskill and Lock Haven formations. These major parameters defining groundwater tie closely with water-quality type.

The Williams et al. (1998) pre-1987 historical data from the Catskill and Lock Haven formations also show that collectively

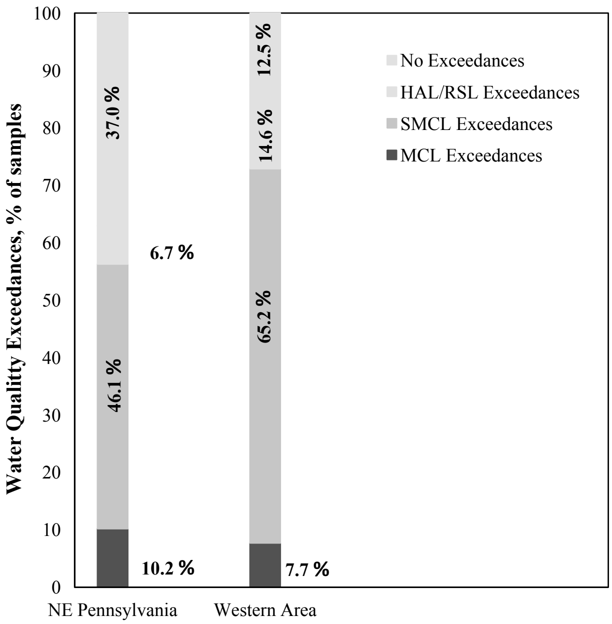

Fig. 6. Figure showing the percentage of water-quality exceedances in NE Pennsylvania and the "Western Area" by guideline. MCLs are USEPA maximum contaminant levels. SMCLs are USEPA secondary maximum contaminant levels. HALs are USEPA health advisory levels and RSLs are USEPA screening levels for water consumed at the tap. Turbidity exceedances are excluded from this chart. Each sample included only once; if there was an exceedance of a primary MCL, then the sample was not considered in the evaluation of the remaining guidelines, and so forth.

22% of historical samples exceeded an MCL, compared to approximately 9.6% (Catskill and Lock Haven samples only) in the 2009–2012 pre-drilling data, with the most common MCL exceedances in the historical data caused by arsenic, barium, selenium, and lead.

Battelle (2013) evaluated pre-2007 historical groundwater data from about 500 water wells (undivided by geological formation) in Bradford and Susquehanna Counties of NE Pennsylvania. Table 3 shows a comparison of the historical groundwater data from the Battelle (2013) report (pre-2007) compared to the pre-drilling data. The median values for all parameters are very similar.

5.2.2. Western area

Table 4 shows the distribution of major solutes and trace metals of interest compared to USEPA water-quality standards for pre-drilling water well samples in the Western Area. Fig. 7a–k shows the spatial distribution of pre-drilling samples that exceed USEPA drinking-water MCLs, SMCLs, RSLs, or HALs for: arsenic, barium, iron, lead, manganese, sulfate, chloride, lithium, sodium, TDS, and turbidity, respectively.

Fig. S4 in the Supplemental Information shows the spatial distribution of pre-drilling sample sites that exceed one or more MCLs, excluding turbidity. In the Western Area 7.7% of all pre-drilling samples exceeded one or more MCLs (excluding turbidity). If turbidity is included then 39% of all pre-drilling samples exceed an MCL. Secondary MCLs were exceeded in 65.2% of the samples.

Excluding turbidity (36.2%), the most commonly exceeded MCLs in pre-drilling samples from the Western Area are lead (4.4%) and arsenic (3.1%). If all of the MCL, SMCL, RSL, and HAL water-quality standards are considered collectively (excluding turbidity) then 87.5% of pre-drilling samples exceed one or more drinking-water standards. About 63.8% of all pre-drilling samples exceeded two or more drinking-water standards, and approximately 31.1% exceeded three or more drinking-water standards. Including turbidity, 88.7% of pre-drilling samples exceeded one or more drinking-water standards. Fig. S4 in the Supplemental Information also shows the spatial distribution of those samples that exceed a SMCL, HAL, or RSL. The most commonly detected natural exceedances of SMCLs were manganese (53.5%), iron (50.0%), and TDS (22.8%). The sodium USEPA HAL of 20 mg/L was exceeded in 51.8% of samples collected from water wells in the Western Area.

Our findings in the Western Area are also consistent with local and national historical water-quality studies (e.g. Ohio USEPA, 2012; Gross and Low, 2012; Chambers et al., 2012; Ayotte et al., 2011; DeSimone et al., 2009; DeSimone, 2008; USGS, 2006; McAuley and Kozar, 2006; White and Mathes, 2006; Stoner et al., 1987; Razem and Sedam, 1985; Matisoff et al., 1982). For example, Razem and Sedam (1985) noted in a study of 100 groundwater samples collected in 1983 that SMCLs were commonly exceeded for TDS (38%), manganese (34%), iron (19%), and sulfate (17%) among other constituents, and that 62% of the samples analyzed exceeded the sodium HAL of 20 mg/L.

5.3. Sample turbidity influence on water-quality

The analysis of unfiltered samples (total analysis) for certain trace metals (e.g. iron and manganese) can produce false positives.

Table 2
Summary of historical analytical data (Williams et al., 1998 and Taylor, 1984) from the Catskill and Lock Haven formations compared to pre-drilling data (2009–2012) for Northeastern Pennsylvania (units are in mg/L). Historical data compared to full pre-drilling dataset for each formation, and also to the dataset where shallower wells were excluded (those <p75) in valleys.

Parameter	Catskill Formation Median concentrations (mg/L)				Lock Haven Formation Median concentrations (mg/L)			
	Historical Pre-1987 Williams et al., 1998 (Taylor, 1984)		Pre-drilling 2009–2012 All Data [Shallow valley wells excluded]		Historical Pre-1987 Williams et al., 1998 (Taylor, 1984)		Pre-drilling 2009–2012 All Data [Shallow valley wells excluded]	
	No.	Median (mg/L)	No.	Median (mg/L)	No.	Median (mg/L)	No.	Median (mg/L)
Calcium	40 (165)	26 (28.1)	5441 [4043]	33.1 [33.7]	54 (45)	39 (39)	4954 [3359]	43.2 [43.6]
Chloride	43 (165)	10 (4)	5441 [4043]	7.4 [7.3]	56 (45)	12 (5)	4954 [3359]	8.6 [8.1]
Iron	41 (165)	0.09 (0.07)	5441 [4043]	<0.05 [<0.05]	55 (45)	0.27 (0.27)	4954 [3359]	0.10 [0.09]
Manganese	36 (165)	0.03 (0.02)	5441 [4043]	<0.015 [<0.015]	52 (45)	0.05 (0.09)	4954 [3359]	0.04 [0.04]
Magnesium	40 (164)	5.5 (5.5)	5441 [4043]	4.9 [5.0]	55 (45)	10 (10.2)	4954 [3359]	10.5 [10.8]
Potassium	36 (164)	2.0 (1.02)	5441 [4043]	1.2 [1.2]	52 (45)	1.1 (2.3)	4954 [3359]	1.5 [1.6]
Sodium	37 (165)	11 (10.1)	5441 [4043]	12.5 [12.6]	54 (45)	28 (23)	4954 [3359]	21.2 [21.5]
Sulfate	41 (164)	10 (10)	5441 [4043]	13.0 [13.0]	55 (45)	16 (15)	4954 [3359]	18.4 [19.0]
TDS	38 (165)	160 (158)	5441 [4043]	165 [167]	51 (45)	300 (238)	4954 [3359]	240 [241]

Notes: Median concentrations for historical samples from Tables 10, 11, 16, 17 and 19 of Williams et al. (1998). Numbers in parenthesis are from Taylor (1984), Table 13 data. These tables also provide the numbers of samples. Calcium, magnesium, and potassium sample counts are from Table 20 of Williams et al. (1998). Historical samples analyzed for dissolved metals; pre-drilling samples analyzed for total metals. Numbers in brackets are pre-drilling data with shallow valley wells removed from dataset.

Table 3
Summary of Battelle's historical (pre-2007) groundwater analytical data summary from Bradford and Susquehanna Counties compared to pre-drilling data (2009–2012) for Northeastern Pennsylvania (units are in mg/L).

Parameter	Historical data, pre-2007 (Battelle, 2013)		Pre-drilling data, 2009–2012	
	Number of samples	Median value (mg/L)	Number of samples	Median value (mg/L)
Calcium (as total)	70	39.7	11,074	35.9
Calcium (as dissolved)	159	31	–	–
Chloride	502	8	11,073	7.5
Iron (as total)	82	0.293	11,075	0.065
Iron (as dissolved)	141	0.110	136	<0.05
Magnesium (as total)	69	8.1	11,074	6.5
Magnesium (as dissolved)	197	6.69	–	–
Manganese (as total)	81	0.080	11,074	<0.015
Manganese (as dissolved)	344	0.0992	135	0.036
Potassium (as total)	82	1.1	11,074	1.3
Potassium (as dissolved)	135	1.05	–	–
Sodium (as total)	82	16.6	11,074	14.4
Sodium (as dissolved)	410	9.86	–	–
Sulfate	231	15	11,075	14.6
TDS	223	196	11,075	189

Notes:
–: No data.
Median concentrations are for historical data from Battelle (2013), Tables 3 and 4. Median concentrations calculated using only the highest parameter values for the duplicate samples.

Table 4
Pre-drilling sample results exceeding drinking water guidelines – "Western Area".

Parameter[a]	Number of samples analyzed[d]	Drinking water guideline or standard (mg/L)	Drinking water guideline or standard type[b]	Number of samples exceeding guideline or standard[d]	Percent of samples exceeding guideline or standard[d]
Arsenic	7933	0.010	MCL	249	3.1
Barium	7944	2.0	MCL	39	0.5
Benzene	7945	0.005	MCL	4	<0.1
Cadmium	7933	0.005	MCL	17	0.2
Chloride	7944	250	SMCL	210	2.6
Chromium	7933	0.1	MCL	1	<0.1
Ethylbenzene	7946	0.7	MCL	0	0.0
Iron	7944	0.3	SMCL	3967	50.0
Lithium	5020	0.04	RSL	113	2.2
Lead	7933	0.015	TTL	347	4.4
Manganese	7944	0.05	SMCL	4251	53.5
Mercury	7933	0.002	MCL	0	0.0
Surfactants[c]	7946	0.5	SMCL	8	0.1
pH	7945	6.5–8.5	SMCL	479	6.0
Selenium	7933	0.05	MCL	6	<0.1
Silver	7932	0.1	SMCL	0	0.0
Sodium	7944	20	Advisory	4113	51.8
Strontium	5931	12	RSL	0	0.0
Sulfate	7943	250	SMCL	387	4.9
TDS	7945	500	SMCL	1810	22.8
Toluene	7945	1	MCL	0	0.0
Turbidity	7946	5 NTU	MCL	2875	36.2
Xylenes	7945	10	MCL	0	0.0

[a] Samples analyzed for metals were not filtered during sampling or analysis. Some of the results may represent samples collected after a treatment system (e.g. particulate filter, water softener, etc.) within the water distribution system.
[b] Guideline or Standard types MCL –USEPA primary Maximum Contaminant Level for public drinking water supplies; SMCL –USEPA Secondary MCL; RSL –USEPA Regional Screening Level for tap water (January 2105); TTL – treatment technology action level for lead defined by USEPA for public drinking water supplies; Advisory –USEPA drinking water health advisory for individuals on a restricted sodium diet.
[c] Surfactants analyzed as methylene blue active substances (MBAS).
[d] For duplicate samples only sample with highest value used for each parameter.

Particulates induced during pumping of water wells are included in analysis of unfiltered samples. When water wells are pumped, the increased velocity of water moving through the formation and the well bore induces clay- to silt-sized sediment into the water. The USEPA recognizes this problem and advocates pumping at rates less than 100 milliliters per minute (mL/min) to minimize turbidity during sampling of monitoring wells (Puls and Barcelona, 1996). Domestic water wells pump at much higher rates, and consequently, turbidity can occur even under normal use. We found about 15.7% of the samples in NE Pennsylvania and 36.2% of samples in the Western Area had turbidity exceeding the USEPA MCL of 5 nephelometric turbidity units (NTU). Many exceedances of total iron and manganese SMCLs relate to this turbidity, confounding true appraisal of trace metals concentrations from a scientific perspective or to characterize potential contamination. In rare cases, low pH in upland areas can react with domestic plumbing, and cause an occasional false-positive issue, but based upon our review, this is likely rare in both regions studied. We provide details on this part of our study in Appendix A of the Supplemental Information.

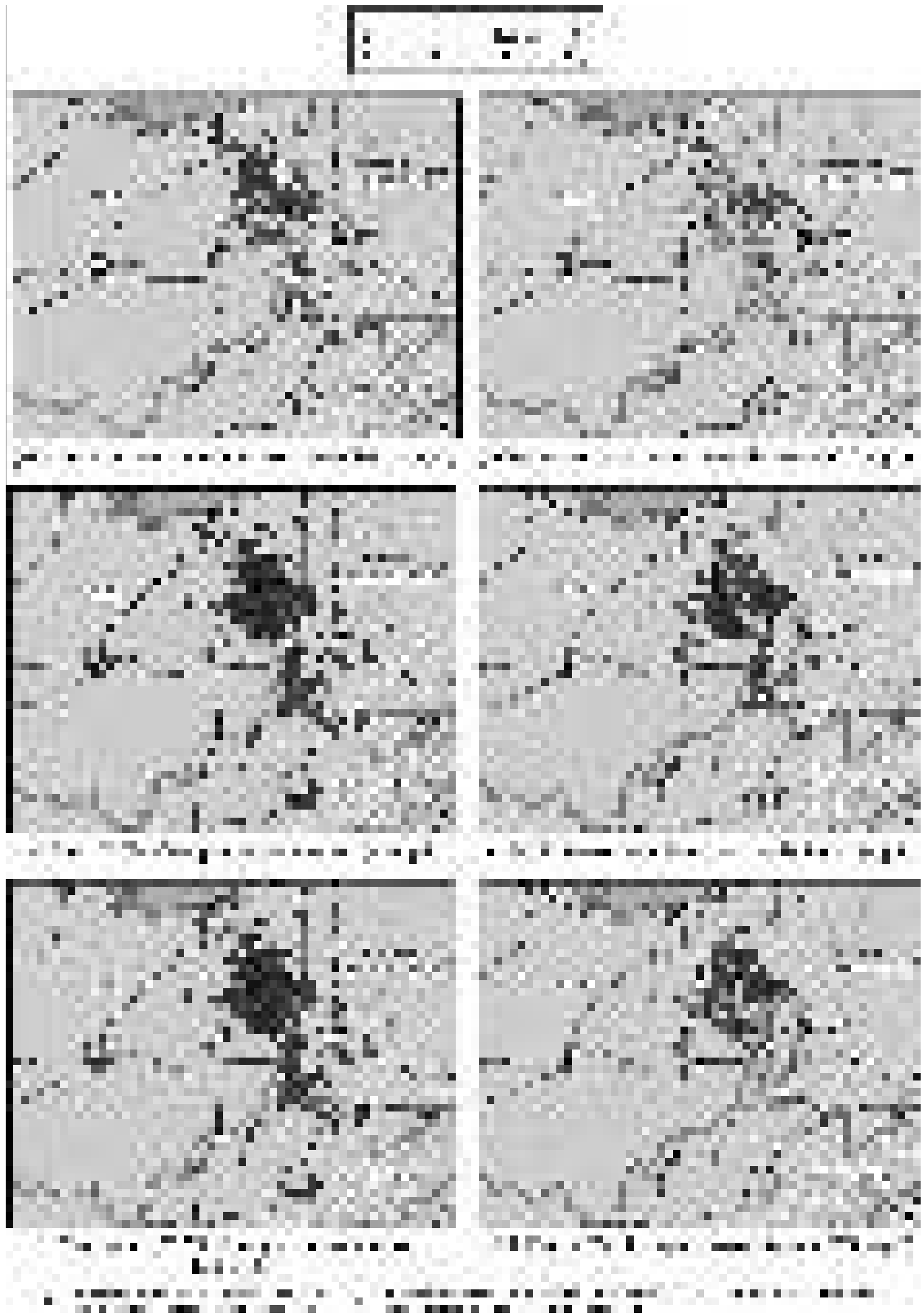

Fig. 7. Distribution of trace metals and other constituents in pre-drilling groundwater samples in the "Western Area". Samples exceeding the MCL or other groundwater standard are shown with a red symbol; such elevated concentrations occur commonly in pre-drilling samples across the study area. The applicable standard and percent of samples exceeding that standard are listed beneath each map. Layers are vertically stacked in order shown in legend and thus some "no exceeds" may be covered. For wells sampled more than once, the sample with the highest value is shown.

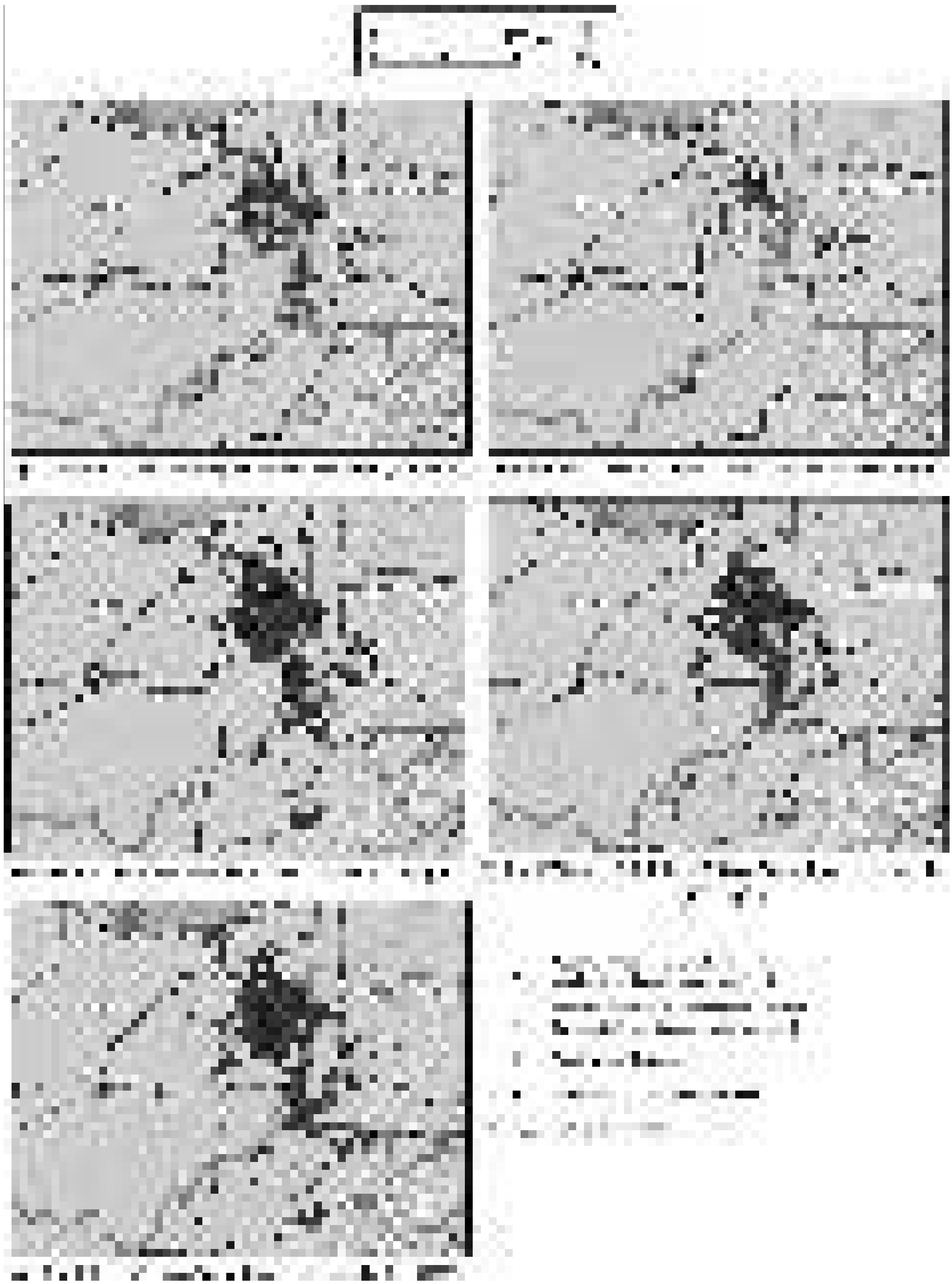

Fig. 7 (*continued*)

5.4. Hydrogeological relationships, regulatory exceedances, and hydrochemical water type

Piper trilinear diagrams for NE Pennsylvania (Fig. 8) and the Western Area (Fig. 9) show a geochemical evolution along flow paths from a Ca-HCO_3 groundwater type on recharging hilltops to sodium enriched waters along valley flanks and in valleys, where discharging groundwater with higher salinities naturally occur. Sodium derives from a combination of natural calcium and magnesium ion exchange with sodium found in clays within the formation matrix and fracture surfaces.

All major-ion water types occur in valley and upland settings in both NE Pennsylvania and the Western Area. But in upland areas in NE Pennsylvania (where deep freshwater groundwater circulation

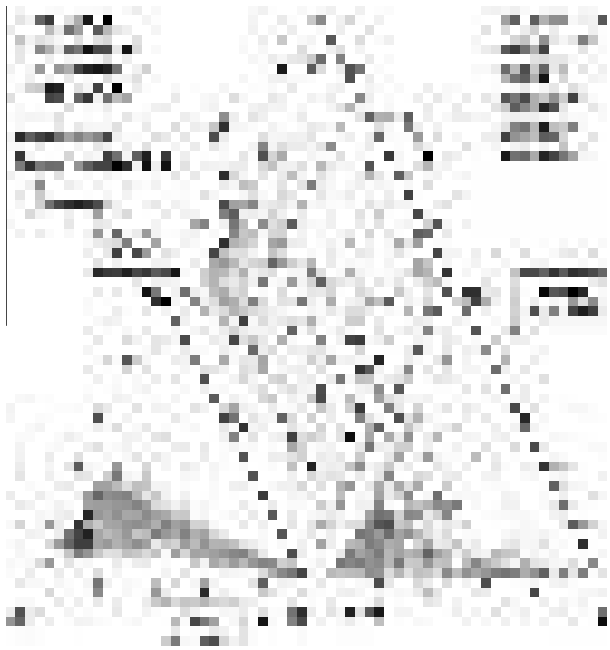

Fig. 8. Piper trilinear plot of pre-drilling samples in Northeastern Pennsylvania in four categories of sodium concentration. Circles in the diamond field are scaled to the concentration of TDS in the sample. The samples included here are known to be untreated and are within the ±7% criteria on the ion charge balance ($n = 2610$). Water types are after Deutsch (1997).

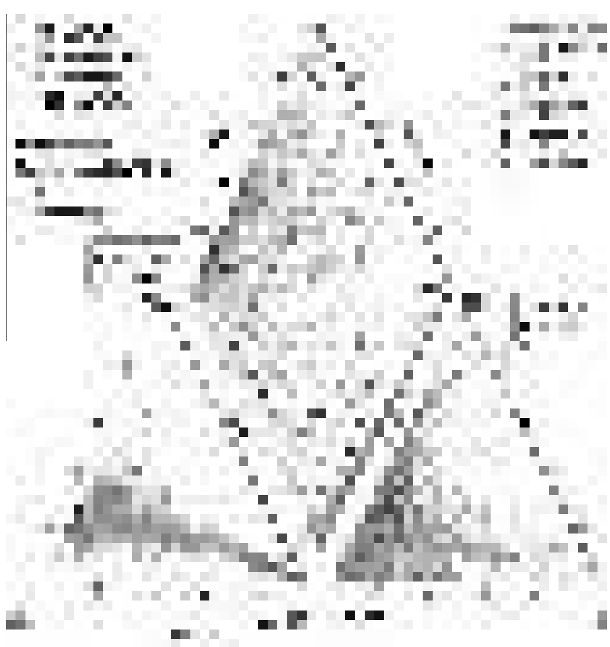

Fig. 9. Piper trilinear plot of pre-drilling samples in the "Western Area" in four categories of sodium concentration. Circles in the diamond field are scaled to the concentration of TDS in the sample. The samples included here are known to be untreated and are within the ±7% criteria on the ion charge balance ($n = 3439$). Water types are after Deutsch (1997).

patterns have developed), most water wells are too shallow to encounter Na-HCO_3/Na-Cl water types. In NE Pennsylvania about 88% of water from Carboniferous-age Burgoon Sandstone and Huntley Mountain formations are Ca-HCO_3 dominated water types. Most of these water wells occur in upland recharge areas. In contrast, the Devonian-age Lock Haven and Catskill formation samples mostly occur in lowland areas (Shultz, 1999) and have more Na-Cl and Na-HCO_3 water types (combined at 22.1%) than the stratigraphically and topographically higher aquifers. The more frequent occurrence of Na-Cl groundwater type in the Lock Haven and Catskill formations is consistent with what Williams et al. (1998) found. The Lock Haven and Catskill formations produce groundwater of a Na-Cl type from naturally saline intervals that Williams et al. (1998) termed "*restricted flow zones*" that are not being continuously flushed by active circulation of fresh groundwater (Risser et al., 2013). In the Western Area freshwater flow systems are shallower due to smaller topographic relief, and wells drilled in both valley and upland settings are equally likely to encounter all water types. There, water type and water quality do not associate with geological formation (except where carbonates and coal formations occur, which add sulfate).

In NE Pennsylvania, lithium and sodium exceeding water-quality standards are more likely to be found in water wells drilled into the Catskill and Lock Haven formations than in others. Likewise, arsenic and lead exceeding standards are more likely to be found in water wells drilled into the Burgoon Sandstone and "Other" formations. Table 5 shows common water-quality exceedances related to samples collected from water wells completed into these formations in NE Pennsylvania. Detailed summary of the sample counts (and exceedances) for each parameter for each geological unit is provided in Table S8 in the Supplemental Information.

Table 6 provides a comparison of common water-quality guideline exceedances from water wells to their geological completion formations in the Western Area. Overall, water wells drilled into the Allegheny-Pottsville Formation have the poorest water quality followed by the Other formations and the Monongahela Group. A detailed summary of the sample counts (and exceedances) for each parameter for each geological unit is provided in Table S9 in the Supplemental Information.

The groundwater type (hydrochemical facies) is closely associated with water-quality exceedances. For purposes of this study we have categorized the water types based upon the dominant single major anion and cation. However, many of the water types actually consist of mixtures of anions and cations which often show as mixed water types such as calcium-sodium-bicarbonate (Ca-Na-HCO_3) or Ca-Mg-HCO_3.

Exceedances of water quality criteria also relate to specific geochemical water types. In NE Pennsylvania more than 50% of the pre-drilling samples with Na-Cl type groundwater exceed drinking-water guidelines for manganese, TDS, and sodium (Table 7). For calcium sulfate (Ca-SO_4) type waters, 50% of samples exceed water-quality guidelines for iron, manganese, sulfate, sodium, and TDS. Table S10 in the Supplemental Information provides detailed sample counts (and exceedances) for each parameters compared to groundwater type.

All but one SMCL and RSL exceedance for chloride and strontium, respectively, occurred in either Na-Cl or Ca-Cl type groundwater. The majority of sulfate SMCL exceedances occur in $CaSO_4$ or Other water types. The Other water types (more than one sample) were magnesium bicarbonate (Mg-HCO_3), magnesium sulfate (Mg-SO_4), and sodium sulfate (Na-SO_4). The Mg-HCO_3 is consistent with shallow recharging groundwater in carbonates, and the latter two types could be reflective of unusual local geochemical processes involving ion exchange, carbonate geochemistry, and/or sulfur oxidation.

In the Western Area (Table 8), iron, manganese, sulfate, and TDS exceed guidelines in over 50% of Ca-SO_4 type groundwater. Over 75% of the samples exceed guidelines for sodium in all water types except Ca-HCO_3 and Ca-Cl. The Ca-SO_4 water type is associated with groundwater sourced from coal-containing formations. The Other water types consist of Mg-HCO_3, Mg-SO_4, and Na-SO_4. The

Table 5
Water-quality exceedances based upon water wells drilled into each geological unit – Northeastern Pennsylvania.

Percent of sample count that exceeds drinking water guideline or standard (%)	Burgoon Sandstone Formation (184)	Huntley Mountain Formation (387)	Catskill Formation (4983)	Lock Haven Formation (4780)	"Other" formations (33)
>1–5	Sodium	Arsenic, lead, sodium	Arsenic, barium, chloride, lead, pH < 6.5, pH > 8.5, TDS	Arsenic, barium, chloride, lead, pH < 6.5, pH > 8.5, sulfate	Lead
>5–10	Arsenic	–	–	TDS	Arsenic, lithium, sodium
>10–25	Lead, pH < 6.5	Manganese, pH < 6.5	Iron, lithium, manganese, turbidity	Lithium, turbidity	
>25–50	Turbidity, iron	Turbidity, iron	Sodium	Iron, manganese	Iron, pH < 6.5, turbidity
>50–75	Manganese	–	–	Sodium	Manganese
>75–100	–	–	–	–	–

Notes:
–: No parameters in this category.
Only parameters with exceedances greater than 1% of sample count included.
"Other" formations category includes: Mauch Chunk Formation, Timmers Rock Formation, Bloomsburg and Mifflintown formations, and Allegheny and Pottsville formations. Number in parenthesis after geological formation are total sample counts from each geological unit.

Table 6
Water-quality exceedances based upon water wells drilled into each geological unit – "Western Area".

Percent of sample count that exceeds drinking water guideline or standard (%)	Allegheny and Pottsville Formation (2952)	Conemaugh Group (2527)	Dunkard Group (440)	Monongahela Group (88)	"Other" formations (83)
>1–5	Arsenic, chloride, lead, lithium, pH < 6.5, pH > 8.5	Arsenic, chloride, lead, pH < 6.5, pH > 8.5, sulfate	Arsenic, chloride, pH < 6.5	Cadmium, chloride, pH < 6.5, pH > 8.5	Chloride, lead, pH < 6.5
>5–10	Sulfate	–	Lead, pH > 8.5, TDS	Lead, sulfate	–
>10–25	–	TDS, turbidity	–	Manganese, turbidity	Arsenic, sulfate
>25–50	TDS, turbidity,	Iron, manganese, sodium	Iron, manganese, sodium, turbidity	Iron, TDS	TDS
>50–75	Iron, manganese, sodium	–	–	Sodium	Iron, manganese, sodium, turbidity
>75–100	–	–	–	–	–

Notes:
–: No parameters in this category.
Only parameters with exceedances greater than 1% of sample count included.
"Other" formations category includes: Kanawha Formation; Ohio Shale; Berea Sandstone and Bedford Shale, undivided; and Maxville Limestone and Rushville, Logan, and Cuyhoga formations, undivided. Number in parenthesis after geological formation are total sample counts from each geological unit.

Table 7
Water-quality exceedances by groundwater type – Northeastern Pennsylvania.

Percent of sample counts in water type that exceeds drinking water guideline or standard (%)	$Ca-HCO_3$ (4582)	$Na-HCO_3$ (1591)	Na-Cl (413)	Ca-Cl (152)	$Ca-SO_4$ (98)	"Other" (29)
1–5	Arsenic, lead, pH < 6.5, TDS	Lead, TDS	Lead, pH < 6.5, strontium	Arsenic	Selenium	Chloride, pH < 6.5
>5–10	Barium	Arsenic, barium, pH > 8.5	pH > 8.5	Barium, chloride, lead	Arsenic, lead	–
>10–25	Iron, sodium, turbidity	Iron, turbidity	Arsenic	Turbidity, pH < 6.5, TDS	pH < 6.5	–
>25–50	Manganese	Manganese	Barium, chloride, iron, turbidity	Iron, manganese	Turbidity	Iron, manganese, sulfate, turbidity
>50–75	–	–	Manganese, TDS	Sodium	Iron, manganese, sulfate, TDS	Sodium, TDS
>75–100	–	Sodium	Sodium	–	Sodium	–

Notes:
–: No parameters in this category. Number in parenthesis is sample count in each category.
Only parameters with exceedances at least 1% of sample count included.
Only samples with ion charge balance within ±7% included.
"Other" groundwater types include: $Mg-HCO_3$, Mg-Cl, $Mg-SO_4$, and $Na-SO_4$.

Table 8
Water-quality exceedances by groundwater type – "Western Area".

Percent of sample count in water type that exceeds drinking water guideline or standard (%)	Ca-HCO_3 (4582)	Na-HCO_3 (1674)	Na-Cl (143)	Ca-Cl (236)	Ca-SO_4 (311)	"Other" (96)
1–5	Arsenic, lead, sulfate	Arsenic, lead, sulfate	Sulfate	Arsenic, lead, sulfate, lithium, pH < 6.5	Arsenic, lead	Arsenic, cadmium, chloride, pH > 8.5
>5–10	–	–	Barium, lead, pH < 6.5, pH > 8.5	–	Lithium	Lead
>10–25	TDS	Iron, manganese, pH > 8.5, TDS, turbidity	Lithium, turbidity	–	pH < 6.5	Lithium, pH < 6.5
>25–50	Turbidity, sodium	–	Iron, manganese	Chloride	Sodium	Iron, manganese, turbidity
>50–75	Iron, manganese	–	Chloride	Iron, manganese, turbidity	Sulfate, TDS, turbidity	Sulfate, TDS
>75–100	–	Sodium	Sodium, TDS	Sodium, TDS	Iron, manganese	Sodium

Notes:
–: No parameters in this category. Number in parenthesis is sample count in each category.
Only parameters with exceedances at least 1% of sample count included.
Only samples with ion charge balance within ±7% included.
"Other" groundwater types include: Mg-HCO_3, Mg-Cl, Mg-SO_4, and Na-SO_4.

Mg-HCO_3 is consistent with shallow recharging groundwater in carbonates, and the latter two types could be reflective of unusual local geochemical processes involving ion exchange, carbonate geochemistry, and sulfur oxidation.

The specific number and percentages of water-quality exceedances for each groundwater type are provided in Table S-11 the Supplemental Information for the Western Area.

5.5. Topographic position relationship to groundwater quality

5.5.1. Northeast Pennsylvania

Williams et al. (1998) found that groundwater with higher concentrations of sodium, chloride, barium, and strontium was associated with water wells located in valley settings in NE Pennsylvania. Table S12 in the Supplemental Information provides a summary of groundwater type per topographic position for NE Pennsylvania and the Western Area.

In NE Pennsylvania, we found the large preponderance (85.7%) of the pre-drilling samples with Na-Cl water type occurring in water wells completed in valley settings with the next greatest water type percentage in valleys being Na-HCO_3 water type (53%). However, depending upon location, well depth, and specific fractures intersected, it is possible to intersect younger and local/intermediate flow systems that contain Ca-HCO_3 or Na-HCO_3 water types.

The pre-drilling samples in NE Pennsylvania show that samples exceeding water-quality guidelines for barium, chloride, and strontium (87.1%, 86.3%, and 100%, respectively) typically occur in water wells found in valley settings. In contrast, most sulfate exceedances (91.4%) occur in water wells completed in upland settings where coal beds are more prevalent and waters are better oxygenated.

Approximately 83% of the water wells drilled to depths greater than 61 m occurred in upland areas. Table S-13 in the Supplemental Information provides a summary of individual water-quality exceedances by topographic setting.

5.5.2. Western area

For the Western Area, as noted in Table S-12 in the Supplemental Information, there seems to be little relationship between water type and topographic position. Potable groundwater circulation in the Western Area extends only a few tens of meters deep and most local to intermediate scale groundwater circulation paths are a few kilometers in length. Most water types occur at a percentage close to the overall percentage of samples in each topographic category (69.9% uplands, 30.1% valleys), but do not necessarily correlate. Na-Cl type groundwater type seems to be more likely in valley settings (37.4%), but the relationship between water quality and topography remains unclear, even with this large dataset.

There also is no substantial relationship between water-quality exceedances and topographic position for the Western Area, with the possible exception of barium and lithium. Barium exceedances of water-quality standards are higher in valley settings (63.0%) compared to upland settings (37.0%). Lithium exceedances are higher in upland settings (82.8%) than valley settings (17.2%). In the Western Area lower topography and dissected terrain coupled to coal seams leads to an inherently more complicated hydrochemical setting compared to NE Pennsylvania, and finding little relationship between water quality types with topographic settings is expected.

Approximately 91% of the water wells drilled to depths greater than 61 m occurred in upland areas. The specific number and percentages of water-quality exceedances for each parameter per topographic setting (upland versus valley) are provided in Table S-14 in the Supplemental Information.

6. Discussion

Our results, based on a dataset that is orders of magnitude greater than previous studies on quality of shallow groundwater of Appalachia, agree broadly with the results of most prior studies. The general pattern of geochemical evolution with their commensurate exceedances of water-quality standards is clear. For example, Na-Cl water in NE Pennsylvania is associated with water-quality standard exceedances for many parameters. These exceedances occur in valley settings and low-lying areas.

Fig. 10, as an example, shows that most (88.8%) of the Na-Cl groundwater type (pink dots) with higher total dissolved solids occur in major river valleys and associated tributaries.

In the Western Area, high salinity may occur in wells located both on valley slopes and even in upland areas where coal seams occur with oxidizable pyrite (Razem and Sedam, 1985). The combination of sulfide oxidation, ion-exchange, carbonate dissolution

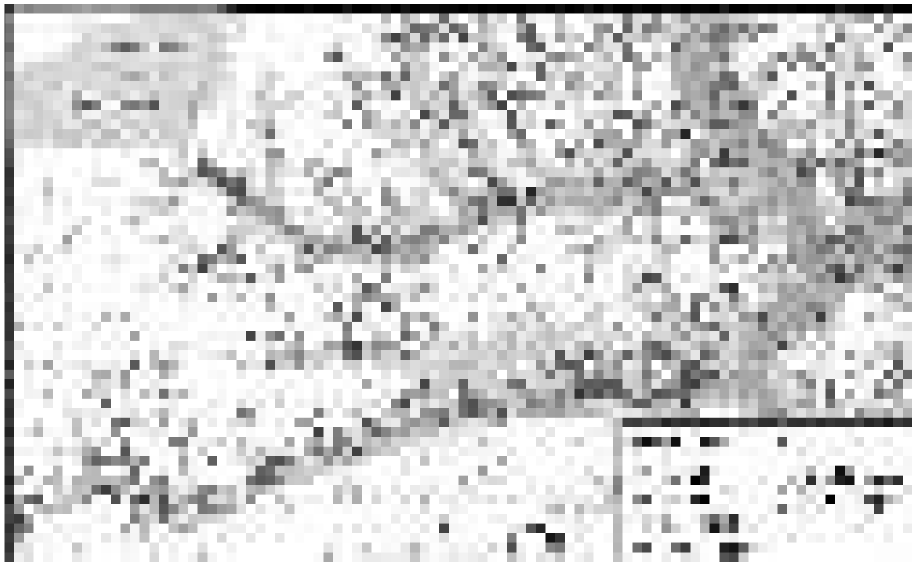

Fig. 10. Relationship between total dissolved solids (TDS) concentration, water type, and topography shown for a detail area in Bradford County, Northeastern Pennsylvania. TDS concentration is represented by symbol size. Water type (based on a simple classification using the dominant major anion and cation present in the sample) is represented by the color of the symbol as described in the legend. Streams and generalized topographic elevations are shown in gray for reference. Samples of Na-Cl and Na-HCO_3 water types are more prevalent in valleys and are associated with higher TDS concentrations.

and the intersection of deeper saline waters produce a broad range of water types. These processes are well known in the Appalachian Basin (Razem and Sedam, 1985). However, the same overall processes of chemical evolution should apply, as water types shift from Ca- to Na-dominated with increased age and residence time.

In both regions, the following reaction largely control the geochemical evolution of the groundwater.

Ion exchangeable clays in the aquifer rocks replace calcium and magnesium with sodium (Eq. (1)):

$$Ca^{2+} + Na_2\text{-Clay} = 2Na^{2+} + Ca\text{-Clay}. \tag{1}$$

This reaction, when driven to completion, result in Na-HCO_3 geochemical facies waters. When these waters mix with saline water near the bottom of deep wells (or restricted flow zones) or in discharge zones in valleys, the water becomes enriched even more in sodium and chloride, typically resulting in a Na-Cl type water.

In the Western Area, in particular, the oxidation of pyrite in coal leads to the generation of sulfate and hydrogen ions from sulfuric acid (e.g. Eq. (2)):

$$2FeS_2 + 7O_2 + 2H_2O = 2Fe^{2+} + 4SO_4^{2-} + 4H^+. \tag{2}$$

This in turn leads to oxidation of the ferrous iron species to ferric iron in solution or as a solid iron hydroxide or oxyhydroxide (e.g. Eq. (3)):

$$FeS_2 + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4^{2-} + 16H^+. \tag{3}$$

The acid generated then dissolves more carbonate and other minerals in the aquifer, increasing the calcium and other metals in solution, causing higher dissolved solids consisting of sulfate from the pyrite oxidation and calcium and alkalinity from the carbonate minerals.

Our study confirms prior understanding of the natural evolution of water quality in shallow aquifers in the Appalachian Basin. Groundwater quality is affected by a number of factors resulting in local variability related to different combinations of geologic formation and topographic position of the water well which, in turn, relates to the underlying position of the well in local, intermediate, and regional flow systems. Most water wells are completed as open holes and so may obtain water from multiple fractures, changing with hydrologic season and recharge events, which add to the variability of the groundwater chemistry.

In the future it may be possible to parse this large dataset into smaller regions to evaluate these relationships in more detail, either statistically or through geochemical modeling approaches. However, these exercises are beyond the scope of this paper, which was intended to re-evaluate previously identified water-quality variability using a much larger dataset. Overall, we see no broad changes in variability of chemical quality in this large dataset to suggest any unusual salinization caused by possible release of produced waters from oil and gas operations, even after thousands of gas wells have been drilled among tens of thousands of domestic wells within the two areas studied. What we find confirms what others have historically reported: there is wide variability in water quality in potable groundwater in the Appalachian Basin, and much of this water naturally exceeds regulatory standards. Those previous reports used much smaller datasets, and were published many decades earlier, thus confirming and supporting our conclusions.

7. Conclusions

The results of our study, from the analysis of 21,044 pre-drilling groundwater samples, have broad implications regarding the characterization of background chemistry in shallow groundwater supplies of the Appalachian Plateau, especially when assessing potential impacts from oil and gas operations. This very large dataset, in combination with previous studies, shows we can broadly conclude that in both NE Pennsylvania and the Western Area the groundwater commonly exceeds drinking-water guidelines, by means of natural processes. These exceedances are not random, but are related to factors such as geological formation/lithology, sample turbidity, age or residence time of the groundwater, and where within the groundwater flow system the sample was collected. There is also an association with groundwater type and TDS/salinity concentrations. In NE Pennsylvania there appears to be an association with water quality and topographic position (valleys versus uplands), whereas, in the Western Area this relationship is not as strong. Based on our study, 63.1% of the water wells sampled in NE Pennsylvania and 88.7% in the Western Area had pre-drilling exceedances for drinking-water guidelines for one or more parameters (including turbidity). In NE Pennsylvania, 10.2% of the samples exceeded one or more of USEPA's MCLs set for drinking-water supplies, 46.1% of the samples exceeded one or more of USEPA's SMCLs, and 6.7% exceeded one or more of USEPA's HALs or RSLs for tap water. In the Western Area 7.7% of samples exceeded one or more MCLs, 65.2% exceeded one or more SMCLs, and 14.6% exceeded one or more health advisory or regional screening levels for tap water.

Our results are neither unusual nor surprising and are consistent with previous results in both areas using much smaller datasets both before and after unconventional oil and gas development (e.g. Molofsky et al., 2013; Gross and Low, 2012; Chambers et al., 2012; Low and Galeone, 2006; Williams et al., 1998; Stoner et al., 1987; Razem and Sedam, 1985; Taylor, 1984; Matisoff et al., 1982).

Comparison of historical groundwater data from NE Pennsylvania that mostly pre-dates unconventional shale gas development (pre-2007) to sampling results from Chesapeake's pre-drill baseline program (2009–2012) shows that the current groundwater quality conditions are similar to those shown in historical data. This comparison thus provides a dataset representative of regional groundwater conditions that pre-dated unconventional shale gas drilling, and provides a comprehensive understanding of aquifer systems and groundwater chemistry in the region.

Fundamentally, water-quality data from domestic water wells need to be understood within the natural geochemical context when evaluating suspected water-quality changes from shale-gas methane production or any other source. The hydrogeological setting of the water well also needs to be fully understood within the context of the mineralogy in the aquifer, the topographic position of the well within the associated flow path, the natural variability of water-quality parameters, and the naturally expected hydrochemical facies. Basic principles of groundwater flow and the hydrochemical evolution of groundwater systems in Appalachia (e.g. Piper, 1933; Lohman, 1937, 1939; Poth, 1963) indicate that saline groundwater underlying fresher groundwater is connate. This connate groundwater was present in the sediments during deposition, thus the saline waters have not migrated long vertical distances from formations hundreds and thousands of meters deeper.

We see no broad changes in variability of chemical quality in this large dataset to suggest any unusual salinization caused by possible release of produced waters from oil and gas operations, even after thousands of gas wells have been drilled among tens of thousands of domestic wells within the two areas studied. What we do find confirms what others previously found with much smaller datasets decades earlier regarding the wide variability of water quality in domestic groundwater in the Appalachian Basin, and that much of this water naturally exceeds regulatory standards.

Finally, caution should be exercised when using total metals analysis to characterize water-quality changes of any type, given the common problems associated with sampling water from domestic wells, which may include turbidity. Samples should be filtered to remove suspended particles and dissolved metals should be analyzed to provide a more accurate analysis of metals concentrations.

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- During the preparation of this specific paper, all authors worked for the organizations whose affiliations are noted in authorship. Mark Hollingsworth is a current employee of Chesapeake Energy Company, having worked at Chesapeake from February 2011 to present. Prior to Mr. Hollingsworth's employment by Chesapeake, he worked for TestAmerica Laboratories, Inc. which provided laboratory analytical consulting services to Chesapeake. Bert Smith is a former employee of Chesapeake Energy having worked there from May 2012 to September 2013, and has been employed by Enviro Clean Products and Services from November, 2013 to the present. Enviro Clean P&S also does consulting work for Chesapeake. Prior to May, 2013 Mr. Smith worked for Science Applications International Corporation (SAIC), which did consulting work for Chesapeake Energy. AECOM provides architecture and engineering services to government and private industry around the world, including the energy sector and Chesapeake. Rikka Bothun worked for AECOM during most of the time this paper was under preparation, but left AECOM in December, 2014 and now works for a private consulting company that does not do consulting work for Chesapeake.
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Appendix A. Supplementary material

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Groundwater

Evaluation of Methane Sources in Groundwater in Northeastern Pennsylvania

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Abstract

Testing of 1701 water wells in northeastern Pennsylvania shows that methane is ubiquitous in groundwater, with higher concentrations observed in valleys vs. upland areas and in association with calcium-sodium-bicarbonate, sodium-bicarbonate, and sodium-chloride rich waters—indicating that, on a regional scale, methane concentrations are best correlated to topographic and hydrogeologic features, rather than shale-gas extraction. In addition, our assessment of isotopic and molecular analyses of hydrocarbon gases in the Dimock Township suggest that gases present in local water wells are most consistent with Middle and Upper Devonian gases sampled in the annular spaces of local gas wells, as opposed to Marcellus Production gas. Combined, these findings suggest that the methane concentrations in Susquehanna County water wells can be explained without the migration of Marcellus shale gas through fractures, an observation that has important implications for understanding the nature of risks associated with shale-gas extraction.

Introduction

Significant media attention has been focused on the potential for methane impacts in drinking water wells located within areas of hydraulic fracturing activities for shale-gas development. Distinguishing among the various sources of methane gas that may affect drinking water wells requires proper assessment of background conditions. In this study, we review the results of background methane and groundwater quality surveys, in conjunction with geologic and historical information, to develop a better understanding of the potential sources of methane levels in drinking water wells in Susquehanna County in northeastern Pennsylvania.

Susquehanna County has experienced substantial gas extraction activities in the Marcellus shale since 2006. Prior to that time, there was not a significant history of oil and gas operations in this county, thereby providing a unique opportunity to evaluate the potential effects of shale-gas extraction on groundwater resources in the Appalachian basin. Other researchers have suggested that elevated methane concentrations in water wells in Susquehanna County are the result of regional impacts from shale-gas extraction activities (e.g., Osborn et al. 2011). To test this hypothesis, we have evaluated data from the sampling and testing of 1701 water wells throughout Susquehanna County to assess the prevalence and distribution of methane concentrations in groundwater. We have also evaluated isotopic and molecular analyses of hydrocarbon gases in the Dimock Township of Susquehanna County, an area of focused sampling by the Pennsylvania Department of Environmental Protection (DEP) and the U.S. Environmental Protection Agency, to determine whether reported methane concentrations above the Pennsylvania DEP action level (7000 µg/L) in local water wells exhibit signatures consistent with Marcellus production gases, or overlying Middle and Upper Devonian gases sampled in annular spaces of local gas wells.

Our research indicates that shale-gas extraction has not resulted in regional impacts on groundwater quality in Susquehanna County, a finding which suggests that

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hydraulic fracturing is not responsible for the creation or enhancement of wide-spread pathways by which Marcellus shale gas can rapidly travel to the surface.

Methods

Our study focused on characterizing the geologic and hydrogeologic context of methane occurrence in water wells in northeastern Pennsylvania. For this purpose, we have collected and reviewed the following types of information: (1) geologic data on regional structure and stratigraphy, (2) studies on aquifer dynamics and geochemical characteristics, and (3) historical documentation regarding the occurrence of hydrocarbon gases in the shallow and deep subsurface. Within this context, we have evaluated data collected from 1701 water wells in Susquehanna County to determine the prevalence and distribution of elevated methane concentrations and other groundwater parameters. We have also assessed isotopic and molecular analyses of gases from 15 water wells sampled as part of an ongoing stray gas investigation in the Dimock Township to determine the origin of methane concentrations above the Pennsylvania DEP action level.

Geologic information discussed in this article was compiled from prior studies on the stratigraphy and structure of the northeastern portion of the Appalachian basin, as well as new data acquired during exploration activities associated with recent Marcellus shale gas development. Specifically, stratigraphic information recorded during the drilling of 24 shale-gas wells located in Susquehanna and Wyoming counties facilitated the development of the geologic cross-section presented in Figure 1b. The seismic line shown in Figure 1c is based on interpretation of 57 miles of 2D seismic data acquired by Evans Geophysical (2007). Historical notations regarding the presence of shallow gas shows, historical gas fields, and bubbling springs and/or water wells in northeastern Pennsylvania were gathered from a detailed review of over a dozen documents dating back to the early 1800s.

The 1701 water samples analyzed as part of an extensive predrill water well survey from 2008 through 2011 were collected in the field and analyzed in NELAC accredited laboratories in accordance with procedures detailed in the supporting information. Statistical analyses of predrill data were conducted using Statistical Software ProUCL 4.1.01, which was developed and provided to the public by the United States Environmental Protection Agency (US EPA). Molecular and isotopic analyses of gases in shale-gas wells and local water wells in the Dimock Township were collected as part of an ongoing stray gas investigation by the Pennsylvania DEP and Cabot Oil and Gas Corporation in 2009 through 2011 using procedures detailed in the supporting information. This article also reports molecular and isotopic data from dissolved gases in Susquehanna County water wells collected by Osborn et al. (2011) and the US EPA (2012). Details of sample collection and analyses of these samples are presented in the respective studies, for which citations are provided.

Regional Geologic and Hydrologic Conditions

Deep and Shallow Stratigraphy

Knowledge of naturally occurring sources of hydrocarbon gases in both shallow and deep stratigraphic units provides critical information for evaluating potential sources and migration pathways of methane into local water wells. Natural gas in the subsurface can be both *microbial* in origin, that is, formed from the microbial breakdown of organic material, and *thermogenic* in origin, that is, formed from the abiotic degradation of organic material in formations under high temperatures and pressures at depth. Microbial methane commonly occurs in shallow bedrock and surficial deposits of alluvium (sands, silts, gravel, peat, and clay) and glacial drift (glacial till and outwash) (Coleman 1994). In Susquehanna County, these deposits can range from a few meters thick in elevated areas to tens of meters thick in lowlands (Aber 1974; Sevon et al. 1975; Inners and Fleeger 2002; Braun 2006). In areas with organic-rich shale formations, recent studies have also suggested an association between microbial methane concentrations and groundwater with longer residence times (in which progressive microbially mediated redox reactions can lead to methane production) (Kresse et al. 2012).

Surficial deposits in Susquehanna County are underlain by the Catskill and Lock Haven Formations, a series of interbedded sandstone, siltstone, and shale deposits formed during the Upper Devonian period (359 to 385 million years ago) (Figure 1b) (ICS 2010). The Catskill and Lock Haven Formations have historically been explored for reservoirs of thermogenic natural gas. These Upper Devonian deposits attained deepest burial depths at approximately 250 million years ago, at which time organic matter entrained in discrete lenses and distributed throughout the formations attained temperatures sufficient to crack to thermogenic gas (Evans 1995). Such elevated past temperatures are evidenced by the vitrinite reflectance of organic material (greater than 2.0%) observed in seams and lag deposits currently exposed in numerous outcrops and sandstone (bluestone) quarries in Susquehanna County (Laughrey et al. 2004; Weatherford Laboratories 2011).

The Bradford Sands, a series of thick sandstone deposits, form the base of the Lock Haven Formation, and are underlain by the Brallier Formation (locally known as the Elk Formation) and the Trimmers Rock Formation, which are comprised of interbedded sandstone, siltstone, and shale stratum of Upper Devonian age (Carter and Harper 2002). The Tully limestone is the youngest unit of the Middle Devonian age strata, which include the Mahantango Formation, consisting of laminated siltstone, sandstone and shales, and the organic-rich Marcellus shale, estimated to contain as much as 127 trillion cubic feet of thermogenic gas (Engelder 2009). The volume of gas contained in the Marcellus shale represents over 5 million times the volume of natural gas consumed in the United States in 2011 (24.4 million cubic feet; EIA 2012).

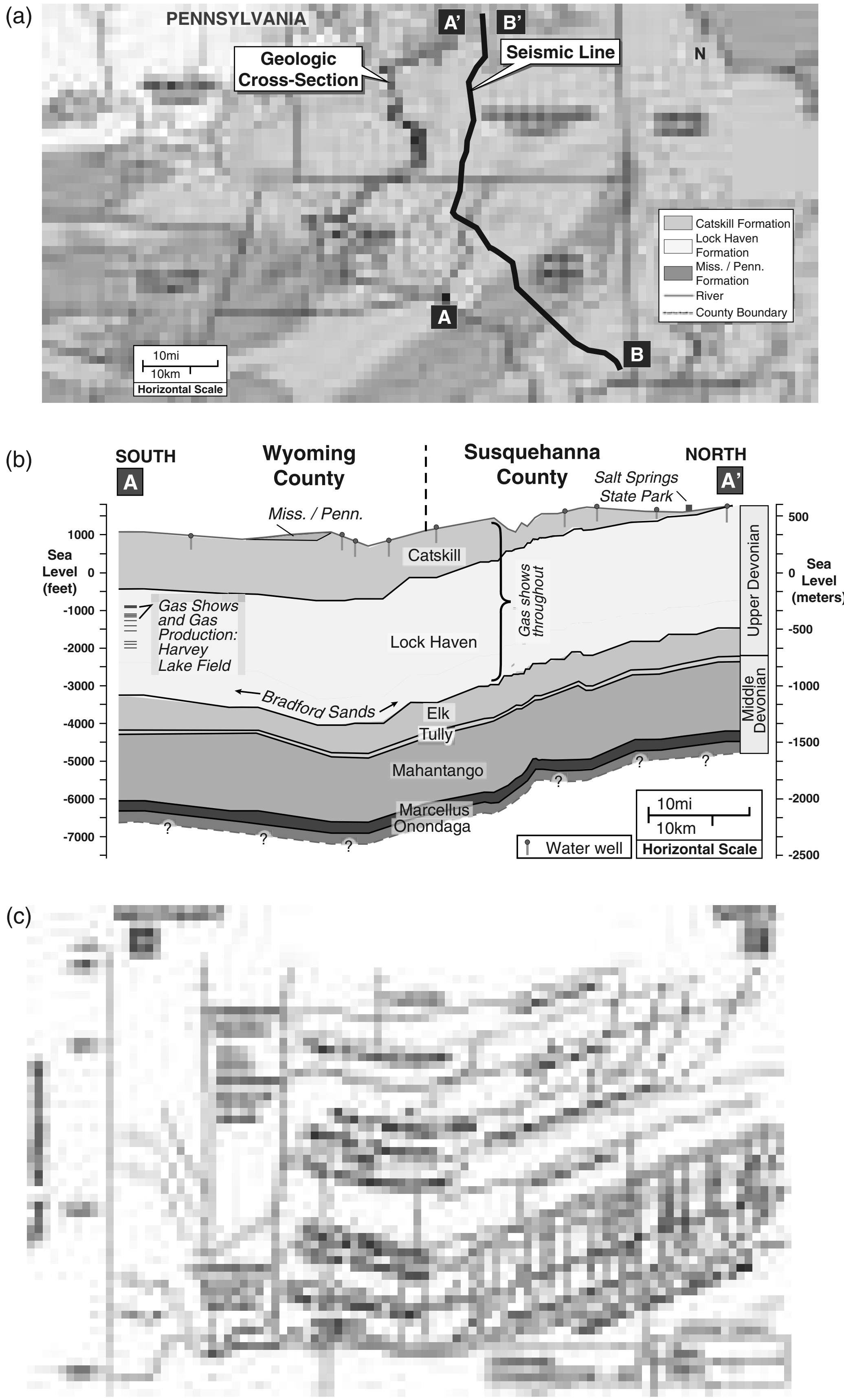


Figure 1. (a) Geographic map showing transect of wells in the cross section and seismic line. (b) Generalized cross section of Upper and Middle Devonian formations in Susquehanna and Wyoming County. (c) North-south 2D Seismic Line through Susquehanna, Wyoming, and Lackawanna County (Evans Geophysical 2007).

Fault and Fracture Systems

An extensive natural fracture network is present in both the Catskill and Lock Haven Formations, with penetrating north-south oriented vertical planar fractures bisected by multiple inferior fracture sets and bedding planes (Hollowell and Koester 1975; Geiser and Engelder 1983; Taylor 1984; Callaghan et al. 2010).

On a larger scale, seismic data provides valuable information regarding the structural setting and presence and/or absence of large-scale faults that could serve as conduits for the transport of fluids or gases. An interpretation of the stratigraphic and structural framework beneath Susquehanna and Lackawanna Counties from a portion of a regional 2D seismic line that transects the area is depicted in Figure 1c (Evans Geophysical 2007). The Upper Devonian section from the Catskill through the base of the Elk Sands shows shallow south-dipping stratigraphy interrupted by low-angle thrust faults that dip from 25° to 40° to the south. These thrust faults flatten and become bed parallel above the Tully limestone. In contrast, the Middle Devonian Marcellus to Silurian Salina Formations are interpreted to be bisected by frequent steep dipping (60° to 90°) reverse faults, back thrusts, and compressional and/or salt-related features that terminate near the top of the Marcellus or in the lower units of the Mahantango Formation in Susquehanna County (Davis and Engelder 1985; Scanlin and Engelder 2003). These faults rarely extend to the Tully stratigraphic level. This seismic section illustrates a disconnection between the distinct structural styles above and below the Tully horizon, interpreted to be a regional structural detachment that separates the shallow Catskill-Lock Haven stratigraphic section from the deeper Mahantango-Marcellus stratigraphic section. Given the presence of overpressured conditions in the Mahantango Formation and absence of such pressure above the Tully Limestone, this newly observed detachment zone, or the Tully Formation itself, is interpreted to act as a restrictive barrier to the present-day upward movement of deep formation fluids and methane from gas-charged units in the Middle Devonian Mahantango and Marcellus Formations in Susquehanna County.

Aquifer Dynamics/Water Well Construction

The great majority of water wells in Susquehanna County penetrate the fractured Catskill Formation, in which groundwater flow in unweathered bedrock appears to be controlled by secondary permeability primarily through vertical to near vertical north-south oriented fractures (as indicated by water seepage and flow, iron staining, and mineral precipitation) (Hollowell and Koester 1975; Geyer and Wilhusen 1982; Geiser and Engelder 1983; Taylor 1984). Most of the bedrock water wells are unsealed open-hole completions, with casing terminating in the shallow bedrock in order to draw water from multiple horizons at typical depths of 100 to 500 feet beneath ground surface (Lohman 1937, 1939; Taylor 1984; PaGWIS 2011). Median yields of 1146 water wells in the Catskill Formation were reported to be 0.76 and 2.2 L/s (12 and 35 gal./min.) for domestic and nondomestic wells, respectively (Taylor 1984). Valleys and streams in Susquehanna County commonly follow a pattern coincident with the presentation of vertical cross-cutting joint sets and lineaments, dense fractures, and fold trends (e.g., the orthogonal drainage patterns of Wyalusing Creek, a tributary of the Susquehanna River) (Taylor 1984; Engelder et al. 2009). Salt springs are also frequently located along linear trends associated with faulting or fracturing, suggesting the potential for connection to the deeper, and more brackish, Lock Haven Formation (Lohman 1937).

Historical Evidence for Occurrence of Shallow Natural Gas

Historical documentation suggests that the presence of methane gases in the shallow subsurface has been observed for over 200 years in Susquehanna County, long before the expansion of shale-gas fracturing in this area in 2006. For example, there are several dozen instances of flammable effervescing springs and water wells dating back to the late 1700s (Blackman 1873; Lohman 1939; Soren 1963; Breen et al. 2007; Susquehanna County Historical Society 2010; Williams 2010; Table S7 for a full list of citations). In addition, water well drillers have frequently reported encountering gas during drilling, particularly in valleys and other low-lying areas (Bell Brothers Well Drilling, Creswell Drilling, Beavers Well Drilling, Karp & Sons Drilling, JIMCON Drilling, Drake Drilling, personal communication, 2010).

Several gas fields in the past century have produced from formations less than 3000 feet below surface in northeastern Pennsylvania (e.g., Shrewsbury Gas Field, Lovelton Gas Field, Harveys Lake Gas Field), and there are numerous reports of gas shows between 80 and 800 feet during the drilling of oil and gas wells (Ashley and Robinson 1922; Hopbottom Well Record 1957; Soren 1961). For example, the 1881 publication "The Geology of Susquehanna and Wayne County" reported significant volumes of gas during the drilling of an oil boring in the Catskill Formation to a depth of 680 feet (White 1881). The presence of methane in natural springs and water wells has also been cited in adjacent New York State, where a survey of 239 water wells from 1999 to 2011 showed that 9% of water wells contained dissolved methane concentrations exceeding 10 mg/L (Kappel and Nystrom 2012). Due to potential gas contamination from natural sources, guidelines issued by the Pennsylvania DEP and other state agencies recommend the routine venting of water wells (PA DEP 2004).

Results of Sampling and Testing Programs

In addition to researching the hydrogeology, geology, and history of Susquehanna County in an effort to better understand natural groundwater conditions, we have also collected and analyzed water samples from this area to assess the presence of methane, ethane, and other water quality parameters. The results of this analysis provide additional information with regards to the distribution of hydrocarbon gases, and their association with physical geography, gas production areas, and groundwater quality.

Predrill Methane Data

This article evaluates data from 1701 water wells sampled in Susquehanna County to characterize baseline groundwater conditions during the period of 2008 to 2011. Over 900 of these samples were collected in accordance with Pennsylvania DEP regulations (Oil and Gas Act 13), which stipulates that an operator must conduct a "predrill or prealteration" water survey prior to the drilling of each gas well in order to maintain the right to contest any subsequent claims of groundwater impact. These samples were analyzed for both dissolved gas concentrations and general water quality parameters related to primary and secondary drinking water standards. Several hundred additional water well samples were collected from an 80-mile^2 area within Brooklyn, Harford, and Gibson Townships (southeast Susquehanna County), which did not have significant gas development operations at the time. These samples were intended to establish baseline methane concentrations in groundwater in areas of Susquehanna County without substantial gas extraction activities and were primarily analyzed for concentrations of dissolved gases (methane, ethane, and propane).

Collectively, these 1701 samples are referred to as "predrill" because they were sampled prior to the drilling of specific *proposed* wells; however, several of the water wells sampled were located in close proximity to *previously* drilled gas wells. As a result, 322 of the water wells have been characterized as located in a "gas production area" (defined as an area within 1 km of gas wells drilled prior to the time of sampling), while 1379 of the water wells are considered to be located in a "nonproduction area" (i.e., with no gas well drilled prior to sampling within 1 km). The 1 km radius used here to define gas production areas is considered a conservative distance, consistent with the radius utilized by Osborn et al. (2011) in northeastern Pennsylvania and New York, and larger than the current predrill sampling radius (2500 feet = 0.76 km) stipulated by the Pennsylvania DEP. Combined, 78% of the 1701 water wells sampled in either gas production or nonproduction areas contained detectable concentrations of methane, with approximately 3.4% exceeding the 7000 µg/L level at which corrective action is recommended by the Pennsylvania DEP. (As a note, the saturation concentration for methane is approximately 28,000 µg/L at atmospheric pressure, though it can vary with groundwater temperature, pressure, and salinity.)

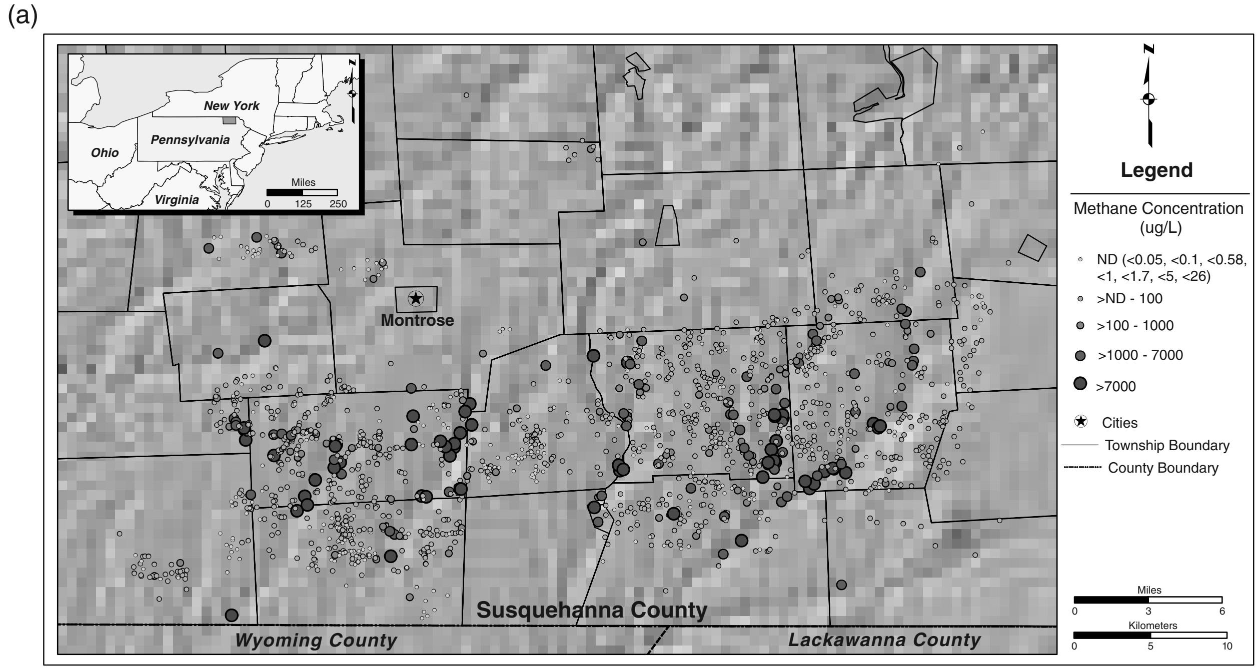
Predrill Methane Data: Valleys vs. Uplands

To evaluate the possibility of a relationship to topography, dissolved methane concentrations measured in water wells were plotted on a light detection and ranging (LiDAR) bare-earth elevation map overlain with the National Hydrography Dataset (NHD). As indicated in Figure 2a, the distribution of methane concentrations appears to be correlated to surface topography, with the higher dissolved methane concentrations located in topographic lows (valleys).

To support statistical analysis of this observed correlation, the water well population was subdivided into wells located in valleys (defined as the area within 1000 feet of a major NHD flowline or 500 feet of minor tributaries to NHD flowlines) vs. wells located in uplands (greater than 1000 feet or 500 feet from a major or minor drainage, respectively). As shown in Figure 2b, the median concentrations of the two datasets differ moderately (0.67 vs. 0.34 µg/L of methane in lowland vs. upland water wells, respectively). However, the 90th percentile concentrations differ much more significantly (1800 vs. 17 µg/L of methane in lowland vs. upland water wells, respectively), indicating that upper range methane concentrations occur considerably more frequently in valley wells. Furthermore, although valley wells represent only 51% of the total water well population, they comprise 88% of the water wells with methane concentrations that exceed 7000 µg/L (the current Pennsylvania DEP action level). A one-way Mann-Whitney U Test supports a statistically significant difference between methane levels in lowland (865 locations) vs. upland water wells (836 locations), where valley water wells contain statistically elevated methane concentrations (p-value < 0.001).

No Regional Association of Methane with Gas Production

In order to evaluate whether elevated methane concentrations in the data set exhibited a relationship to gas development activities, methane concentrations in water wells located in gas production areas were compared to those located in nonproduction areas. Of the water wells in gas production areas (322 wells), approximately 3.7% contained methane concentrations that exceeded 7000 µg/L, while 3.3% of water wells in nonproduction areas (1379 wells) contained methane concentrations above 7000 µg/L. This slight difference may be attributable to the fact that gas production areas contain a greater percentage of valley water wells (61% of 322 water wells) than nonproduction areas (49% of 1379 water wells). To evaluate whether the prevalence of valley water wells in gas-production areas was a factor, the methane concentrations of valley water wells in gas production areas were exclusively compared to those of valley water wells in nonproduction areas. Using a one-way Mann-Whitney U-test, methane concentrations in valley gas-production valley areas were found to be less than or equal to those in valley nonproduction valley areas (p-value = 0.007). A similar comparison of methane concentrations in upland water wells showed that methane concentrations in upland gas production areas were approximately equal (i.e., no statistically significant difference was observed) to those in upland nonproduction areas (two-way Mann-Whitney U-test p-value = 0.154). Furthermore, as a whole, no statistically significant difference was observed between methane concentrations



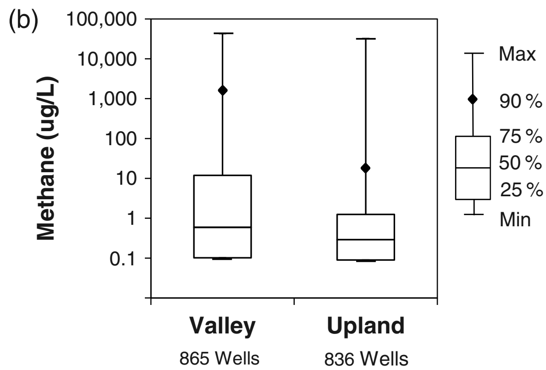


Figure 2. (a) LiDAR bare-earth elevation map showing dissolved methane concentrations from 1701 "predrill" water wells sampled in Susquehanna County. (b) Population distribution of methane concentrations in "predrill" water samples from valley and upland water wells in Susquehanna County.

in water wells in gas production areas vs. water wells in nonproduction areas (regardless of topographic location) (two-way Mann-Whitney U-test p-value = 0.503).

Origin of Methane Concentrations in Water Wells

Determining the microbial vs. thermogenic (or mixed) origin of methane concentrations in the predrill water well samples can narrow the list of potential sources. One conventional approach to assessing thermogenic vs. microbial origin relies upon the ratios of methane to ethane, where gases with ratios less than 100 have been characterized as thermogenic, while those with ratios greater than 1000 have been characterized as microbial (with ratios between 100 and 1000 typically characterized as an indeterminate origin or a mixture of thermogenic/microbial gases) (Bernard et al. 1976; Schoell 1980; Taylor et al. 2000).

However, comparison of methane to ethane ratios to available isotopic data in predrill water well samples in nearby Bradford County (located directly west of Susquehanna County) suggest that the conventional approach for characterizing thermogenic vs. microbial origin based on methane to ethane ratios may not apply in the geologic context of northeastern Pennsylvania. Specifically, Baldassare (2011) observed methane to ethane ratios ranging from 55 to 6900 in over a dozen water well samples from Bradford County that displayed traditionally thermogenic carbon and hydrogen isotopic signatures (i.e., $\delta^{13}C > -50‰$, $\delta^{2}H > -250‰$). This suggests that these water wells could contain a notable thermogenic component, despite their traditionally microbial methane to ethane ratio.

In Susquehanna County, 1540 (91%) of the 1701 predrill water well samples were analyzed for both methane and ethane concentrations. Exactly 217 of these samples contained *detected* methane and ethane, and of those samples, approximately 72% (156 samples) displayed ratios of methane to ethane greater than 1000 (Figure S2). Isotopic analyses were not performed on these predrill samples. However, several water wells in Susquehanna County were sampled as part of a stray gas investigation after nearby shale-gas extraction activities (section *Discussion* below of isotopic and molecular analyses in the Dimock Township). Samples from three of these water wells exhibited classically microbial (i.e., greater than 1000) methane to ethane ratios but traditionally thermogenic carbon and hydrogen isotopic signatures. Consequently, at this time, the thermogenic, microbial, mixed thermogenic/microbial, or microbially altered (i.e., oxidized) origin of the predrill dissolved gas samples from water wells cannot be determined.

Predrill Data: Correlation Between Methane and Other Groundwater Quality Parameters

To assess potential differences in groundwater sources, predrill water well samples in Susquehanna County were classified with regard to the relative concentrations of major cations (calcium, magnesium, potassium, and sodium) and anions (bicarbonate, chloride, and sulfate). The cation/anion composition of the groundwater is a product of the aquifer matrix (e.g., sandstone vs. shale), groundwater residence and/or rock-water interaction times, and the prominence of redox processes (Kresse et al. 2012). Consequently, for groundwater containing elevated methane levels, the water type can prove useful to understanding the flow path and retention time of water within the subsurface (i.e., stratum(s) of origin).

The Piper diagrams shown in Figure 3a and 3b plot the results for the population of 408 predrill samples for which all of the major cations and anions were analyzed. As shown, each water sample is characterized with regards to water "type" based upon the relative milliequivalent concentrations of major ions. In this sample population, five predominant water types have been identified: Ca-HCO_3 (281 samples), Ca-Na-HCO_3 (46 samples), Na-HCO_3 (20 samples), Ca-HCO_3-Cl (32 samples), and Ca-Na-HCO_3-Cl (20 samples). To characterize the relationship between water type and dissolved methane concentrations, the distribution of methane concentrations has been determined for each of the five principal water types. As shown in Figure 3b, the percentage of water samples exceeding 1000 µg/L of dissolved methane differs dramatically among water types, with 11 and 30% of the samples matching Ca-Na-HCO_3 and Na-HCO_3 water types, respectively, exceeding this level. By contrast, none of the 281 Ca-HCO_3 water type samples exceed 1000 µg/L of methane. Only a limited number of groundwater samples matched the Na-HCO_3-Cl and Na-Cl water types (four samples total); nevertheless, these samples exhibit several of the highest methane concentrations observed in the predrill dataset, with three of the four reported concentrations exceeding 10,000 µg/L.

Concentrations of individual groundwater quality parameters measured during predrill sampling were also evaluated to assess a possible correlation with dissolved methane concentrations. These data provide a comprehensive data set for the following parameters: alkalinity, barium, calcium, chloride, magnesium, manganese, pH, potassium, sodium, strontium, sulfate, and TDS. Data for an additional eight groundwater analytes (i.e., arsenic, aluminum, boron, bromide, iron, lead, nitrate, and sulfide) contained greater than 50% nondetect values and were therefore not amenable to this analysis (Figure S1).

To detect possible water quality trends related to methane, the concentration distribution for each parameter was determined for four ranges of methane concentrations: nondetect, detected to 1000 mg/L, greater than 1000 µg/L to 7000 µg/L, and greater than 7000 µg/L. The median concentrations of the following parameters were found to increase with increasing methane levels: barium, chloride, manganese, pH, sodium, strontium, and TDS (Figure S1). Median concentrations of alkalinity and potassium showed slight increases with increasing methane concentration, while median magnesium concentrations exhibited a minor decrease. Median sulfate and, to a lesser degree, calcium concentrations in groundwater were observed to inversely correlate with dissolved methane concentrations (i.e., lower sulfate at higher dissolved methane levels).

Dimock Township: Stray Gas Investigation

Isotopic and molecular analyses of dissolved gases can provide a means to differentiate between: (1) microbial vs. thermogenic gas sources, (2) distinct sources of microbial gas, and (3) distinct sources of thermogenic gas (e.g., Fuex 1977; Schoell 1980; Clayton 1991; Coleman 1994; Wiese and Kvenvolden 1994; Baldassare and Laughrey 1997). In this study, we have evaluated isotopic and molecular data from 7 gas wells, and 15 water wells sampled by the Pennsylvania DEP and Cabot Oil and Gas Corporation in the Dimock Township of Susquehanna County as part of a site-specific stray-gas investigation in 2009 through 2011. In addition, we reviewed data from 9 water wells sampled by researchers from Duke University (Osborn et al. 2011) and 11 water wells sampled by the USEPA (USEPA 2012) in the same area from 2010 through 2012. One gas sample from a bubbling salt spring approximately 13 miles northeast of Dimock, in nearby Franklin Township, was also included in our evaluation. The goal of the evaluation presented here is to utilize available molecular and isotopic data to assess whether gases in Dimock area water wells were originating from the Marcellus Shale or overlying Middle and Upper Devonian deposits.

Potential Source Gases: Isotopic and Molecular Analyses, 2009 to 2011

In 2009 and 2010, several water wells in the Dimock Township of Susquehanna County were reported to contain methane concentrations above the Pennsylvania DEP action level of 7000 µg/L (PA DEP 2009). To characterize potential proximate sources of gas migration, the Pennsylvania DEP conducted molecular and isotopic analyses of gas samples from the well cellar, the annular

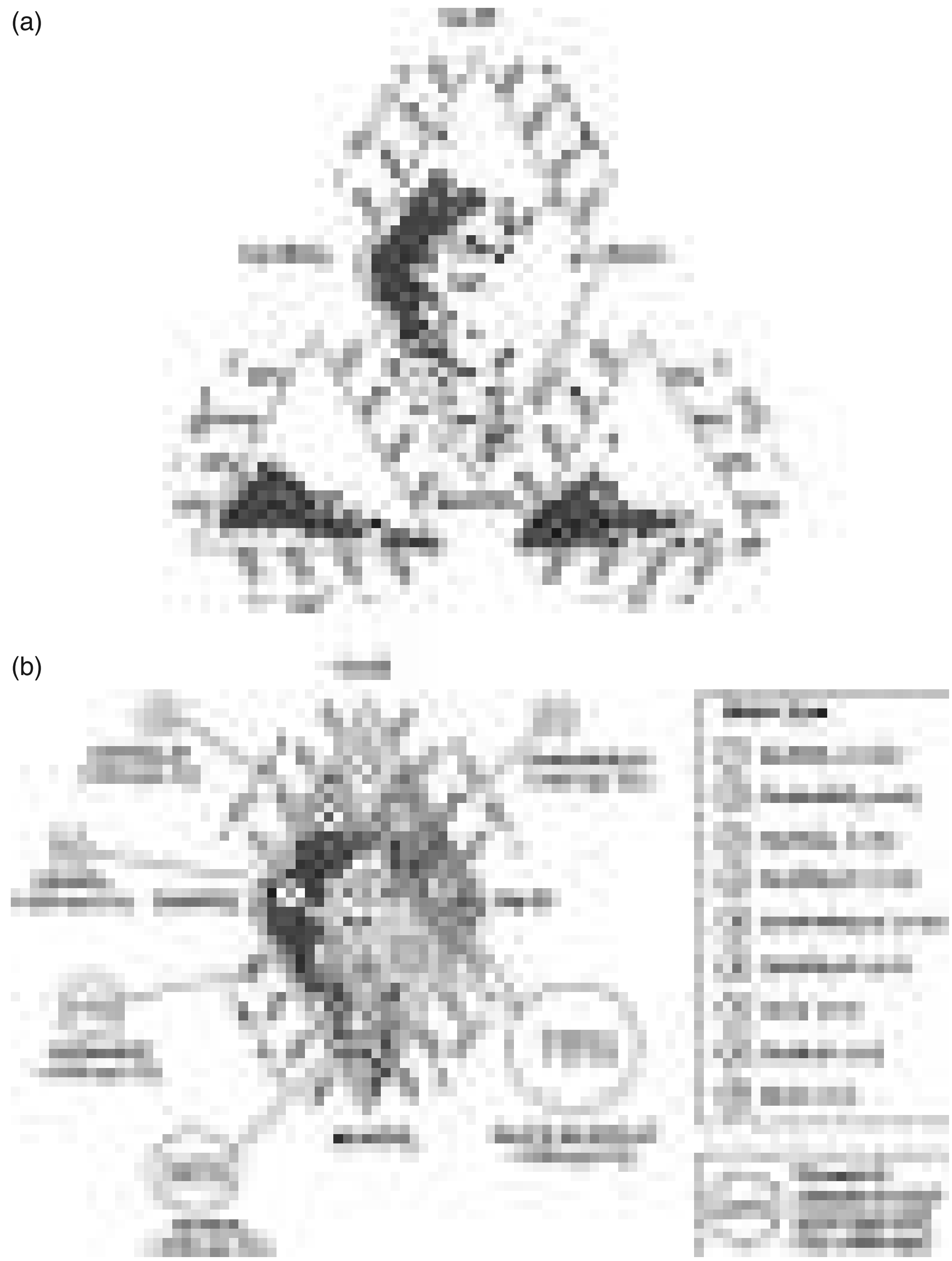


Figure 3. (a) Piper diagrams of major cation and anion composition of 408 predrill water samples. *Note: HCO_3 concentrations estimated from reported alkalinity as $CaCO_3$. (b) Percent of samples in each water type exceeding 1000 μg/L of methane. Water types determined based on those presented in Deutsch (1997).

spaces surrounding casing strings, and the production pipeline and production casing of three nearby shale-gas extraction wells (PA DEP 2010a, 2010b).

Shale-gas wells are constructed of a series of steel casing strings (i.e., long sections of steel pipe) of various diameters that are installed concentrically within the wellbore, in a telescoping fashion. These casing strings serve the dual purpose of providing structural integrity of the wellbore and isolating surrounding formations (e.g., drinking water aquifers) from the underlying production zone. The space between the casing string and the wellbore is referred to as the "annular space," which is commonly cemented in shallower casing strings, but may be left uncemented (i.e., "open") in deeper sections, depending on site-specific conditions. The annular space of a gas well is in contact with surrounding formations, and can therefore contain gases from adjacent strata that may be present naturally above the targeted gas-producing shale formation.

In the three Dimock area gas wells, gas samples were collected from annular spaces surrounding the (1) shallow casing string adjoining potential freshwater aquifers, (2) the intermediate casing string, which penetrates to a depth several thousand feet beneath surface, and (3) the production casing string, which terminates within the Marcellus shale. Within these gas wells, the annular space surrounding the production casing string was cemented from the bottom of the wellbore to a depth above the Marcellus shale. The cementing of this portion of annular space is designed to prevent Marcellus formation gases and fluids from entering the overlying section of wellbore via an open annulus. Consequently, gases sampled from annular spaces surrounding the casing strings in Dimock Township gas wells were considered to represent a mixture of those originating in formations above the Marcellus Shale (i.e., Middle Devonian Mahantango and Upper Devonian Formations), which were penetrated in some sections by open annular spaces. One gas sample was also collected from the well cellar, a sunken pit that provides access to the gas wellhead which serves as a functional interface for the casing strings and the gas production line at the surface. This sample is considered to be a composite of gases found within the annular spaces of the casing strings.

The Pennsylvania DEP also collected Marcellus shale gas (two samples, one gas well) from inside the production casing and production pipeline of a local gas well. These samples are supplemented by Marcellus shale gas (four samples, four gas wells) collected by Cabot (a primary operator in the area) from within the production casing of additional gas wells in the Dimock Township. Collectively, these analyses represent 12 samples of potential source gases from 7 shale-gas wells located within the greater Dimock Township area (see Table S2 for a full data table).

As described previously, ratios of methane to ethane can be used as an initial tool to evaluate possible differences between hydrocarbon gas sources (Bernard et al. 1976; Schoell 1980). Gases originating from the production pipeline/production casing, and annular spaces/well cellar all exhibit methane to ethane ratios less than 60, which is consistent with the anticipated thermogenic origin of these samples (i.e., methane to ethane ratio < 100). However, methane to ethane ratios of the six samples from the production pipeline/production casing (Marcellus shale gas) are slightly lower (43 to 47) than those displayed by gases from annular spaces and well cellar, where all but one of the six samples contain methane to ethane ratios between 49 and 56.

The use of $\delta^{13}C\text{-}CH_4$ values in combination with $\delta^{2}H\text{-}CH_4$ values can provide another method to distinguish different thermogenic (and microbial) gases. Values of $\delta^{13}C\text{-}CH_4$ and $\delta^{2}H\text{-}CH_4$ represent the difference between the $^{13}C/^{12}C$ and $^{2}H/^{1}H$ isotopic ratios of a sample, and that of a recognized standard. More negative values of $\delta^{13}C\text{-}CH_4$ and $\delta^{2}H\text{-}CH_4$ are said to be isotopically "depleted" in the heavier isotopes (i.e., ^{13}C and ^{2}H), whereas more positive values are said to be isotopically "enriched."

As shown in Figure 4a, the $\delta^{13}C\text{-}CH_4$ and $\delta^{2}H\text{-}CH_4$ values of Marcellus shale gas and overlying Middle and Upper Devonian gases sampled from the annular spaces of casing strings in the Dimock Township plot along a continuous trend. Based on the well-documented occurrence of less thermally mature methane in Middle and Upper Devonian Formations throughout the Northern Appalachian basin (Jenden et al. 1993; Osborn and McIntosh 2010), this trend may be interpreted to represent increasing thermal maturity, where Marcellus shale gas (buried at greatest depths) displays the most enriched isotopic signature (i.e., greatest thermal maturity).

It is important to note that mixing lines among gases of differing thermal maturities can, in certain instances, produce a pattern similar to that of a thermal maturity trendline. In this regard, our study faces similar challenges to prior studies on the origin of thermogenic gases in the Appalachian basin (e.g., Jenden et al. 1993). Nevertheless, the differences in $\delta^{13}C\text{-}CH_4$ and $\delta^{2}H\text{-}CH_4$ values for Marcellus production gases in the Dimock Township vs. Middle and Upper Devonian gases from the annular spaces or well cellar of local gas wells indicates that, although plotting along a continuous trend, the two sources may be distinguished.

The $\delta^{13}C$ value of ethane has also been utilized to provide additional information on the source of thermogenic hydrocarbon gases in the subsurface (Coleman 1994). Of the four samples of annular space and well cellar gases for which the $\delta^{13}C$ value of ethane were measured, reported values range between −34 and −35‰. The six samples of Marcellus gas from the production pipeline/production casing display similar, though slightly depleted, $\delta^{13}C\text{-}C_2H_6$ values ranging between −35 and −36‰ (Figure 4b). All reported $\delta^{13}C\text{-}C_2H_6$ values of gases from annular spaces/well cellar, and production pipeline/production casing are more depleted than the corresponding $\delta^{13}C\text{-}CH_4$ value, and therefore are said to exhibit an "isotope reversal" (i.e., a reversal of the trend traditionally observed in conventional hydrocarbon gases in which ethane is isotopically heavier, or more "enriched" in heavier isotopes, than methane). Such isotope reversals have commonly been observed in Marcellus and other Middle Devonian gases below the Tully limestone (Baldassare 2011). Comprehensive ethane isotope data are not currently available for gases present throughout the stratigraphic section in the Dimock Township. However, the magnitude of the methane-ethane $\delta^{13}C$ isotope reversal in samples from the annular spaces and well cellar (3 to 4‰ difference between methane and ethane $\delta^{13}C$ values) and those from the production pipeline/production casing (5 to 7‰ difference) suggests that Marcellus production gases may be distinguished from Middle and Upper Devonian gases in the annular spaces/well cellar of local gas wells.

It is important to note that, because of local variations in thermal maturity and possible source material, the isotopic and molecular signatures of formation gases in the Dimock Township cannot be extrapolated to represent the signatures of gases from the similar stratigraphic units in different parts of Susquehanna County or Pennsylvania. Consequently, the signatures of gases from the Marcellus and overlying Middle and Upper Devonian Units, as presented in Figure 4a and 4b, should only be considered representative of those found in the local Dimock area.

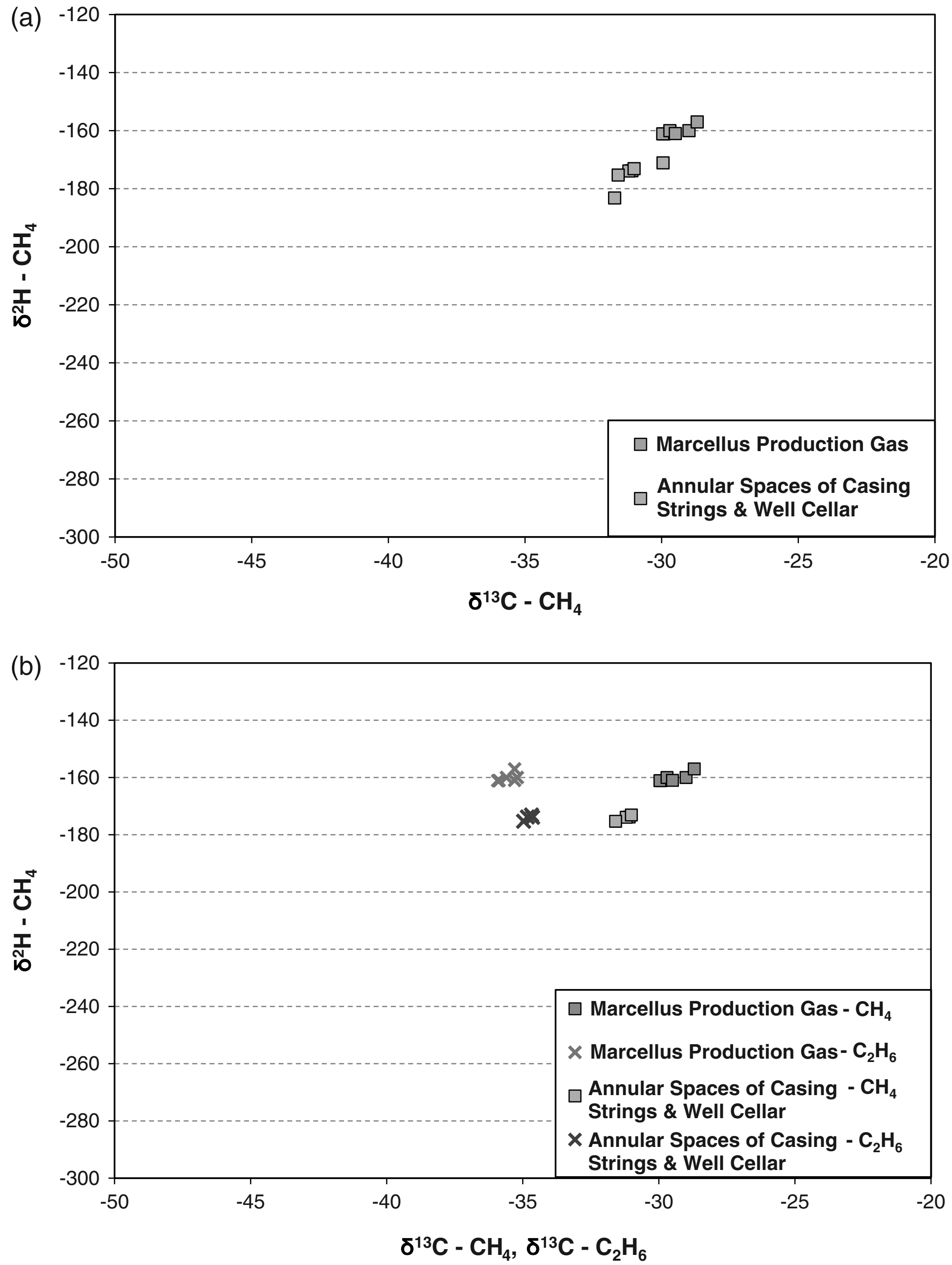


Figure 4. (a) Schoell plot of gases sampled by the Pennsylvania DEP and Cabot from production casing/production pipelines, the annular spaces of casing strings, and the well cellar of gas wells in the Dimock Township. (b) Plot of $\delta^{13}C$ values of methane and ethane (x-axis) vs. δ^2H value of methane (y-axis) in gases sampled by the Pennsylvania DEP and Cabot from production casing/production pipelines, the annular spaces of casing strings, and the well cellar of gas wells in the Dimock Township. *Only samples with reported values of both $\delta^{13}C$-CH_4 and $\delta^{13}C$-C_2H_6 are shown.

Water Well and Salt Spring Gas Samples: Isotopic and Molecular Analyses, 2009 to 2010

In addition to characterizing gases originating from the Marcellus shale and overlying deposits in the Dimock Township, the Pennsylvania DEP and Cabot analyzed groundwater samples from 15 local Dimock area water wells for this same suite of isotopic analyses (details of well depth and construction provided in Table S1). Cabot also sampled one salt spring in Salt Springs State Park, a known historical source of methane gas located approximately 13 miles northeast of the Dimock Township. Although this sample is not representative of the immediate Dimock area, it does serve as an illustrative example of the composition of gases that have been migrating naturally in the shallow subsurface in south-central Susquehanna County prior to commencement of shale-gas extraction activities. In several instances, multiple samples were obtained over the 2-year period, resulting in 23 total water well samples (8 samples of dissolved gas, 15 samples of free gas) and 1 salt spring

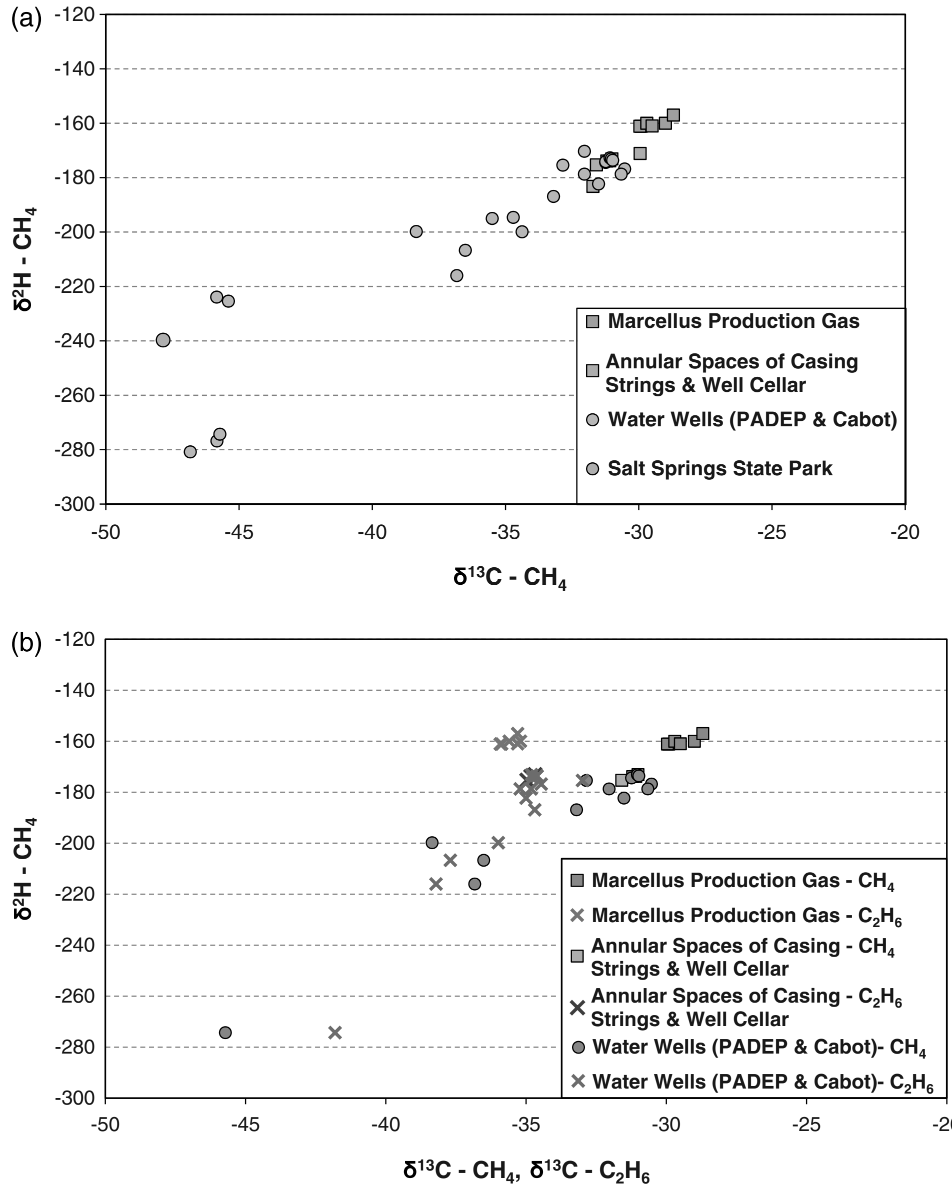


Figure 5. (a) Schoell plot showing Pennsylvania DEP and Cabot isotope data on methane samples from Dimock area water wells and Salt Springs State Park (Franklin Township). (b) Plot of δ^{13}C values of methane and ethane (x-axis) vs. δ^2H value of methane (y-axis) in gases sampled by the Pennsylvania DEP and Cabot from water wells in the Dimock Township. *Only samples with reported values of both δ^{13}C-CH_4 and δ^{13}C-C_2H_6 are shown.

sample (free gas) analyzed at Isotech Laboratories in 2009 and 2010.

For 15 of the 23 water well samples, the ratio of methane to ethane is less than 100, suggesting thermogenic origin. The remaining eight water wells and one salt spring contain methane to ethane ratios ranging from 100 to over 6000. This wide spread of values could represent several distinct thermogenic and microbial gas sources, a mixture of different sources, and/or gases that have undergone alteration during migration. The δ^{13}C and δ^2H values of methane for these same samples are presented in Figure 5a. Each of the water well samples and the salt spring sample exhibit δ^{13}C values (less than or equal to −30‰) and δ^2H values (less than or equal to 170‰) that appear consistent with those of thermogenic Middle and Upper Devonian gases sampled from the annular spaces/well cellar, microbial gases, or a mixture of the two. Based on the δ^{13}C-CH_4 and δ^2H-CH_4 values alone, samples from at least 2 of the 15 water wells contain a clear microbial component (i.e., samples from these two wells plot close to or within the traditional Schoell "microbial-fermentation" gas zone: (i.e., δ^{13}C-CH_4 between −45 and −65‰, δ^2H-$CH_4 \leq -275$‰).

The δ^{13}C value of ethane was measured in 13 of the water well samples (originating from 10 different water wells). Of these 13 samples, 11 exhibited a methane-ethane δ^{13}C isotope reversal (Figure 5b). Nine of these samples display combined methane and ethane δ^{13}C values that appear most consistent with Middle and Upper

Devonian gases sampled in the annular spaces of casing strings and the well cellar. An additional two samples still show a methane-ethane $\delta^{13}C$ isotope reversal, but exhibit more depleted methane and/or ethane $\delta^{13}C$ values than annular space/ well cellar gases. Lastly, two water samples did not exhibit an isotope reversal, and displayed notably more depleted methane and ethane $\delta^{13}C$ values than gases sampled in the annular spaces and well cellar.

Isotopic and Molecular Analyses by Osborn et al. (2011)

In 2011, Osborn et al. (2011) presented δ^2H and $\delta^{13}C$ values for dissolved gases in nine water wells sampled the previous year in "active-gas extraction areas" (i.e., within 1 km of an active gas well) in Susquehanna County. A map of sampling locations in Susquehanna County (provided in Osborn et al. 2011) indicates that a majority of the water wells sampled in active gas extraction areas were located in close proximity to the Dimock Township. These dissolved gases exhibited δ^2H and $\delta^{13}C$ values of methane that are generally more depleted than, or on the border of, the range characterized by local Marcellus production gases sampled by the Pennsylvania DEP and Cabot in the Dimock area, consistent with the signatures of Middle and Upper Devonian gases in the annular spaces/well cellar (Figure 6).

Isotopic and Molecular Analyses by USEPA (2012)

The USEPA also analyzed dissolved gases in 11 water wells in the Dimock Township in 2012 as part of follow-up sampling to the original Pennsylvania DEP stray-gas investigation (Table S6). Eight of the 11 water wells display combined δ^2H and $\delta^{13}C$ values of methane that are distinct (i.e., more depleted) from local Marcellus production gases. However, 3 of the 11 water wells contain methane with δ^2H and $\delta^{13}C$ values that plot within the elevated range of values that characterizes Marcellus production gas in the Dimock Township ($\delta^{13}C$-$CH_4 > -30‰$, δ^2H-$CH_4 > -162‰$). On this basis alone, these dissolved gases appear to match production gas samples from the Marcellus shale. However, closer examination of the data from these three water wells suggests that microbial oxidation in the subsurface is likely responsible for the elevated δ^2H and $\delta^{13}C$ values.

Specifically, samples from two of these three water wells (House 4 and 14) exhibit $\delta^{13}C$-CH_4 values that are slightly more enriched ($\delta^{13}C$-$CH_4 = -25$ to $-27‰$) than those exhibited by Marcellus production gas samples ($\delta^{13}C$-$CH_4 = -28$ to $-30‰$). However, these same samples display δ^2H-CH_4 values that are significantly more enriched (-122 and $-140‰$) than local Marcellus production gas (δ^2H-$CH_4 = -161$ to $-157‰$). The particularly elevated δ^2H signatures of the methane in these two samples is suggestive of microbial oxidation of methane, whereby both the δ^2H and $\delta^{13}C$ values of methane are altered to be enriched, but the change in the δ^2H value of methane is 8 to 14 times greater than the change in the $\delta^{13}C$ value of methane (Coleman et al. 1981).

The sample from the third water well (House 2) exhibits δ^2H and $\delta^{13}C$ values ($-160.5‰$ and $-29.4‰$, and respectively) that plot in the depleted end of the isotopic range characterized by Marcellus shale gas. However, the sample from House 2 does not exhibit an isotope reversal between the $\delta^{13}C$ values of methane and ethane that characterizes Marcellus shale gas samples.

Evaluation of a more complete suite of geochemical analyses over time could help discern the degree to which mixing of different gas sources and alteration processes in the subsurface (e.g., oxidation) have affected the signatures of water well gas samples collected by various parties in Susquehanna County.

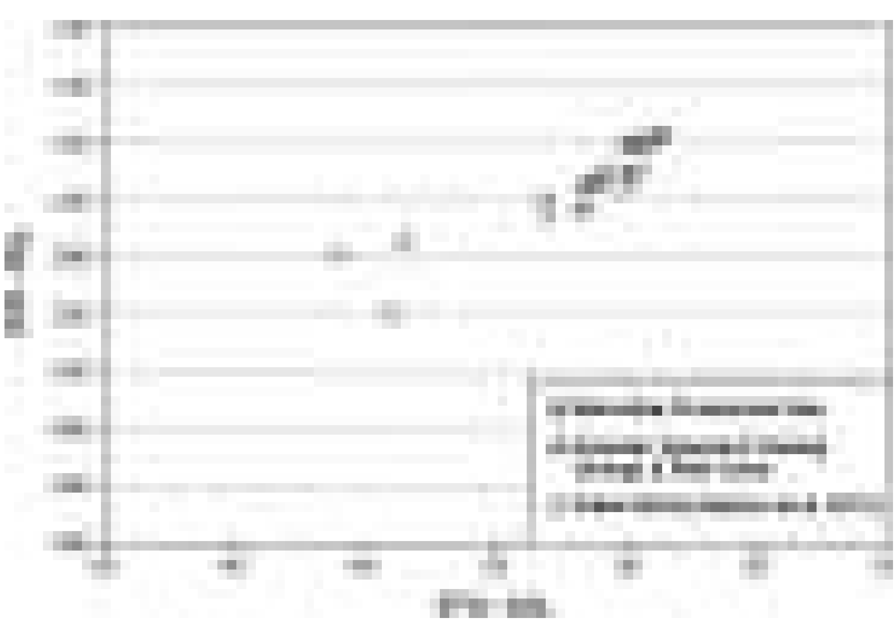


Figure 6. Schoell plot showing Osborn et al. (2011) isotope data on dissolved methane samples from Dimock area water wells.

Discussion

Isotopic and Molecular Analyses in Dimock Township

The δ^2H and $\delta^{13}C$ values of gases in Dimock area water wells sampled by the Pennsylvania DEP, Cabot, Osborn et al., and the USEPA indicate that hydrocarbon gases present in the majority of the water wells are consistent with gases in the well cellar and annular spaces of casing strings that intersect Middle and Upper Devonian formations above the Marcellus. The presence of a $\delta^{13}C$ methane-ethane isotope reversal in numerous water wells would suggest a source below the Tully Limestone in the Middle Devonian Mahantango Formation penetrated by several of the annular spaces sampled. These findings support the conclusion that the methane concentrations in these water wells can be explained with no necessary contribution from deeper Marcellus shale gas.

Isotopic and molecular analyses of hydrocarbon gases can provide valuable information on the source of natural gases, but not necessarily the mechanism of migration. In this article, for the Dimock Township, we have evaluated available isotopic and molecular analyses of local water well samples with the primary goal of assessing gas origin. To identify gas migration pathways, geochemical data should be assessed in conjunction with other lines of evidence which are not within the scope of this article. These lines of evidence include details on the construction, completion, and integrity of local gas wells and water wells. In addition, the variation of methane concentrations and groundwater quality in potentially impacted water wells over time, as well as the spatial distribution of methane concentrations in groundwater relative to potential locations of gas sources, can provide crucial information on potential pathways of migration.

The isotopic and molecular analyses presented in this paper primarily represent measurements of hydrocarbon gases in local strata and water wells in the Dimock Township. Without additional analyses of samples throughout the greater Susquehanna County area, the signatures of hydrocarbon gases in local area gas and water wells cannot be extrapolated to represent those of methane or ethane that may be found in strata or water wells throughout the region. However, the evaluation of Dimock area isotope and molecular analyses of methane and ethane provide valuable insight into the complexity associated with using geochemical fingerprinting to identify the origin of hydrocarbon gases in regions where multiple thermogenic gas sources are present naturally.

Consideration of Regional Lines of Evidence Regarding Sources of Methane in Groundwater

Methane has been present naturally throughout Susquehanna County in the shallow subsurface for at least 200 years, as indicated by the well-documented history of hydrocarbon gas shows during water well and gas well drilling, as well as natural seeps of hydrocarbon gases observed in the form of bubbling springs, ponds, and water wells. Our evaluation of 1701 recent predrill water well samples from Susquehanna County confirms that methane is common in drinking water aquifers today, with approximately 78% of sampled water wells exhibiting detected methane concentrations, and 3.4% exceeding the 7000 µg/L Pennsylvania DEP action level. Elevated methane concentrations show a clear relation to topography, with levels above the Pennsylvania DEP action level disproportionately found in valleys.

Furthermore, methane concentrations in valley water wells in gas production areas (i.e., located within 1 km of an active gas well) versus valley water wells in nonproduction areas (i.e., greater than 1 km from a gas well) show no statistical difference, indicating that the regional presence of elevated methane concentrations in valleys is a natural phenomenon. This is not an original finding, as the presence of elevated methane levels in valley water wells in the Appalachian basin has also been observed in a study in West Virginia by Mathis and White (2006), which found that methane concentrations from 170 water wells that exceeded 10,000 µg/L occurred in wells located in valleys and hillsides, as opposed to hill tops (Mathis and White 2006).

The correlation of methane concentrations with topography, rather than areas of active shale gas extraction, indicates that the use of hydraulic fracturing for shale gas in northeastern Pennsylvania has not resulted in widespread gas migration into the shallow subsurface. Certainly, as described in the 1980s by Harrison (1983, 1985), and by Gorody (2012) and King (2012), there have been instances of stray gas migration associated with the accumulation of gas pressure within and around the sides of the annular spaces of gas well casing strings in Pennsylvania, Ohio, and New York. However, the absence of a regional-scale relationship of methane concentrations to shale-gas well locations is consistent with the experience that gas well integrity and over-pressurization problems commonly result in localized, rather than regional, effects on water quality (van Stempvoort et al. 2005).

Valley water wells in Susquehanna County exhibit median methane concentrations similar to that of upland water wells; however, the 90th percentile concentrations of methane in valley wells is significantly elevated relative to upland wells. This observation suggests that some valley water wells access natural sources of elevated methane via interconnection with specific groundwater units and/or enhanced pathways of methane migration. As shown in Figures 3a, 3b, and S1, the elevated methane concentrations observed in this data set are predominantly associated with Ca-Na-HCO_3 or Na-HCO_3 water types, which in combination comprise only 16% of the sample population, yet represent 69% of the methane concentrations greater than 1000 µg/L. In addition, although only four water samples match a Na-HCO_3-Cl or Na-Cl water type, three of these samples exhibit methane concentrations exceeding 10,000 µg/L. Samples with methane concentrations greater than 1000 µg/L also exhibit relatively elevated levels of barium, chloride, manganese, pH, sodium, strontium, and TDS, and relatively lower levels of sulfate and calcium. Similar geochemical relationships with elevated methane concentrations were previously reported in a study by Perry et al. (2012) for over 14,000 predrill water well samples in Pennsylvania, Ohio, and West Virginia, as well as a study by Warner et al. (2012) for 426 shallow groundwater samples from six counties in northeastern Pennsylvania.

The association of sodium and sodium-chloride rich water types with elevated methane concentrations suggest that wells exhibiting higher methane levels are connected to deeper groundwater aquifers, which have experienced longer groundwater residence times (and therefore, longer rock-water interaction times), and/or are in contact with sodium-chloride rich waters that occur in deeper bedrock and aquifers. Specifically, water in the deeper parts of sandstone aquifers that contain carbonate components (e.g., shells or carbonate rich shale lenses) commonly transitions from Ca-HCO_3 type water to Na-HCO_3 or Na-Cl-HCO_3 type water as a result of (1) the mixing of circulating freshwater with sodium-chloride water and/or brine that occurs in deeper bedrock and aquifers, and/or (2) progressive cation exchange, whereby calcium, magnesium (and to a lesser degree, potassium) ions replace sodium on mineral exchange sites, thereby liberating sodium in groundwater (Cushing et al. 1973, Kresse et al. 2012). It follows that longer rock-water interaction results in increased cation exchange, and a transition towards sodium-bicarbonate water, containing greater concentrations of barium, boron, chloride, lithium, strontium, and TDS associated with the dissolution of carbonate and other minerals from the rock matrix. At the same time, concentrations of calcium and magnesium will decrease in groundwater with increased residence time in the subsurface, as these ions replace sodium in mineral exchange sites (Kresse et al. 2012).

Longer groundwater residence times are also associated with anaerobic groundwater conditions related to microbial consumption of oxygen and other electron acceptors (e.g., sulfate, manganese, and iron oxides) in the presence of an organic substrate. Evidence for the occurrence of these redox processes can include increased concentrations of manganese and iron from the reduction of iron and manganese oxides, as well as decreased sulfate concentrations (and associated sulfide odors) related to sulfate reduction (Kresse et al. 2012).

In Susquehanna County, deeper groundwater with longer residence times may be preferentially accessed by water wells that penetrate to the depth of deeper groundwater units or are in communication with fault and fracture networks. Comprehensive information on the depth of water wells sampled in our predrill dataset is not available. However, available information on aquifer characteristics in northeastern Pennsylvania indicates that the Lock Haven Formation contains discrete saline groundwater zones, which have been noted to produce methane gas and/or hydrogen-sulfide (Williams et al. 1998). Analysis of water quality by Warner et al. (2012) in northeastern Pennsylvania showed that water wells drawing from the Lock Haven Formation (45 wells) exhibit significantly elevated chloride and sodium concentrations (75th percentile values of 73 and 103 mg/L, respectively) compared to water wells drawing from the Catskill Formation (102 wells, with 75th percentile values of 15 and 16 mg/L, respectively). Consequently, water wells that are in contact with Lock Haven groundwater, via either their greater depth within the overlying Catskill formation and/or fracture flow networks are more likely to exhibit decreased water quality and higher methane concentrations.

Valley formation in Susquehanna County has been associated with a greater density of fractures and lineaments, which likely serve as enhanced pathways for both water and gas migration. In West Virginia, Rauch (1984) found that water wells located within 0.1 km of photolineaments (short straight lineaments representing linear stream channel or valley segments related to the surface presentation of fault and fracture networks) had significantly higher water yields than other water wells. This increased water yield was not simply related to the influence of topography on groundwater flow, as valley water wells located near photolineaments exhibited greater water flow than all other valley water wells (Rauch 1984). It follows that water wells penetrating fault and fracture networks are not only likely to produce greater quantities of water, but higher methane levels as well.

It is important to note that the presence of methane itself can alter the concentrations of certain groundwater parameters. Specifically, the oxidation of methane is associated with the reduction of oxidizing reactants (i.e., sulfate, manganese, and iron oxide), an increase in pH and bicarbonate (product of CH_4 oxidation) concentrations, and a decrease in oxidation-reduction potential (Kelly and Matisoff 1985). However, microbial oxidation should not affect the prevalence of chloride, which is a conservative ion (i.e., an ion that is not utilized in redox reactions, does not sorb readily to mineral surfaces or complex with other ions, and is not easily removed from solution) (Kresse et al. 2012). Consequently, the association of increasing chloride concentrations with methane concentrations strongly suggests the contribution of a deeper groundwater source with elevated chloride concentrations.

The significant history of gas production from the Lock Haven (and, to a lesser extent, the Catskill Formation) in northeastern Pennsylvania suggests that methane naturally present in many Susquehanna County water wells contains thermogenic components from the Catskill and Lock Haven formations. In addition, microbial methane, produced by methanogenesis in anaerobic groundwater units, may also be a strong component of the methane migrating into local drinking water wells. As discussed earlier, the present-day migration of deep thermogenic gases from the Mahantango and Marcellus Formations into the shallow subsurface in Susquehanna County appears to be limited by the presence of a regional structural detachment above the Tully stratigraphic level, which has been noted in local seismic interpretations. The data presented in this paper show that elevated methane concentrations exhibit a common association with (1) topographic lows and (2) groundwater geochemistry that reflects longer residence times and/or mixing with deeper NaCl waters. This observation suggests that elevated sources of either thermogenic or microbial methane from the Catskill and Lock Haven Formations are accruing in deep groundwater units, and/or within long discrete groundwater flow paths to lowland discharge points, that are preferentially accessed by certain valley water wells.

This evaluation represents a regional-scale assessment of trends observed in groundwater geochemistry in a large dataset during a 3-year period. However, for an individual water well, it is important to remember that factors such as proximity to roads (and therefore, the application of sodium and calcium chloride during the winter season) can result in localized variations in chloride, total dissolved solids, and other water quality parameters. In addition, seasonal fluctuations in aquifer recharge (i.e., increased influx of meteoric water between November and May), residential water use, changing weather patterns, and local well construction can also affect site-specific water quality.

Conclusion

Our evaluation of 1701 groundwater quality analyses shows that methane is common in Susquehanna county water wells and is best correlated with topography and groundwater geochemistry, rather than shale-gas extraction activities. This finding suggests that shale-gas extraction in northeastern Pennsylvania has not resulted in regional gas impacts on drinking water resources and that, in turn, the hydraulic fracturing process has not created extensive pathways by which gas from the Marcellus shale could rapidly migrate into the shallow subsurface.

Our evaluation of site-specific isotopic and molecular data from water wells in the Dimock Township suggests that hydrocarbon gases present in these water wells are consistent with Middle and Upper Devonian gases above the Marcellus sampled in the annular spaces of local gas wells. This evaluation also emphasizes the complexity associated with differentiating between multiple thermogenic gas sources that may exhibit subtle variations. Isotopic analyses were not performed on the 1701 predrill water well samples to determine the origin of elevated methane concentrations observed throughout Susquehanna County. However, consideration of regional geology, historical publications, structural data, water well completion records, and water quality data suggest that methane naturally migrating into Susquehanna County water wells is either thermogenic, likely originating from Upper Devonian deposits overlying the Marcellus shale, or microbial, originating from anaerobic groundwater units with long residence times.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Supporting Information for L.J. Molofsky et al. (2013).
Figure S1. Concentration distribution of groundwater analytes. Plots showing the concentrations of alkalinity, barium, calcium, chloride, magnesium, manganese, pH, potassium, sodium, strontium, sulfate, and TDS by methane concentration range (i.e., nondetect (ND), greater than ND to 1000 μg/L, greater than 1000 to 7000 μg/L, greater than 7000 μg/L).
Figure S2. Methane concentrations vs. methane to ethane ratios of water well samples for which both methane and ethane were detected.
Figure S3. Map of water and gas wells sampled for isotopic and molecular analyses in the Dimock township.
Figure S4. Locations of Historical Gas Wells, Gas Fields, Salt/Mineral Springs, and Shows of Saline Water and Combustible Gases in Water Wells.
Table S1. Construction Details of Water Wells Sampled for Isotopic and Molecular Analyses of Gases in the Dimock Township
Table S2. Isotopic and Molecular Analyses of Gas Samples from Water and Gas Wells in the Dimock Township
Table S3. Isotopic and Molecular Analyses of Split Gas Samples from Water Wells and Salt Springs State Park Analyzed at GeoMark Research, Ltd.
Table S4. Isotopic and Molecular Analyses of Desorption Canister Gas Samples from Gas Wells in the Dimock Township
Table S5. Osborn et al. (2011) Plotted Isotopic Data of Dissolved Gases in Water Wells Penetrating the Catskill Aquifer in an "Active Gas Extraction Area"
Table S6. U.S. EPA (2012) Isotopic and Molecular Analyses of Dissolved Gases in Water Wells Located in the Dimock Residential Groundwater Site
Table S7. Details of Historical References to Locations of Gas Wells/Fields (yellow), Water Wells Containing Combustible Gases (red), Water Wells Containing Saline Water (green), and Salt/Mineral Springs (blue)
Table S8. Groundwater Quality Data for 1701 Water Well Samples from Susquehanna County, Pennsylvania

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