5. Global Warming Effects of Unconventional Shale Gas Development

Presenter – Mav Moorhead

Support for natural gas development appears to be based on the mistaken premise that natural gas is a “clean” fossil fuel, that it is “good” in our efforts to combat climate change. These are characterizations that shale gas cannot claim when fugitive methane emissions from development, transportation and use are taken into account.

Methane is a far more powerful greenhouse gas than carbon dioxide. For the first 20 years of its lifetime in the atmosphere, one pound of methane traps as much heat as at least 80 pounds of CO_2. Its potency declines until it is about 25 to 30 times more powerful than CO_2 over a hundred years. Although when burned gas emits half the CO_2 of coal, methane leakage eviscerates this advantage because of its greenhouse power. (Shindell et al., 2009)

And methane is leaking. At the downstream end of the methane life-cycle, recent measurements in Boston, Washington, DC, and New York City have revealed a shocking number of leaks in aging distribution pipelines and methane concentrations in the air in these major cities up to 5 times the natural background level (Phillips et al. 2013; Ackley and Payne, 2013). Recent field measurements led by scientists at the National Oceanic and Atmospheric Administration (NOAA) have found upstream/midstream only (not including transmission and distribution losses) emissions in a region of Colorado between 2.3 and 7 percent of production; upstream/midstream emissions only up to 9 percent in Utah; and upstream/midstream/downstream emissions up to 17 percent in the Los Angeles CA basin (Petron et al., 2012; Nature, 2013; Peischl et al. 2013).

These measurements validate the range predicted in the seminal paper on this topic published by scientists and engineers at Cornell University in 2011 (Howarth et al. 2011; Howarth and Ingraffea, 2011; Howarth et al. 2012; Howarth et al., 2012). A subsequent 2011 study from the National Center for Atmospheric Research (NCAR) concluded that unless leaks can be kept below about 2%, gas lacks any climate advantage over coal (Wigley, 2011). A 2012 paper from the Environmental Defense Fund pegs this crossover rate at about only 3% (Alvarez et al., 2013). A recent study by the science group Climate Central shows that the alleged 50%

climate advantage of natural gas is unlikely to be achieved for many decades, if at all (Larson, 2013).

Unfortunately, we don't have that long to address climate change—the next two decades are crucial. Shindell et al. (2012) note that the climate system is more immediately responsive to changes in methane (and black carbon) emissions than carbon dioxide emissions. They predict that unless emissions of methane and black carbon are reduced immediately, the Earth will warm to 1.5° C by 2030 and to 2.0° C by 2045 to 2050 whether or not carbon dioxide emissions are reduced. Reducing methane and black carbon emissions, even if carbon dioxide is not controlled, would significantly slow the rate of global warming and postpone reaching the 1.5° C and 2.0° C marks by 12 to 15 years. Controlling carbon dioxide as well as methane and black carbon emissions further slows the rate of global warming after 2045, through at least 2070. The life-cycle of shale gas produces all three of these climate change culprits: carbon dioxide, methane, and black carbon.

While it is possible to reduce fugitive emissions from shale gas development, the technologies to do so have not been embraced by operators because the costs are prohibitive from their view. For example, in 2012 the industry demanded a delay from the EPA until January 1, 2015 of the mandatory implementation of the simplest of these technologies: green completions. It is also certain that any efforts to adequately regulate the industry will be vigorously opposed by this well-resourced industry and its lobbyists.

The other unfounded assumption of some shale gas promoters is that natural gas is a bridge fuel to a cleaner low carbon economy. Not only does the evidence show that shale gas development is more problematic than continued use of oil and even coal, certainly over the short term, the supposed bridge period, there is no scientific basis for assuming that curbing methane emissions will be easier than implementing the conservation, efficiency and renewable energy strategies that will reduce our reliance upon fossil fuels including natural gas.

We have renewable wind, water, solar and energy-efficiency technology options now to avoid the enormous risks of fracking for shale gas (Jacobson et al., 2013). We can scale these quickly and affordably, creating economic growth, jobs, and a truly clean energy future to address climate change. Political will is the missing ingredient. Meaningful carbon reduction is impossible while the fossil fuel industry has captured too much of our energy policies and regulatory agencies, plus intentionally distorted public debate. Policy-makers, including the President, need to listen more closely to the voices of independent scientists over the din of industry lobbyists.

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Please see***The whole story-including the Extended Report, Press Release and media coverage:*** *http://www.damascuscitizensforsustainability.org/2013/03/manhattan-natural-gas-pipeline-emissions-2/*

PRESS RELEASE: **for immediate release** 3/25/13

Media: New Study Exposes How Natural Gas Isn't the Clean Fossil Fuel It's Hyped up to Be

Actual Methane Emissions Measured in Manhattan Showing No Advantage to Natural Gas: Two Reports

Report on a Preliminary Investigation of Ground-Level ambient Methane Levels in Manhattan, New York City, New York

Extended Report on Preliminary Investigation of Ground-Level Ambient Methane Levels in Manhattan, New York City, New York

Report on a Preliminary Investigation of Ground-Level Ambient Methane Levels in Manhattan, New York City, New York

[This is an initial report subject to revision. First revision 29 March 2013]

16 December 2012

Robert Ackley and Bryce F. Payne Jr. , PhD
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BACKGROUND

There are serious environmental concerns with the development of shale gas and the related new gas industry infrastructure, and recent investigations have raised concerns about the role of cities in assuring the public and environmental safety of natural gas use. In cities gas will be distributed and delivered through existing and new gas lines, almost all buried under city streets and sidewalks. In most U.S. cities the gas lines have been in place for decades. Consolidated Edison, Inc. (ConEd) in New York City, for example, has been installing gas lines underground since the early 1800s and now has a system of 4320 miles of gas pipe.[1] ConEd has installed pipes under almost every street or sidewalk in their service territory (except northern Westchester). The ConEd gas system in the 23-square mile service area in Manhattan delivers gas through 336,000 customer gas meters. All underground pipes, as in the ConEd gas system, are subject to stresses and strains of corrosion, and physical damage during excavation or due to natural forces. It follows that such extensive, complex and largely aged pipe systems will have maintenance requirements and will develop leaks and other problems that have to be

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[1] http://www.coned.com/PublicIssues/PDF/GLRP1210c.pdf

managed to prevent explosion hazards and property damage, e.g. to urban trees, and to assure public and worker safety.

In addition to the more obvious concerns about safety, (such as explosions and wasted gas) there is an additional concern that arises from the fact that commercial natural gas is almost entirely comprised of methane. This naturally occurring gas is formed deep in the earth during the geological processes that form oil and coal, and near or at the earth's surface by biological processes, like decay of sewage, or in the gut of mammals. Until recently, CO2 has received most of the attention as a problematic greenhouse gas; yet now there is an increasing awareness of the role of methane, which has an unusual potency as a greenhouse gas. Depending on how it is calculated, methane is 20 to 100 times more potent as a greenhouse gas than carbon dioxide.[2] However, because burning natural gas generates less carbon dioxide than burning coal or oil, natural gas has been considered a cleaner energy source. **However, because methane is such a potent greenhouse gas, if only a small amount leaks into the atmosphere during extraction, transport and delivery of natural gas to the consumer, the smaller carbon footprint of natural gas burned as fuel grows quickly.** Recent estimates are that if more than 2% of natural gas produced at a well is lost to the atmosphere before it is burned by the consumer, then natural gas will no longer be a cleaner fuel than coal with respect to global warming.[3] How much urban gas distribution and delivery systems may be contributing to exceeding that 2% loss rate is only beginning to be understood.

To begin to better understand the role of NYC with regard to these and other concerns about natural gas safety and global climate concerns a group of private donors in NYC funded Damascus Citizens for Sustainability (DCS) to commission a preliminary investigation of natural gas leaks in parts of the Manhattan Borough. DCS engaged Gas Safety, Inc. (GSI) of Southboro, Massachusetts to perform the preliminary investigation.

METHOD

The investigation involved a road survey of ground level ambient air methane levels using a methane (natural gas) leak surveyor system comprised of a cavity ring-down spectrometer combined with a GPS system and computer control system. The leak surveyor was installed in an automobile with an air sampling line mounted over the rear

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[2] Differences in the greenhouse potency of methane compared to carbon dioxide arise from differences in how long these two gases typically remain in the atmosphere. Once released into the air both methane and carbon dioxide are removed relatively slowly, but carbon dioxide disappears about ten times more slowly than methane. Consequently, if compared on a ten-year time frame the faster removed methane has a relatively higher effect (methane 100 times CO_2) than when compared over a one-hundred-year time frame during which the longer-lived carbon dioxide will have a stronger overall effect (methane 20 times CO_2). See Intergovernmental Panel on Climate Change (2007) IPCC fourth assessment report (AR4). Working Group 1, The Physical Science Basis. http://www.ipcc.ch/publications_and_data/ar4/wg1/en/contents.html, and Shindell DT, Faluvegi G, Koch DM, Schmidt GA, Unger N, Bauer SE (2009) Improved attribution of climate forcing to emissions. Science 326:716–718

[3] Robert W. Howarth , Renee Santoro and Anthony Ingraffea, 2011. Methane and the greenhouse-gas footprint of natural gas from shale formations -- A letter. Climatic Change. DOI 10.1007/s10584-011-0061-5

bumper to ride with the inlet facing down approximately 1 foot above the pavement surface, and the GPS antenna on the roof. The instrument measures and records methane levels in the air above the pavement with an accuracy of a few parts per billion (ppb) about 4 times per second. The onboard GPS system simultaneously records the location of the instrument as sampling occurs.

To confirm the reliability of the methane surveyor several leaks were confirmed by locating the actual points in the road surface from which methane was actually being released into the air. Methane levels just below the surface at the actual methane release points were too high to be measured using the spectrometer and were instead measured using a conventional combustible gas indicator.

RESULTS

The surveyor was driven over 160 miles of selected roads in Manhattan from 27-30 November and 9 December 2012 (see Images 1-5). Methane measurement functions were normal during the survey. However, in some areas in Manhattan tall buildings block GPS satellite signals. Consequently GPS data was intermittent, with deviations from actual driven paths apparent in the visualization of the data in the Google Earth images in this report. Loss of GPS signal caused the plotted survey course in the images to appear to occasionally randomly curve off roadways (see Images 1-5). Those random deviations are minor location errors in the plotted survey course, had no functional connection or impact on the methane data, and did not impact the reliability of the methane leak survey. The survey generated over 700,000 methane measurements, and associated numbers of time and location data points. Those data are presented visually in Images 1 through 6 in this report.

During the survey the periphery of the island was driven at different times. Also, the surveyor was intentionally left on during GSI travel from and to Southboro, MA. The data collected on the cross-country drives from and to Massachusetts provided reference methane levels for comparison to those measured in Manhattan (see Image 6 and DISCUSSION below). Methane levels measured along the upwind periphery of Manhattan were similar to those measured on the cross-country drives.

Images 1-5. Results for each day of the methane survey of ground level ambient air in Manhattan on 27-30 November and 9 December 2012. The height of the red line (curtain) indicates ambient air methane levels (in ppm) 1 foot above the road surface along the survey course. One or more peaks are labeled with the associated methane level (in ppm) to provide scale. The viewer should be aware of the perspective in the images, i.e., similarly sized peaks will appear smaller at visually more distant areas of Manhattan in the images.

Image 6. Preliminary gas leak survey of Manhattan 27-30 November 2012 and 9 December. This image provides a visual impression of the relative levels of methane in ambient air in Manhattan compared to levels on open country highways travelled to and from Manhattan. The height of the red line (curtain) indicates ambient air methane levels 1 foot above the road surface along the survey course. One or more peaks are labeled with the associated methane level (in ppm) to provide scale.

DISCUSSION

The survey indicated that natural gas leaks are occurring generally throughout the Manhattan Borough (see Images 1-5). This preliminary study was more intense in some southeastern and southern parts of Manhattan. Leaks appeared more common in those areas. A more thorough study would be necessary to definitively discriminate areas that may have more or larger leaks than other areas. The preliminary investigation results indicated hundreds to thousands of likely leaks in the surveyed parts of Manhattan.

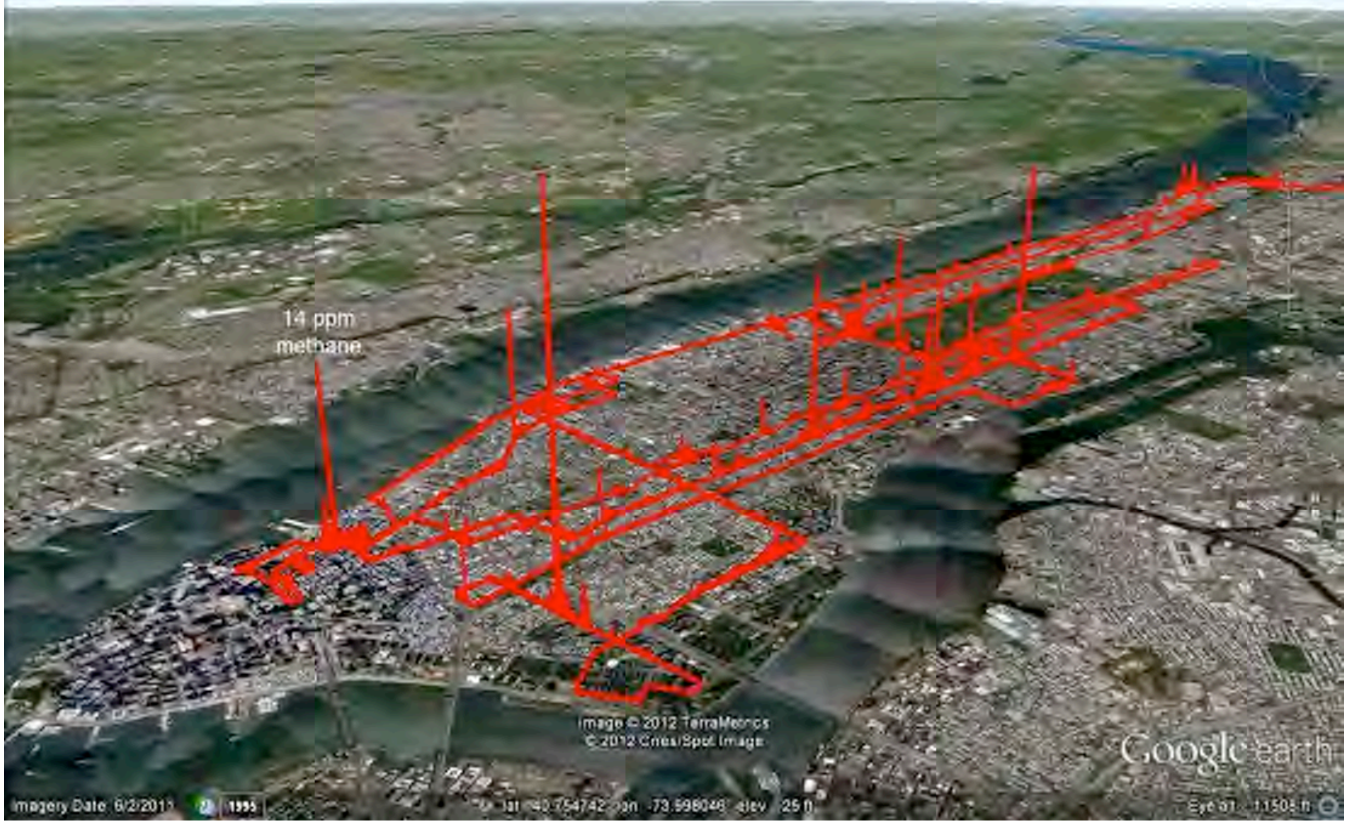
Six methane (natural gas) leaks were tested by inserting a gas probe approximately 6 inches through a valve box cover, pre-existing drill holes, or accessible manhole opening. All of these were likely Grade 2 leaks (in need of repair but not posing immediate danger of explosion) with combustible gas concentrations at the tested locations as follows: 0.35%, 15%, 55%, 55%, 67%, and 70%. Determining the exact location of a leak requires excavation of the probable leaking gas line until the exact location of the leak or leaks is determined. Such efforts were beyond the scope of this methane survey.

Image 6 was prepared from the survey data to provide a visualization of the potential relative importance of the methane leakage from the gas system in Manhattan on a regional atmospheric scale. Further work is needed to determine whether an approximate estimate of the amount of methane being released to the atmosphere can be developed from the data generated by this preliminary methane survey. For this initial report the following table presents a brief comparison of two randomly selected one-hour data sets for Manhattan and an open country drive. The methane measurements in Manhattan indicated many leaks (8.44% of all measurements were >2.5 ppm), some intense (measured levels up to 90 ppm), and almost no measurements at normal background methane levels (only 0.05% of the measurements were ≤2.0 ppm). In contrast, in the open country data, 86.37% of the measured methane levels were ≤2.0 ppm and only 0.03% in a range indicating substantial methane leaks or sources in the vicinity of the measurements.

Date-Time	1129-1959Z	1127-1514Z
Location	Manhattan	Open Country
Methane (ppm)		
Max	90.000	2.484
Mean	2.186	1.858
Min	1.897	1.787
Distribution of measured methane levels		
Total # measurements	13215	13101
% ≤ 2.0 ppm	0.050	86.370
% > 2.5 ppm	8.44	0.03

Work is planned for further analysis and interpretation of the data produced during this preliminary investigation. This report reveals the need and provides a foundation for additional work to better evaluate the apparently substantial amounts of methane being released into the atmosphere from pipeline leaks in New York City.

Image 1. Results of methane survey of parts of Manhattan on 27 November 2012



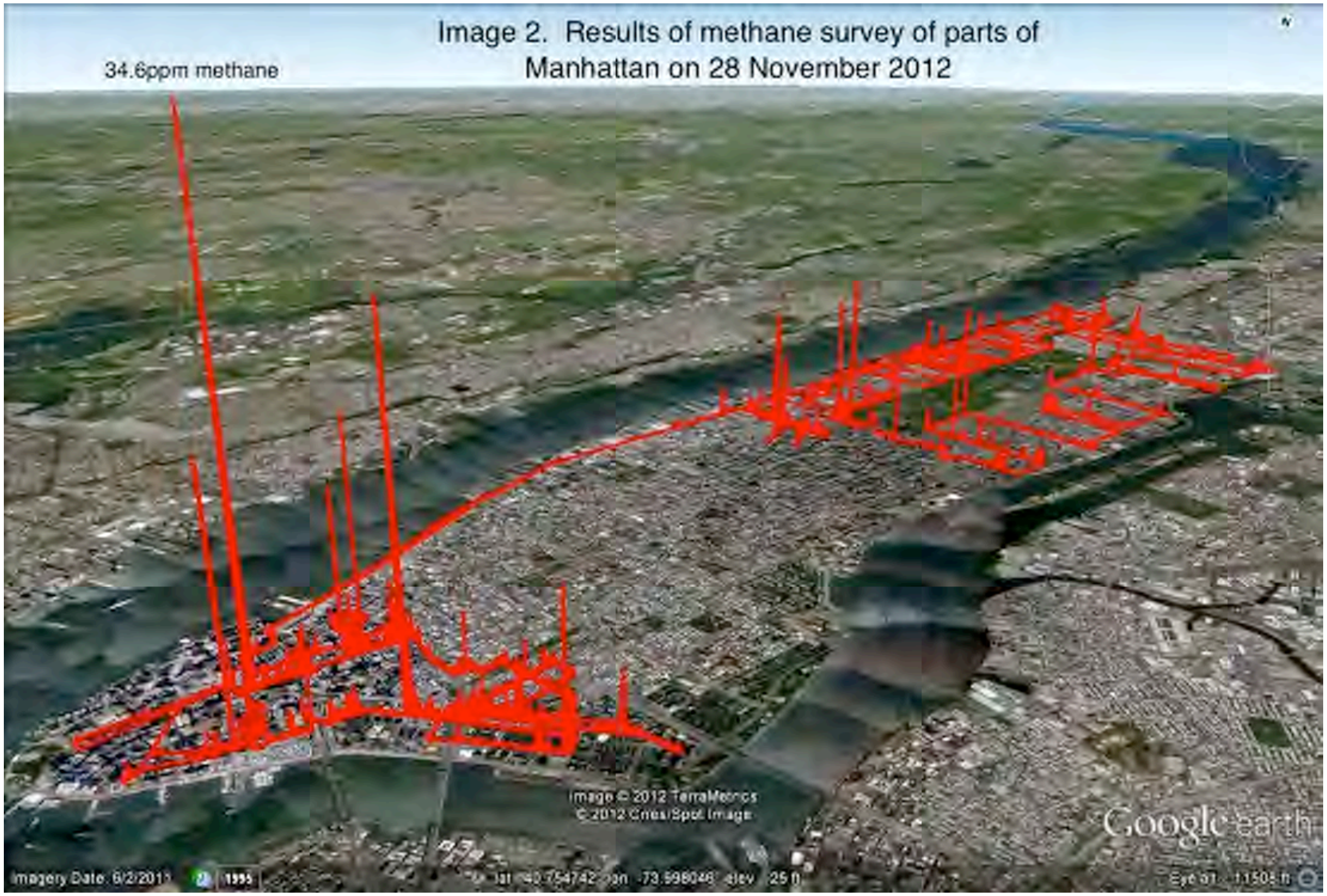


Image 2. Results of methane survey of parts of Manhattan on 28 November 2012

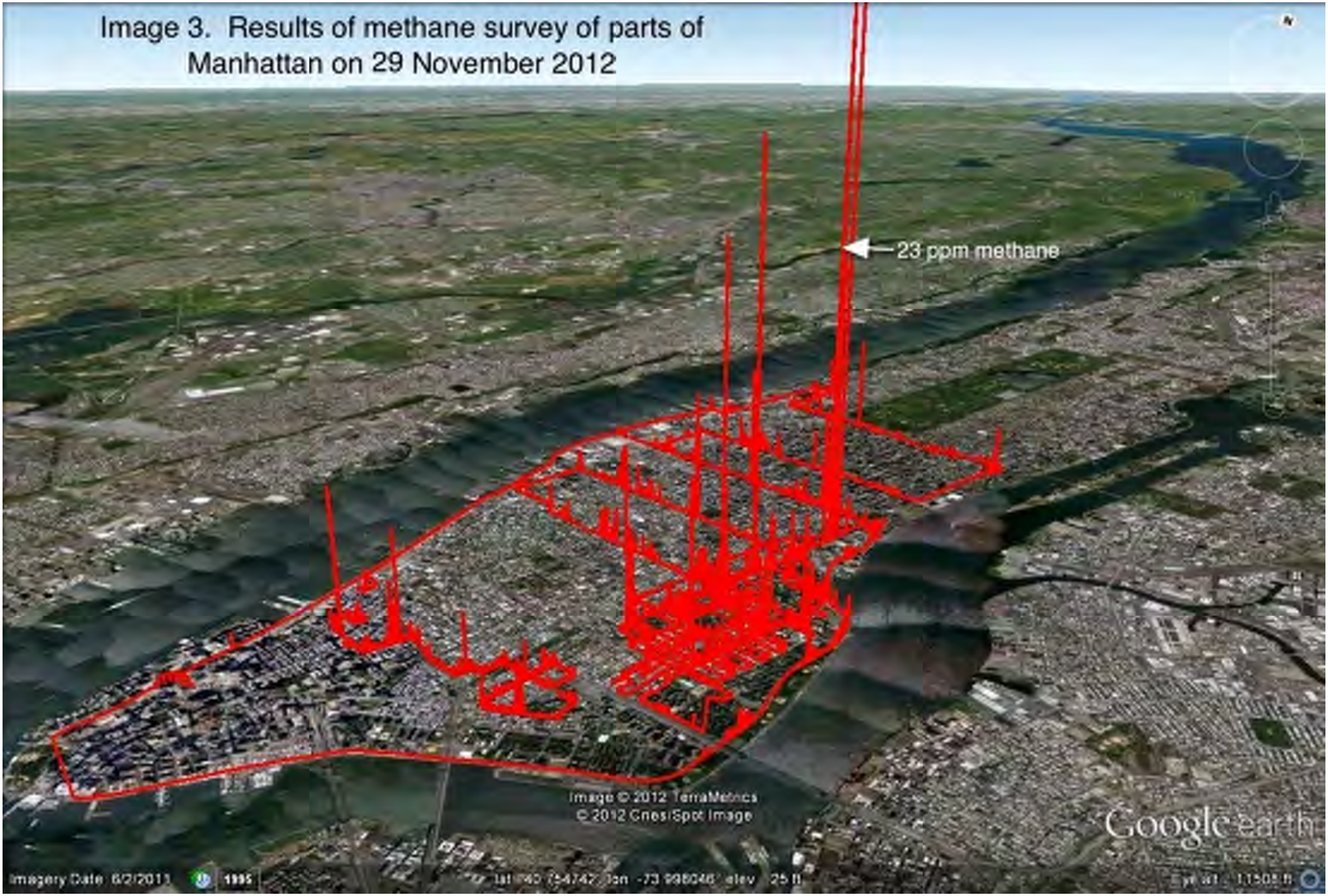


Image 3. Results of methane survey of parts of Manhattan on 29 November 2012

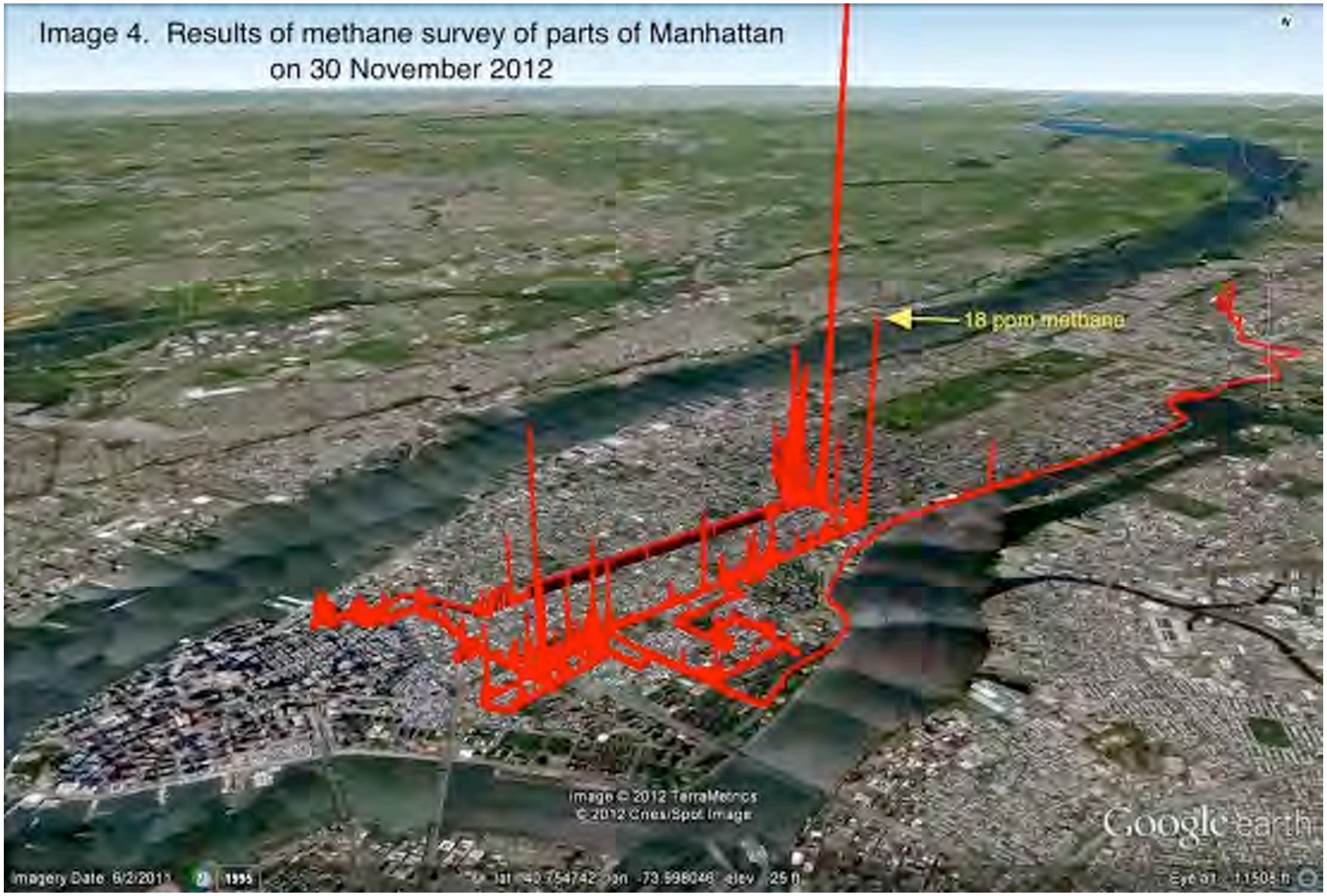


Image 4. Results of methane survey of parts of Manhattan on 30 November 2012

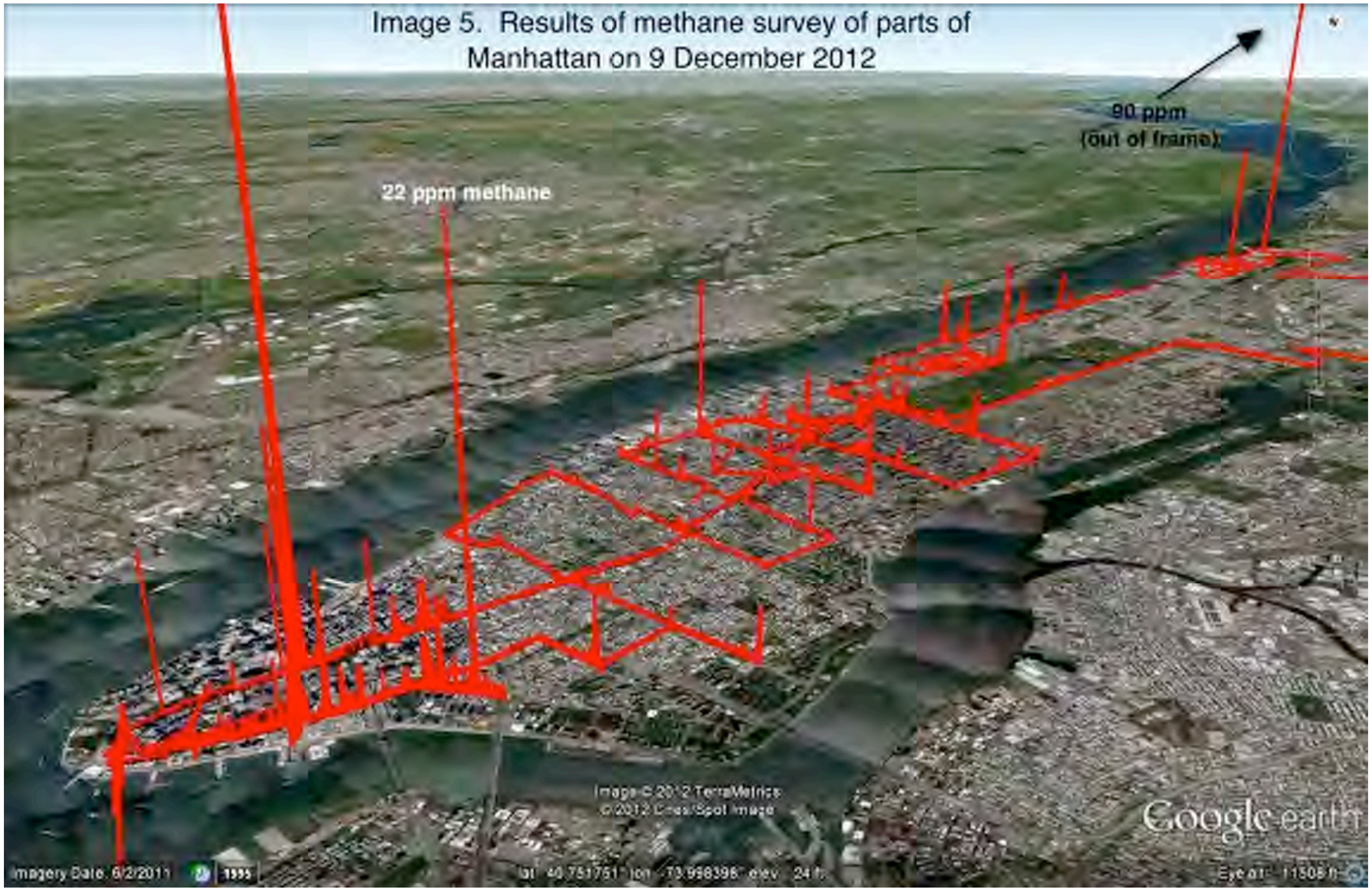
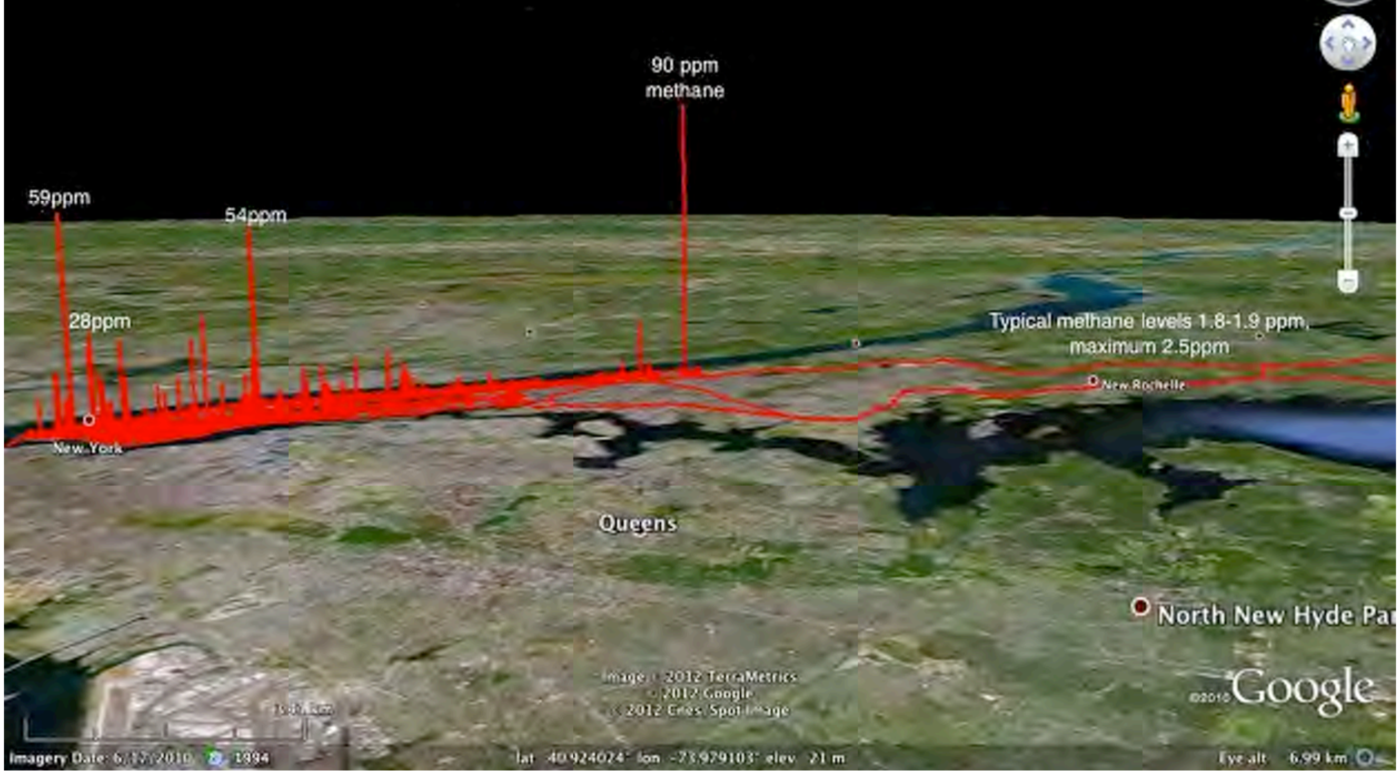


Image 5. Results of methane survey of parts of Manhattan on 9 December 2012

Image 6. Results of methane survey of parts of Manhattan and rural areas to the east on 27-30 November and 9 December 2012



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Venting and leaking of methane from shale gas development: response to Cathles et al.

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Abstract In April 2011, we published the first comprehensive analysis of greenhouse gas (GHG) emissions from shale gas obtained by hydraulic fracturing, with a focus on methane emissions. Our analysis was challenged by Cathles et al. (2012). Here, we respond to those criticisms. We stand by our approach and findings. The latest EPA estimate for methane emissions from shale gas falls within the range of our estimates but not those of Cathles et al. which are substantially lower. Cathles et al. believe the focus should be just on electricity generation, and the global warming potential of methane should be considered only on a 100-year time scale. Our analysis covered both electricity (30% of US usage) and heat generation (the largest usage), and we evaluated both 20- and 100-year integrated time frames for methane. Both time frames are important, but the decadal scale is critical, given the urgent need to avoid climate-system tipping points. Using all available information and the latest climate science, we conclude that for most uses, the GHG footprint of shale gas is greater than that of other fossil fuels on time scales of up to 100 years. When used to generate electricity, the shale-gas footprint is still significantly greater than that of coal at decadal time scales but is less at the century scale. We reiterate our conclusion from our April 2011 paper that shale gas is not a suitable bridge fuel for the 21st Century.

1 Introduction

Promoters view shale gas as a bridge fuel that allows continued reliance on fossil fuels while reducing greenhouse gas (GHG) emissions. Our April 2011 paper in *Climatic Change* challenged this view (Howarth et al. 2011). In the first comprehensive analysis of the GHG emissions from shale gas, we concluded that methane emissions lead to a large

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GHG footprint, particularly at decadal time scales. Cathles et al. (2012) challenged our work. Here, we respond to the criticisms of Cathles et al. (2012), and show that most have little merit. Further, we compare and contrast our assumptions and approach with other studies and with new information made available since our paper was published. After carefully considering all of these, we stand by the analysis and conclusions we published in Howarth et al. (2011).

2 Methane emissions during entire life cycle for shale gas and conventional gas

Cathles et al. (2012) state our methane emissions are too high and are "at odds with previous studies." We strongly disagree. Table 1 compares our estimates for both conventional gas and shale gas (Howarth et al. 2011) with 9 other studies, including 7 that have only become available since our paper was published in April 2011, listed chronologically by time of publication. See Electronic Supplementary Materials for details on conversions and calculations. Prior to our study, published estimates existed only for conventional gas. As we discussed in Howarth et al. (2011), the estimate of Hayhoe et al. (2002) is very close to our mean value for conventional gas, while the estimate from Jamarillo et al. (2007) is lower and should probably be considered too low because of their reliance on emission factors from a 1996 EPA report (Harrison et al. 1996). Increasing evidence over the past 15 years has suggested the 1996 factors were low (Howarth et al. 2011). In November 2010, EPA (2010) released parts of their first re-assessment of the 1996 methane emission factors, increasing some emissions factors by orders of magnitude. EPA (2011a), released just after our paper was published in April, used these new factors to re-assess and update the U.S. national GHG inventory, leading to a 2-fold increase in total methane emissions from the natural gas industry.

Table 1 Comparison of published estimates for full life-cycle methane emissions from conventional gas and shale gas, expressed per unit of Lower Heating Value (gC MJ^{-1}). Studies are listed by chronology of publication date

	Conventional gas	Shale gas
Hayhoe et al. (2002)	0.57	*
Jamarillo et al. (2007)	0.15	*
Howarth et al. (2011)	0.26–0.96	0.55–1.2
EPA (2011a)	0.38	0.60^{+}
Jiang et al. (2011)	*	0.30
Fulton et al.(2011)	0.38^{++}	*
Hultman et al. (2011)	0.35	0.57
Skone et al. (2011)	0.27	0.37
Burnham et al. (2011)	0.39	0.29
Cathles et al. (2012)	0.14–0.36	0.14–0.36

See Electronic Supplemental Materials for details on conversions

* Estimates not provided in these reports

$^{+}$ Includes emissions from coal-bed methane, and therefore may under-estimate shale gas emissions

$^{++}$ Based on average for all gas production in the US, not just conventional gas, and so somewhat over-estimates conventional gas emissions

The new estimate for methane emissions from conventional gas in the EPA (2011a) inventory, 0.38 g C MJ^{-1}, is within the range of our estimates: 0.26 to 0.96 g C MJ^{-1} (Table 1). As discussed below, we believe the new EPA estimate may still be too low, due to a low estimate for emissions during gas transmission, storage, and distribution. Several of the other recent estimates for conventional gas are very close to the new EPA estimate (Fulton et al. 2011; Hultman et al. 2011; Burnham et al. 2011). The Skone et al. (2011) value is 29% lower than the EPA estimate and is very similar to our lower-end number. Cathles et al. (2012) present a range of values, with their high end estimate of 0.36 g C MJ^{-1} being similar to the EPA estimate but their low end estimate (0.14 g C MJ^{-1}) far lower than any other estimate, except for the Jamarillo et al. (2007) estimate based on the old 1996 EPA emission factors.

For shale gas, the estimate derived from EPA (2011a) of 0.60 g C MJ^{-1} is within our estimated range of 0.55 to 1.2 g C MJ^{-1} (Table 1); as with conventional gas, we feel the EPA estimate may not adequately reflect methane emissions from transmission, storage, and distribution. Hultman et al. (2011) provide an estimate only slightly less than the EPA number. In contrast, several other studies present shale gas emission estimates that are 38% (Skone et al. 2011) to 50% lower (Jiang et al. 2011; Burnham et al. 2011) than the EPA estimate. The Cathles et al. (2012) emission estimates are 40% to 77% lower than the EPA values, and represent the lowest estimates given in any study.

In an analysis of a PowerPoint presentation by Skone that provided the basis for Skone et al. (2011), Hughes (2011a) concludes that a major difference between our work and that of Skone and colleagues was the estimated lifetime gas production from a well, an important factor since emissions are normalized to production. Hughes (2011a) suggests that Skone significantly overestimated this lifetime production, and thereby underestimated the emissions per unit of energy available from gas production (see Electronic Supplemental Materials). We agree, and believe this criticism also applies to Jiang et al. (2011). The lifetime production of shale-gas wells remains uncertain, since the shale-gas technology is so new (Howarth and Ingraffea 2011). Some industry sources estimate a 30-year lifetime, but the oldest shale-gas wells from high-volume hydraulic fracturing are only a decade old, and production of shale-gas wells falls off much more rapidly than for conventional gas wells. Further, increasing evidence suggests that shale-gas production often has been exaggerated (Berman 2010; Hughes 2011a, 2011b; Urbina 2011a, 2011b).

Our high-end methane estimates for both conventional gas and shale gas are substantially higher than EPA (2011a) (Table 1), due to higher emission estimates for gas storage, transmission, and distribution ("downstream" emissions). Note that our estimated range for emissions at the shale-gas wells ("upstream" emissions of 0.34 to 0.58 g C MJ^{-1}) agree very well with the EPA estimate (0.43 g C MJ^{-1}; see Electronic Supplementary Materials). While EPA has updated many emission factors for natural gas systems since 2010 (EPA 2010, 2011a, 2011b), they continue to rely on the 1996 EPA study for downstream emissions. Updates to this assumption currently are under consideration (EPA 2011a). In the meanwhile, we believe the EPA estimates are too low (Howarth et al. 2011). Note that the downstream emission estimates of Hultman et al. (2011) are similar to EPA (2011a), while those of Jiang et al. (2011) are 43% less, Skone et al. (2011) 38% less, and Burnham et al. (2011) 31% less (Electronic Supplemental Materials). One problem with the 1996 emission factors is that they were not based on random sampling or a comprehensive assessment of actual industry practices, but rather only analyzed emissions from model facilities run by companies that voluntarily participated (Kirchgessner et al. 1997). The average long-distance gas transmission pipeline in the U.S. is more than 50 years old, and many cities rely on gas distribution systems that are 80 to 100 years old, but these older systems were not part of the 1996 EPA assessment. Our range of estimates for methane emissions during gas storage, transmission, and distribution falls well within the range given by Hayhoe et al. (2002), and our mean estimate is virtually identical to their "best estimate" (Howarth et al. 2011). Nonetheless, we readily admit that these estimates are highly uncertain. There is an urgent need for better measurement of methane fluxes from all parts of the natural gas industry, but particularly during completion of unconventional wells and from storage, transmission, and distribution sectors (Howarth et al. 2011).

EPA proposed new regulations in October 2009 that would require regular reporting on GHG emissions, including methane, from natural gas systems (EPA 2011c). Chesapeake Energy Corporation, the American Gas Association, and others filed legal challenges to these regulations (Nelson 2011). Nonetheless, final implementation of the regulations seems likely. As of November 2011, EPA has extended the deadline for the first reporting to September 2012 (EPA 2011c). These regulations should help evaluate methane pollution, although actual measurements of venting and leakage rates will not be required, and the reporting requirement as proposed could be met using EPA emission factors. Field measurements across a range of well types, pipeline and storage systems, and geographic locations are important for better characterizing methane emissions.

3 How much methane is vented during completion of shale-gas wells?

During the weeks following hydraulic fracturing, frac-return liquids flow back to the surface, accompanied by large volumes of natural gas. We estimated substantial methane venting to the atmosphere at this time, leading to a higher GHG footprint for shale gas than for conventional gas (Howarth et al. 2011). Cathles et al. (2012) claim we are wrong and assert that methane emissions from shale-gas and conventional gas wells should be equivalent. They provide four arguments: 1) a physical argument that large flows of gas are not possible while frac fluids fill the well; 2) an assertion that venting of methane to the atmosphere would be unsafe; 3) a statement that we incorrectly used data on methane capture during flowback to estimate venting; and 4) an assertion that venting of methane is not in the economic interests of industry. We disagree with each point, and note our methane emission estimates during well completion and flowback are quite consistent with both those of EPA (2010, 2011a, b) and Hultman et al. (2011).

Cathles et al. state that gas venting during flowback is low, since the liquids in the well interfere with the free flow of gas, and imply that this condition continues until the well goes into production. While it is true that liquids can restrict gas flow early in the flow-back period, gas is freely vented in the latter stages. According to EPA (2011d), during well cleanup following hydraulic fracturing "backflow emissions are a result of free gas being produced by the well during well cleanup event, when the well also happens to be producing liquids (mostly water) and sand. The high rate backflow, with intermittent slugs of water and sand along with free gas, is typically directed to an impoundment or vessels until the well is fully cleaned up, where the free gas vents to the atmosphere while the water and sand remain in the impoundment or vessels." The methane emissions are "vented as the backflow enters the impoundment or vessels" (EPA 2011d). Initial flowback is 100% liquid, but this quickly becomes a two-phase flow of liquid and gas as backpressure within the fractures declines (Soliman & Hunt 1985; Willberg et al. 1998; Yang et al. 2010; EPA 2011a, d). The gas produced is not in solution, but rather is free-flowing with the liquid in this frothy mix. The gas cannot be put into production and sent to sales until flowback rates are sufficiently decreased to impose pipeline pressure.

Is it unsafe for industry to vent gas during flowback, as Cathles et al. assert? Perhaps, but venting appears to be common industry practice, and the latest estimates from EPA (2011b, page 3–12) are that 85% of flowback gas from unconventional wells is vented and less than 15% flared or captured. While visiting Cornell, a Shell engineer stated Shell never flares gas during well completion in its Pennsylvania Marcellus operations (Bill Langin, pers. comm.). Venting of flow-back methane is clearly not as unsafe as Cathles et al. (2012) believe, since methane has a density that is only 58% that of air and so would be expected to be extremely buoyant when vented. Under sufficiently high wind conditions, vented gas may be mixed and advected laterally rather than rising buoyantly, but we can envision no atmospheric conditions under which methane would sink into a layer over the ground. Buoyantly rising methane is clearly seen in Forward Looking Infra Red (FLIR) video of a Pennsylvania well during flowback (Fig. 1). Note that we are not using this video information to infer any information on the rate of venting, but simply to illustrate that venting occurred in the summer of 2011 in Pennsylvania and that the gas rose rapidly into the atmosphere. Despite the assertion by Cathles et al. that venting is illegal in Pennyslvania, the only legal restriction is that "excess gas encountered during drilling, completion or stimulation shall be flared, captured, or diverted away from the drilling rig in a manner than does not create a hazard to the public health or safety" (PA § 78.73. *General provision for well construction and operation*).

Cathles et al. state with regard to our paper: "The data they cite to support their contention that fugitive methane emissions from unconventional gas production is [sic] significantly greater than that from conventional gas production are actually estimates of gas emissions that were captured for sale. The authors implicitly assume that capture (or even flaring) is rare, and that the gas captured in the references they cite is normally vented directly into the atmosphere." We did indeed use data on captured gas as a surrogate for vented emissions, similar to such interpretation by EPA (2010). Although most flowback gas appears to be vented and not captured (EPA 2011b), we are aware of no data on the rate of venting, and industry apparently does not usually measure or estimate the gas that is vented during flowback. Our assumption (and that of EPA 2010) is that the rate of gas flow is the same during flowback, whether vented or captured. Most of the data we used were reported to the EPA as part of their "green completions" program, and they provide some of the very few publicly available quantitative estimates of methane flows at the time of flowback. Note that the estimates we published in Howarth et al. (2011) for emissions at the time of well completion for shale gas could be reduced by 15%, to account for the estimated average percentage of gas that is not vented but

Fig. 1 Venting of natural gas into the atmosphere at the time of well completion and flowback following hydraulic fracturing of a well in Susquehanna County, PA, on June 22, 2011. Note that this gas is being vented, not flared or burned, and the color of the image is to enhance the IR image of this methane-tuned FLIR imagery. The full video of this event is available at http://www.psehealthyenergy.org/resources/view/198782. Video provided courtesy of Frank Finan

rather is flared or captured and sold (EPA 2011b). Given the other uncertainty in these estimates, though, our conclusions would remain the same.

Cathles et al. also assert that we used initial production rates for gas wells, and that in doing so over-estimated flowback venting. Our estimates of flowback emissions for the Barnett, Piceance, Uinta, and Denver-Jules basins were not based on initial production rates, but rather solely on industry-reported volumes of gas captured, assuming. We estimated emissions for the Haynesville basin as the median of data given in Eckhardt et al. (2009), who reported daily rates ranging from 400,000 m^3 (14 MMcf) to 960,000 m^3 (38 MMcf). We assumed a 10-day period for the latter part of the flowback in which gases freely flow, the mean for the other basin studies we used. The use of initial production rates applied to the latter portion of flowback duration as an estimate of venting is commonly accepted (Jiang et al. 2011; NYS DEC 2011).

Finally, Cathles et al. state that economic self-interest would make venting of gas unlikely. Rather, they assert industry would capture the gas and sell it to market. According to EPA (2011b), the break-even price at which the cost of capturing flowback gas equals the market value of the captured gas is slightly under $4 per thousand cubic feet. This is roughly the well-head price of gas over the past two years, suggesting that indeed industry would turn a profit by capturing the gas, albeit a small one. Nonetheless, EPA (2011b) states that industry is not commonly capturing the gas, probably because the rate of economic return on investment for doing so is much lower than the normal expectation for the industry. That is, industry is more likely to use their funds for more profitable ventures than capturing and selling vented gas (EPA 2011b). There also is substantial uncertainty in the cost of capturing the gas. At least for low-energy wells, a BP presentation put the cost of "green" cleanouts as 30% higher than for normal well completions (Smith 2008). The value of the captured gas would roughly pay for the process, according to BP, at the price of gas as of 2008, or approximately $6.50 per thousand cubic feet (EIA 2011a). At this cost, industry would lose money by capturing and selling gas not only at the current price of gas but also at the price forecast for the next 2 decades (EPA 2011b).

In July 2011, EPA (2011b, e) proposed new regulations to reduce emissions during flowback. The proposed regulation is aimed at reducing ozone and other local air pollution, but would also reduce methane emissions. EPA (2011b, e) estimates the regulation would reduce flowback methane emissions from shale gas wells by up to 95%, although gas capture would only be required for wells where collector pipelines are already in place, which is often not the case when new sites are developed. Nonetheless, this is a very important step, and if the regulation is adopted and can be adequately enforced, will reduce greatly the difference in emissions between shale gas and conventional gas in the U.S. We urge universal adoption of gas-capture policies.

To summarize, most studies conclude that methane emissions from shale gas are far higher than from conventional gas: approximately 40% higher, according to Skone et al. (2011) and using the mean values from Howarth et al. (2011), and approximately 60% higher using the estimates from EPA (2011a) and Hultman et al. (2011). Cathles et al. assertion that shale gas emissions are no higher seems implausible to us. The suggestion by Burnham et al. (2011) that shale gas methane emissions are less than for conventional gas seems even less plausible (see Electronic Supplementary Materials).

4 Time frame and global warming potential of methane

Methane is a far more powerful GHG than carbon dioxide, although the residence time for methane in the atmosphere is much shorter. Consequently, the time frame for comparing methane and carbon dioxide is critical. In Howarth et al. (2011), we equally presented two time frames, the 20 and 100 years integrated time after emission, using the global warming potential (GWP) approach. Note that GWPs for methane have only been estimated at time scales of 20, 100, and 500 years, and so GHG analyses that compare methane and carbon dioxide on other time scales require a more complicated atmospheric modeling approach, such as that used by Hayhoe et al. (2002) and Wigley (2011). The GWP approach we follow is quite commonly used in GHG lifecycle analyses, sometimes considering both 20-year and 100-year time frames as we did (Lelieveld et al. 2005; Hultman et al. 2011), but quite commonly using only the 100-year time frame (Jamarillo et al. 2007; Jiang et al. 2011; Fulton et al. 2011; Skone et al. 2011; Burnham et al. 2011). Cathles et al. state that a comparison based on the 20-year GWP is inappropriate, and criticize us for having done so. We very strongly disagree.

Considering methane's global-warming effects at the decadal time scale is critical (Fig. 2). Hansen et al. (2007) stressed the need for immediate control of methane to avoid critical tipping points in the Earth's climate system, particularly since methane release from permafrost becomes increasingly likely as global temperature exceeds 1.8°C above the

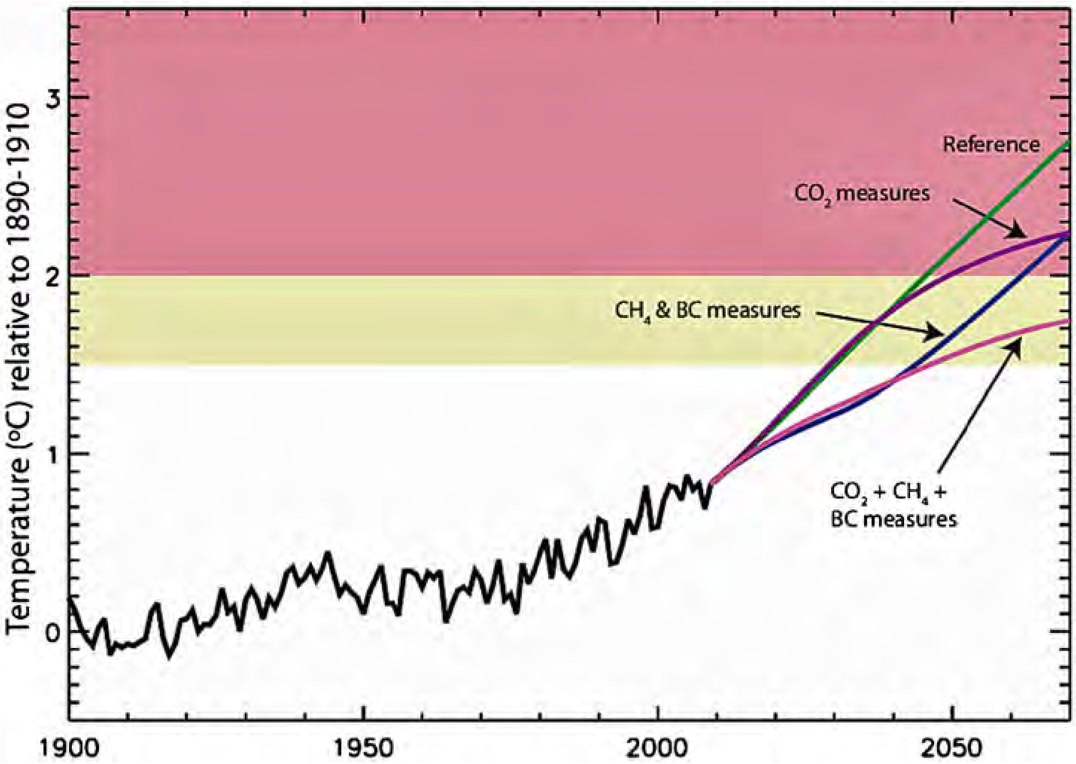

Fig. 2 Observed global mean temperature from 1900 to 2009 and projected future temperature under four scenarios, relative to the mean temperature from 1890–1910. The scenarios include the IPCC (2007) reference, reducing carbon dioxide emissions but not other greenhouse gases ("CO_2 measures"), controlling methane and black carbon emissions but not carbon dioxide ("CH_4 + BC measures"), and reducing emissions of carbon dioxide, methane, and black carbon ("CO_2 + CH_4 + BC measures"). An increase in the temperature to 1.5° to 2.0°C above the 1890–1910 baseline (illustrated by the yellow bar) poses high risk of passing a tipping point and moving the Earth into an alternate state for the climate system. The lower bound of this danger zone, 1.5° warming, is predicted to occur by 2030 unless stringent controls on methane and black carbon emissions are initiated immediately. Controlling methane and black carbon shows more immediate results than controlling carbon dioxide emissions, although controlling all greenhouse gas emissions is essential to keeping the planet in a safe operating space for humanity. Reprinted from UNEP/WMO (2011)

baseline average temperature between 1890 and 1910 (Hansen and Sato 2004; Hansen et al. 2007). This could lead to a rapidly accelerating positive feedback of further global warming (Zimov et al. 2006; Walter et al. 2007). Shindell et al. (2012) and a recent United Nations study both conclude that this 1.8°C threshold may be reached within 30 years unless societies take urgent action to reduce the emissions of methane and other short-lived greenhouse gases now (UNEP/WMO 2011). The reports predict that the lower bound for the danger zone for a temperature increase leading to climate tipping points – a 1.5°C increase – will occur within the next 18 years or even less if emissions of methane and other short-lived radiatively active substances such as black carbon are not better controlled, beginning immediately (Fig. 2) (Shindell et al. 2012; UNEP/WMO 2011).

In addition to different time frames, studies have used a variety of GWP values. We used values of 105 and 33 for the 20- and 100-year integrated time frames, respectively (Howarth et al. 2011), based on the latest information on methane interactions with other radiatively active materials in the atmosphere (Shindell et al. 2009). Surprisingly, EPA (2011a) uses a value of 21 based on IPCC (1995) rather than higher values from more recent science (IPCC 2007; Shindell et al. 2009). Jiang et al. (2011), Fulton et al. (2011), Skone et al. (2011), and Burnham et al. (2011) all used the 100-year GWP value of 25 from IPCC (2007), which underestimates methane's warming at the century time scale by 33% compared to the more recent GWP value of 33 from Shindell et al. (2009). We stand by our use of the higher GWP values published by Shindell et al. (2009), believing it appropriate to use the best and most recent science. While there are considerable uncertainties in GWP estimates, inclusion of the suppression of photosynthetic carbon uptake due to methane-induced ozone (Sitch et al. 2007) would further increase methane's GWP over all the values discussed here.

In Fig. 3, we present the importance of methane to the total GHG inventory for the US, considered at both the 20- and 100-year time periods, and using the Shindell et al. (2009) GWP values. Figure 3 uses the most recently available information on methane fluxes for the 2009 base year, reflecting the new methane emission factors and updates through July 2011 (EPA 2010; 2011a, b); see Electronic Supplemental Materials. Natural gas systems dominate the methane flux for the US, according to these EPA estimates, contributing 39% of the nation's total. And methane contributes 19% of the entire GHG inventory of the US at the century time scale and 44% at the 20-year scale, including all gases and all human activities. The methane emissions from natural gas systems make up 17% of the entire anthropogenic GHG inventory of the US, when viewed through the lens of the 20-year integrated time frame. If our high-end estimate for downstream methane emissions during gas storage, transmission, and distribution is correct (Howarth et al. 2011), the importance of methane from natural gas systems would be even greater.

5 Electricity vs. other uses

Howarth et al. (2011) focused on the GHG footprint of shale gas and other fuels normalized to heat from the fuels, following Lelieveld et al. (2005) for conventional gas. We noted that for electricity generation – as opposed to other uses of natural gas – the greater efficiency for gas shifts the comparison somewhat, towards the footprint of gas being less unfavorable. Nonetheless, we concluded shale gas has a larger GHG footprint than coal even when used to generate electricity, at the 20-year time horizon (Howarth et al. 2011). Hughes (2011b) further explored the use of shale gas for electricity generation, and supported our conclusion. Cathles et al. criticize us for not focusing exclusively on electricity.

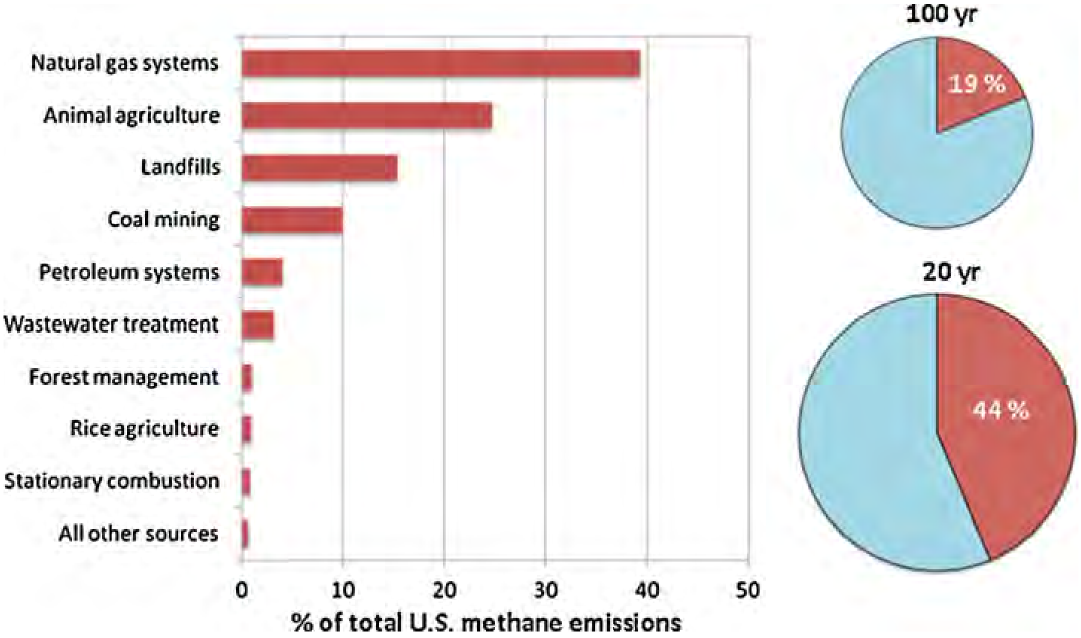

Fig. 3 Environmental Protection Agency estimates for human-controlled sources of methane emission from the U.S. in 2009 (bar graph) and percent contribution of methane to the entire greenhouse gas inventory for the U.S. (shown in red on the pie charts) for the 100-year and 20-year integrated time scales. The sizes of the pie charts are proportional to the total greenhouse gas emission for the U.S. in 2009. The methane emissions represent a greater portion of the warming potential when converted to equivalents of mass of carbon dioxide at the shorter time scale, which increases both the magnitude of the total warming potential and the percentage attributed to methane. Data are from EPA (2011a, b), as discussed in Electronic Supplemental Material, and reflect an increase over the April 2011 national inventory estimates due to new information on methane emissions from Marcellus shale gas and tight-sand gas production for 2009 (EPA 2011b). Animal agriculture estimate combines enteric fermentation with manure management. Coal mining combines active mines and abandoned mines. The time-frame comparisons are made using the most recent data on global warming potentials from Shindell et al. (2009)

We stand by our focus on GHG emissions normalized to heat content. Only 30% of natural gas in the U.S. is used to generate electricity, while most is used for heat for domestic, commercial, and industrial needs, and this pattern is predicted to hold over coming decades (EIA 2011b; Hughes 2011b). Globally, demand for heat is the largest use of energy, at 47% of use (International Energy Agency 2011). And natural gas is the largest source of heat globally, providing over half of all heat needs in developed countries (International Energy Agency 2011). While generating electricity from natural gas has some efficiency gains over using coal, we are aware of no such advantage for natural gas over other fossil fuels for providing heat.

Many view use of natural gas for transportation as an important part of an energy future. The “Natural Gas Act” (H.R.1380) introduced in Congress in 2011 with bipartisan support and the support of President Obama would provide tax subsidies to encourage long-distance trucks to switch from diesel to natural gas (Weiss and Boss 2011). And in Quebec, industry claims converting trucks from diesel to shale gas could reduce GHG emissions by 25 to 30% (Beaudine 2010). Our study suggests this claim is wrong and indicates shale gas has a larger GHG footprint than diesel oil, particularly over the 20-year time frame (Howarth et al. 2011). In fact, using natural gas for long-distance trucks may be worse than our analysis suggested, since it would likely depend on liquefied natural gas, LNG. GHG emissions from LNG are far higher than for non-liquified gas (Jamarillo et al. 2007). See Electronic Supplemental Materials for more information on future use of natural gas in the U.S.

6 Conclusions

We stand by our conclusions in Howarth et al. (2011) and see nothing in Cathles et al. and other reports since April 2011 that would fundamentally change our analyses. Our methane emission estimates compare well with EPA (2011a), although our high-end estimates for emissions from downstream sources (storage, transmission, distribution) are higher. Our estimates also agree well with earlier papers for conventional gas (Hayhoe et al. 2002; Lelieveld et al. 2005), including downstream emissions. Several other analyses published since April of 2011 have presented significantly lower emissions than EPA estimates for shale gas, including Cathles et al. but also Jiang et al. (2011), Skone et al. (2011), and Burnham et al. (2011). We believe these other estimates are too low, in part due to overestimation of the lifetime production of shale-gas wells.

We reiterate that all methane emission estimates, including ours, are highly uncertain. As we concluded in Howarth et al. (2011), "the uncertainty in the magnitude of fugitive emissions is large. Given the importance of methane in global warming, these emissions deserve far greater study than has occurred in the past. We urge both more direct measurements and refined accounting to better quantify lost and unaccounted for gas." The new GHG reporting requirements by EPA will provide better information, but much more is needed. Governments should encourage and fund independent measurements of methane venting and leakage. The paucity of such independent information is shocking, given the global significance of methane emissions and the potential scale of shale gas development.

We stress the importance of methane emissions on decadal time scales, and not focusing exclusively on the century scale. The need for controlling methane is simply too urgent, if society is to avoid tipping points in the planetary climate system (Hansen et al. 2007; UNEP/WMO 2011; Shindell et al. 2012). Our analysis shows shale gas to have a much larger GHG footprint than conventional natural gas, oil, or coal when used to generate heat and viewed over the time scale of 20 years (Howarth et al. 2011). This is true even using our low-end methane emission estimates, which are somewhat lower than the new EPA (2011a) values and comparable to those of Hultman et al. (2011). At this 20-year time scale, the emissions data from EPA (2011a, b) show methane makes up 44% of the entire GHG inventory for the U.S., and methane from natural gas systems make up 17% of the entire GHG inventory (39% of the methane component of the inventory).

We also stress the need to analyze the shale-gas GHG footprint for all major uses of natural gas, and not focus on the generation of electricity alone. Of the reports published since our study, only Hughes (2011b) seriously considered heat as well as electricity. Cathles et al. (2012), Jiang et al. (2011), Fulton et al. (2011), Hultman et al. (2011), Skone et al. (2011), and Wigley (2011) all focus just on the generation of electricity. We find this surprising, since only 30% of natural gas in the U.S. is used to generate electricity. Other uses such as transportation should not be undertaken without fully understanding the consequences on GHG emissions, and none of the electricity-based studies provide an adequate basis for such evaluation.

Can shale-gas methane emissions be reduced? Clearly yes, and proposed EPA regulations to require capture of gas at the time of well completions are an important step. Regulations are necessary to accomplish emission reductions, as economic considerations alone have not driven such reductions (EPA 2011b). And it may be extremely expensive to reduce leakage associated with aging infrastructure, particularly distribution pipelines in cities but also long-distance transmission pipelines, which are on average more than 50 years old in the U.S. Should society invest massive capital in such improvements for a bridge fuel that is to be used for only 20 to 30 years, or would the capital be better spent on constructing a smart electric grid and other technologies that move towards a truly green energy future?

We believe the preponderance of evidence indicates shale gas has a larger GHG footprint than conventional gas, considered over any time scale. The GHG footprint of shale gas also exceeds that of oil or coal when considered at decadal time scales, no matter how the gas is used (Howarth et al. 2011; Hughes 2011a, b; Wigley et al. 2011). Considered over the century scale, and when used to generate electricity, many studies conclude that shale gas has a smaller GHG footprint than coal (Wigley 2011; Hughes 2011b; Hultman et al. 2011), although some of these studies biased their result by using a low estimate for GWP and/or low estimates for methane emission (Jiang et al. 2011; Skone et al. 2011; Burnham et al. 2011). However, the GHG footprint of shale gas is similar to that of oil or coal at the century time scale, when used for other than electricity generation. We stand by the conclusion of Howarth et al. (2011): “The large GHG footprint of shale gas undercuts the logic of its use as a bridging fuel over coming decades, if the goal is to reduce global warming.”

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Natural Gas & Climate Change

Eric D. Larson, PhD

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Key Findings

Knowing how much methane is leaking from the natural gas system is essential to determining the potential climate benefits of natural gas use. Climate Central's extensive review of the publicly available studies finds that a pervasive lack of measurements makes it nearly impossible to know with confidence what the average methane leak rate is for the U.S. as a whole. More measurements, more reliable data, and better understanding of industry practices are needed.

It has been widely reported that shifting from coal to gas in electricity generation will provide a 50 percent reduction in greenhouse gas emissions. In reality, the extent of reduced global warming impact depends largely on three factors:

1. The methane leak rate from the natural gas system;
2. How much time has passed after switching from coal to gas, because the potency of methane as a greenhouse gas is 102 times that of carbon dioxide (on a pound-for-pound basis) when first released into the atmosphere and decays to 72 times CO_2 over 20 years and to 25 times CO_2 over 100 years, and;
3. The rate at which coal electricity is replaced by gas electricity.

Climate Central has developed an interactive graphic incorporating all three factors. This makes it easy to visualize the greenhouse benefits of converting power generation from coal to natural gas for different assumptions of methane leak rates and coal-to-gas conversion rates while also considering methane's greenhouse potency over time.

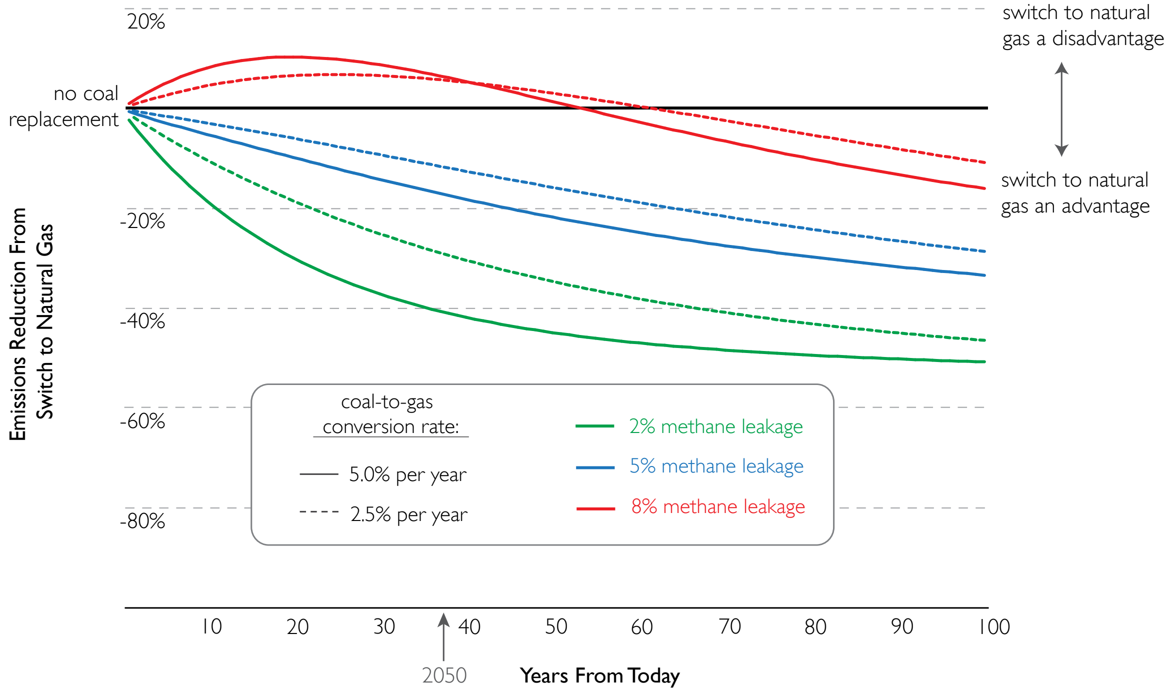
The EPA recently estimated methane leaks in the natural gas system at 1.5 percent. A 1.5 percent leak rate would achieve an immediate 50 percent reduction in greenhouse gas (GHG) emissions, at the individual power plant level. However, EPA's estimate contains significant uncertainty, and like all estimates available in the peer-reviewed literature, lacks sufficient real-world measurements to guide decision-making at the national level. Climate Central found that the ongoing shift from coal to gas in power generation in the U.S. is unlikely to provide the 50 percent reduction in GHG emissions typically attributed to it over the next three to four decades, unless gas leakage is maintained at the lowest estimated rates (1 to 1.5 percent) and the coal replacement rate is maintained at recent high levels (greater than 5 percent per year).

The climate benefits of natural gas are sensitive to small increases in leak rates. Assuming that natural gas replaces 2.5 percent of coal-fired power each year (the average over the past decade) even a relatively low overall leak rate of 2 percent would not achieve a 50 percent reduction in GHG emissions compared to the current fleet of coal-fired power plants, for over 100 years. If the leak rate were as high as 8 percent, there would be no climate benefit at all from switching to natural gas for more than 60 years.

To compute these estimates, we analyzed first the potential GHG benefits from replacing the electricity generated by a single coal power plant with electricity from natural gas instead. For an individual power plant, if the leak rate were 2 percent it would take 55 years to reach a 50 percent reduction in greenhouse impacts compared to continued coal use. If the leak rate is more than 6 percent of methane production, switching to natural gas provides zero global warming benefit for the first 5 years compared to continuing with coal. The switch achieves a modest 17 percent reduction in GHG emissions after 37 years (or by 2050, if the switch occurs in 2013). An 8 percent leak rate increases GHG emissions until 2050 compared with continued coal use, and produces only about 20 percent less climate pollution than continued coal use after 100 years of operation.

But unlike converting a single power plant from coal to natural gas, the U.S. cannot switch its entire fleet of coal-fired power plants to natural gas all at once. When substitution is analyzed across the entire fleet of coal-fired plants, the rate of adoption of natural gas is a critical factor in achieving greenhouse benefits. The rate of adoption is analyzed together with the powerful but declining potency of methane emissions over time. Each year, as a certain percentage

It will be Decades Before Switching to Natural Gas From Coal Power Brings a 50 Percent Reduction in Emissions



of coal plants are converted to natural gas, a new wave of highly potent methane leaks into the atmosphere and then decreases in potency over time.

When the rate of adoption is included, the GHG benefits of switching to natural gas can be even more elusive. With a 2 percent methane leak rate, and an average annual conversion rate of electricity from coal to gas of 2.5 percent (a rate that would be supportable with new gas production projected by the U.S. Department of Energy) the reductions would be 29 percent by 2050 and 16 percent by 2030. If methane leakage is 5 percent of production, by 2050 the U.S. would reduce the global warming impact of its fleet of coal fired power plants by 12 percent. By 2030, the reductions would be just 5 percent. With an 8 percent leak rate, GHG emissions would be greater than with coal for more than 50 years before a benefit begins to be realized.

What is the natural gas leak rate in the U.S.? There are large differences among published estimates of leakage from the natural gas supply system, from less than 1 percent of methane production to as much as 8 percent. At the basin level, studies have reported methane leak rates as high as 17 percent. The EPA's 2012 annual greenhouse gas emissions inventory estimate was 2.2 percent. Its 2013 inventory estimate made a large adjustment that reduced the estimate to 1.5 percent. The degree of methane leakage is uncertain, but it is likely to be reduced in the future since it also represents lost profits for gas companies. Nevertheless, our analysis indicates that the ongoing shift from coal to gas in power generation in the U.S. over the next three to four decades is unlikely to provide the 50 percent benefit that is typically attributed to such a shift.

Determining methane leakage is complicated by various uncertainties:

- Large variability and uncertainty in industry practices at wellheads, including:
    - Whether methane that accompanies flowback of hydraulic fracking fluid during completion of shale gas wells is captured for sale, flared, or vented at the wellhead. Industry practices appear to vary widely.

    - Liquids unloading, which must be done multiple times per year at most conventional gas wells and at some shale gas wells. Gas entrained with the liquids may be vented to the atmosphere. There have been relatively few measurements of vented gas volumes, and estimating an average amount of methane emitted per unloading is difficult due to intrinsic variations from well to well.

- Lack of sufficient production experience with shale gas wells:

    - There are orders of magnitude in variability of estimates of how much gas will ultimately be recovered from any given shale well. This makes it difficult to define an average lifetime production volume per well, which introduces uncertainty in estimating the percentage of gas leaked over the life of an average well.

    - The frequency with which a shale gas well must be re-fractured to maintain gas flow. This process, known as a well workover, can result in methane emissions. The quantity of emissions per workover is an additional uncertainty, as it depends on how workover gas flow is handled.

- The leak integrity of the large and diverse gas distribution infrastructure:

    - Leakage measurements are challenging due to the large extent of the distribution system, including more than a million miles of distribution mains, more than 60 million service line connections, and thousands of metering and regulating stations operating under varying gas pressures and other conditions.

    - Recent measurements of elevated methane concentrations in the air above streets in Boston, San Francisco and Los Angeles strongly suggest distribution system leakages. Additional measurements are needed to estimate leak rates based on such measurements.

Report in Brief

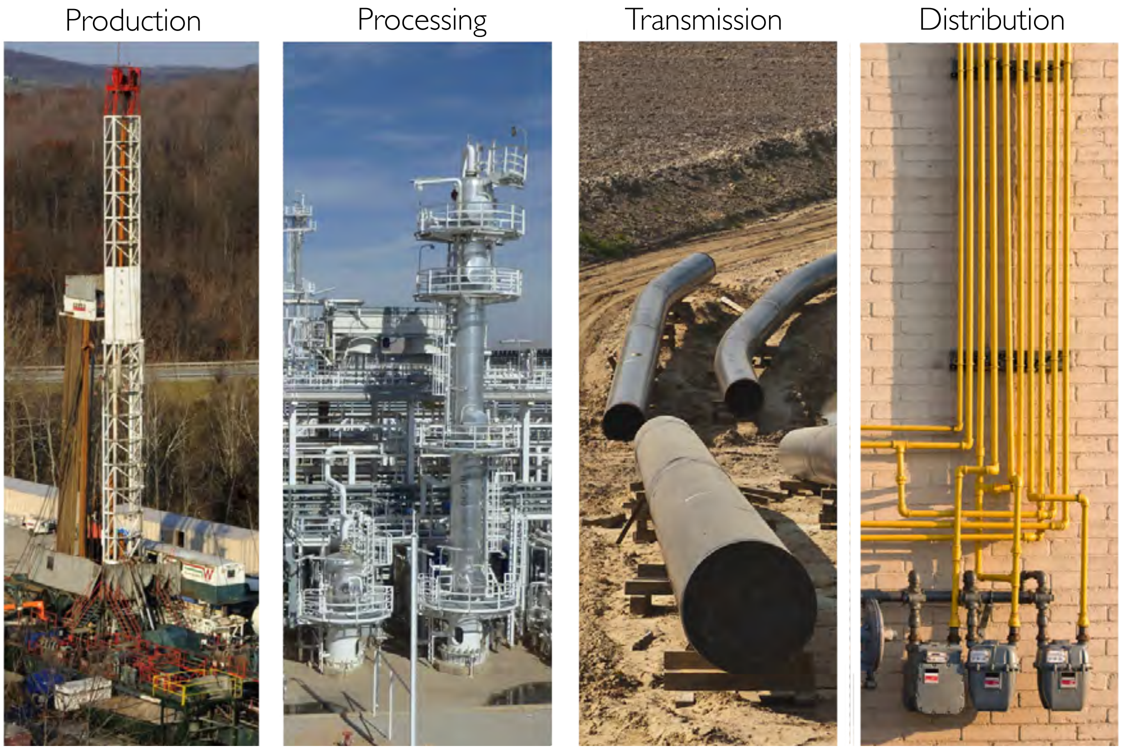
Natural gas use in the U.S. grew by 25 percent from 2007 to 2012. Within the power sector natural gas use grew from 30 percent to 36 percent of all gas use. Shale gas produced by hydraulic fracturing has grown especially rapidly, from close to zero a decade ago to about one-third of all gas today. Continued growth is projected, and shale gas could account for half of all gas in another two decades.

As gas production has grown, electricity generated using gas has grown, from less than 19 percent of all electricity in 2005 to more than 30 percent in 2012. During the same period coal electricity fell from 50 percent to 37 percent. Many associate the shift from coal to gas with significant reductions in U.S. greenhouse gas emissions from electricity because of the lower carbon content of natural gas compared to coal and the higher efficiency with which gas can be converted to electricity.

However, the main component of natural gas, methane, is a much stronger global warming gas than CO_2, and any methane leakage to the atmosphere from the natural gas supply system offsets some of the carbon benefit of a coal-to-gas shift. Here we review a wide set of studies that have been published and provide analysis to put the question of methane leakage in perspective: Depending on the rate of methane leakage, how much more climate friendly is natural gas than coal for electricity generation, and how does the rate at which gas is substituted for coal change that answer?

The two most recent official estimates of U.S. methane emissions from the natural gas supply system (published by the EPA) are that from 1.5 percent to 2.2 percent of methane extracted from the ground in 2010 leaked to the atmosphere, from well drilling and production, through gas processing, transmission, and final distribution to end users.

The range in the EPA's leakage estimates and our review of a large number of others' methane leakage estimates indicate significant uncertainty in the leakage rate. The largest uncertainties are for the production and distribution stages. Peer-reviewed studies, which have focused almost exclusively on assessing leakage rates in the first three stages (excluding distribution), have estimated average leakage for these three stages from less than 1 percent up to 4.5


Production

Processing

Transmission

Distribution

Figure 1. *The four stages of the U.S. natural gas supply system.*

percent of gas produced, with uncertainty bands extending this range on the high end up to as much as 7 percent. The production stage in most studies accounts for 60 to 85 percent or more of the total estimated leakage across the three stages.

The large uncertainties in leakage estimates arise from the sheer size and diversity of the gas supply system and a lack of sufficient measurements and other data for calculating leak rates.

Gas Production

There are more than half a million gas wells in the U.S., and an average of about 20,000 new wells have been drilled each year over the past several years.

During the production of gas from conventional wells (not hydraulically fractured wells), a significant leakage source is the periodic unloading of liquids that seep into and accumulate in a well over time. A typical gas well undergoes liquids unloading multiple times each year, and the gas that accompanies liquids to the surface when they are unloaded is vented, burned, or diverted to a pipeline. Burning converts methane to CO_2, a less potent greenhouse gas. Estimating the methane vented during liquids unloading requires estimating the number of liquid unloadings that occur each year and the amount of methane vented at each unloading. The EPA made significant revisions in its most recent inventory in estimates of both the number of wells using liquids unloading and the annual emissions from unloadings at such wells. The revisions resulted in a greater than 90 percent reduction in estimated liquids unloading emissions between EPA's 2012 and 2013 estimates. Such a large adjustment raises questions as to the uncertainties in such estimates. Having confidence in emissions estimates at the national level is challenging because of the large variations in liquids unloading requirements across wells, the differing industry practices for handling the gas streams that accompany liquids unloading, and the lack of measurements.

Average methane leakage rates for conventional gas production based on different studies in the literature range from 0.3 to 2.2 percent of gas produced. The large range reflects a lack of agreement among authors due in part to the poor quality and limited amount of publicly available data.

With shale gas, the largest emissions during production occur during well completion, the process of preparing the well for the start of marketed production. This includes drilling, hydraulic fracturing, and flow back of the fracturing fluid to the surface. In some cases, maintaining gas production requires periodic well re-fracturing, called a workover. Whether the gas that accompanies the flowback fluid to the surface is vented, burned, or captured for sale significantly affects the overall leakage rate. How flowback gas is handled at different wells is not well known, which further contributes to uncertainties in average estimates of well completion emissions.

An additional significant source of uncertainty in methane leakage during production is the amount of gas that a well will produce over its lifetime. This estimated ultimate recovery (EUR) is important because the one-time methane emissions that occur during well completion are allocated across the total expected production from the well to estimate the percentage of gas production that leaks. An appropriate average EUR to use in leakage estimates is difficult to know with confidence because few shale wells have yet operated for their full lifetime. Moreover, it is likely that EUR values for wells in different shale basins will vary by an order of magnitude or more, and wells within the same basin are expected to have variations in EUR of 2 or 3 orders-of-magnitude.

Beginning in 2013, all natural gas producers are required to report data to the EPA on their production practices, and these data are expected to help reduce some of the uncertainties around estimated leakage rates during gas production. In addition, beginning in August 2011, EPA regulations required that methane be either burned or captured during completion of hydraulically fractured wells. Starting in 2015, all hydraulically fractured wells will be required to use "green completion" technologies to capture the methane. The EPA estimates that methane leakage is reduced by 95 percent with a green completion compared with venting of the methane.

The average methane leakage rate for gas production from hydraulically fractured shale wells estimated in different studies ranges from 0.6 to 3.0 percent.

Gas Processing

An estimated 60 percent of gas coming out of wells in the U.S. contain CO_2 and other contaminants at unacceptably high levels for market sale, so this gas must first undergo processing. A gas processing plant is a collection of chemical reactors that strip contaminants, along with a series of electric and engine-driven compressors that move gas through the plants. Most of the methane leakage during gas processing is believed to come from compressor seals and from incomplete gas combustion in the engines. A major EPA-sponsored study published in 1996 reported measured leak rates from more than 100 different emission sources in the natural gas supply system. Measurements included compressors and engines at gas processing plants, on the basis of which representative daily leakage rates were determined. These are the basis for most of the EPA's gas processing emission estimates today. Additionally, when required, CO_2 that originated in the natural gas is separated from the gas during processing and vented to the atmosphere. This is not a methane emission, but contributes to the overall upstream greenhouse gas emissions footprint of natural gas.

Average methane leakage from gas processing is 0.1 to 0.3 percent of the methane produced, based on different studies. Because there is a well-documented number of gas processing facilities – one facility will handle gas from many wells – and because emission factors are based on measurements of compressor and engine leak rates (albeit measurements made nearly two decades ago), the level of confidence in estimates of gas processing methane leakage rates is relatively high. Moreover, based on EPA's estimates, gas processing accounts for the least methane leakage among the four stages in the natural gas supply system, so uncertainties in gas processing estimates are of less significance overall than uncertainties around leakage in other stages.

Gas Transmission

There are more than 300,000 miles of natural gas transmission pipelines in the U.S., some 400 storage reservoirs of varying types, more than 1400 pipeline-gas compressor stations, and thousands of inter-connections to bulk gas users (such as power plants) and distribution networks. Essentially all gas passes through the transmission system, and about half is delivered directly from a transmission line to large customers like power plants. Transmission pipelines are relatively well maintained, given the risks that poor maintenance entails. The EPA estimates that most methane emissions associated with transmission are due to leakage at compressors and from engines that drive compressors.

Most studies estimate that average methane leakage in gas transmission ranges from 0.2 to 0.5 percent of production. Because the number of compressors and engines in the transmission system are relatively well documented and because emission factors are based on leakage measurements (albeit made in the mid-1990s), the level of confidence in estimates of gas transmission leakage is relatively high. However, variations in leakage associated with the large seasonal movements of gas in and out of storage reservoirs was not considered when measurements were made, and this introduces some uncertainties.

Gas Distribution

About half of all gas leaving the transmission system passes through a distribution network before it reaches a residential, commercial, or small industrial user. Next to gas production, the uncertainties in methane leakage estimates are most significant for gas distribution. Aside from EPA estimates, there are few systematic studies of leakage in gas distribution. The uncertainties in estimating distribution leakage arise in part because of the large number and varying vintages of distribution mains (an estimated 1.2 million miles of pipes in the U.S.), the large number of service lines connecting distribution lines to users (more than 60 million), and the large number and variety of metering and pressure-regulating stations found at the interface of transmission and distribution systems and elsewhere within the distribution network.

The EPA's leakage estimates are based on measurements made in the 1996 study mentioned earlier, and nearly half of distribution system leakage is estimated to occur at metering/regulating stations. Leakage from distribution and service pipelines accounts for most of the rest. The EPA assumes there is no leakage on the customer side of gas meters, though at least one recent study has suggested this may not be the case.

More recent measurement-based studies help highlight some of the uncertainties with estimating distribution emissions. One study in Sao Paulo, Brazil, measured leakage rates from distribution mains made of cast iron, pipe material that leaks the most. Cast iron was the standard material for U.S. distribution mains in the 1950s, and there are an estimated 35,000 miles of cast-iron pipe still in everyday use in the U.S. The EPA assumes the annual leakage rate for a mile of cast-iron pipe is 78 times that for an equivalent pipe made of steel, a principal replacement pipe for cast iron. The Brazilian study, based on measurements at more than 900 pipe sections, estimated an annual leakage rate per mile at least three times that assumed by the EPA.

There have not been many assessments of total leakage in distribution systems other than that of the EPA, which estimates leakage of 0.3 percent of production. However, several recent studies have measured elevated methane concentrations above the streets of Boston, San Francisco, and Los Angeles. These concentration measurements cannot be converted into estimates of leak rates without additional companion measurements. Follow-up measurements are in progress. Given the poor quality of available data on methane leaks from the distribution system, such measurements will be essential in reducing the uncertainties in distribution leakage estimates.

Natural Gas System Leakage in Total and Implications for Electricity Generation

Electric power generation is the largest gas-consuming activity in the U.S. When considering natural gas electricity generation, leakage from the production, processing, and transmission stages are important to consider, since nearly all power plants receive gas directly from the transmission system. The EPA has estimated methane leakage across the production, processing, and transmission stages of the U.S. natural gas supply system to be 1.2 percent to 2 percent of production, but our review of other assessments finds leakage estimates ranging from less than 1 percent to 2.6 percent for conventional gas and from 1 percent to 4.5 percent for shale gas. When uncertainties in the individual estimates are included, the range extends to 3.8 percent for conventional gas and 7 percent for shale gas. Our review finds that additional leakage measurements are needed to better understand actual leakage rates.

Absent more certainty about methane leak rates, we can assess global warming impacts of different leak rates to identify important threshold leakage levels. For illustration, we consider gas-fired electricity generation, which has been increasing rapidly in recent years primarily at the expense of coal-fired generation. In 2012, 30 percent of all electricity was generated from gas. Many authors have suggested that displacing existing coal-fired generation with natural gas electricity provides a 50 percent reduction in global warming impact because of the lower carbon content of gas and the higher efficiency with which it can be used to generate electricity. But the claim of a 50 percent reduction ignores the global warming impact of methane leaks and the related fact that the potency of methane as a greenhouse gas is far higher than that of CO_2. On a pound-for-pound basis methane has a global warming potential about 100 times that of CO_2 initially, although over 20- or 100-year timeframes, this reduces to 72 or 25 times.

Taking into consideration the time-dependent global warming potential of methane relative to CO_2, we estimated the potential greenhouse benefits from replacing the electricity generated by a single coal power plant with electricity from natural gas instead. Our analysis indicates that if total methane leakage from the gas supply system were 4 percent of production, this substitution of gas-fired electricity for coal-fired electricity would result in only about a 25 percent climate benefit over the next decade, a 35 percent benefit over a 50-year horizon, and a 41 percent benefit over a century (i.e., less than the often cited 50 percent reduction). At higher methane leak rates, the benefits would be lower over the same time horizons. For a switch from coal to gas to provide any positive climate benefit over any time horizon, methane leakage needs to be 6 percent per year or less, and to achieve a 50 percent or better climate benefit over any time horizon leakage needs to be 1.5 percent or less. This analysis applies to a situation in which a coal plant retires and its electricity output is provided instead by a natural gas plant.

At the national level, one must also consider the rate at which coal plants are substituted by gas plants. Here we consider a scenario in which there is a steady substitution of coal electricity by gas-generated power at some average annual rate over time, assuming the total electricity supplied by gas plus coal remains constant. This has roughly been the situation in the U.S. over the past decade, when coal electricity generation decreased at an average rate of 2.4 percent per year, with generation from natural gas making up most of the reduction. (The rate of reduction in coal generation has been accelerating. It averaged 5.5 percent per year over the last 5 years, and 9.4 percent per year over the past 3 years.)

With a coal-to-gas shift, every year there is more gas-fired electricity produced than the previous year, and the methane leakage associated with each new increment of gas electricity has a warming potency that is initially very high and falls with time. When the global warming potential of each new annual pulse of methane is considered, the impact of shifting from coal to gas is less than for the one-time coal-to-gas conversion considered above.

For example, if existing coal electricity were substituted by gas at 5 percent per year, requiring 59 years to reach 95 percent coal replacement, then in 2050 – 37 years from today – the global warming impact (compared to continued coal use) would be lower by 17 or 41 percent, assuming methane leakage of 5 or 2 percent, respectively (Figure 2). If leakage were 8 percent there would be no global warming benefit from switching to gas for at least 50 years.

The 5 percent per year coal substitution rate assumed in the previous paragraph may be difficult to sustain with the gas supply levels the U.S. Department of Energy currently projects will be available over the next three decades. A more realistic coal substitution rate may be 2.5 percent per year, which will require 118 years to reach 95 percent coal replacement. At this rate, the reduction in global warming potential over the next 37 years relative to continued coal use would be only 12 or 29 percent for methane leakage of 5 or 2 percent, respectively (Figure 2). To achieve better than these levels would require other lower-carbon options, such as reduced electricity consumption and/or increased electricity supply from nuclear, wind, solar, or fossil fuel systems with CO_2 capture and storage to provide some of the substitution in lieu of gas.

This analysis considers no change in leakage rate or in the efficiencies of power generation over time. The benefit of a switch from coal to gas would obviously increase if leakage were reduced and/or natural gas power-generating efficiency increased over time.

In summary, the coal-to-gas transition rate, the changing potency of methane over time, and the methane leakage fraction all significantly affect future global warming. Knowing with greater certainty the level of methane leakage from the natural gas supply system would provide a better understanding of the actual global warming benefits being achieved by shifting from coal to gas.

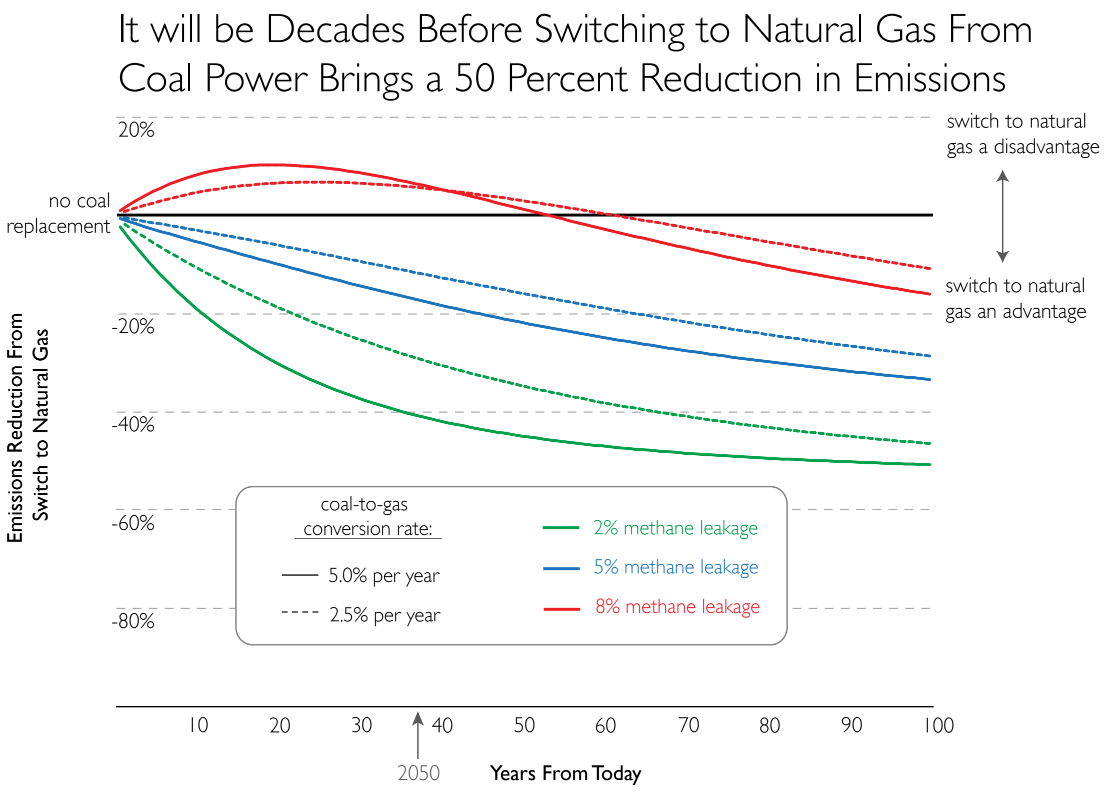


Figure 2. Impact on global warming of shifting existing coal generated electricity to natural gas over time relative to maintaining existing coal generation at current level. The impacts are calculated for two different annual coal-to-gas substitution rates and for three assumed methane leakage rates.

1. Introduction

Natural gas is the second most abundant fossil fuel behind coal, in both the U.S. and the world. At the rate it was used in 2011, the U.S. has an estimated (recoverable) 91-year supply of natural gas. Coal would last 140 years (Table 1). Oil, the most-used fossil fuel in the U.S., would last 36 years.

The estimates of the total amount of natural gas stored under the U.S. increased dramatically in the past decade with the discovery of new forms of unconventional gas, which refers broadly to gas residing in underground formations requiring more than a simple vertical well drilling to extract. Shale, sandstone, carbonate, and coal formations can all trap natural gas, but this gas doesn't flow easily to wells without additional "stimulation".[4] The production of shale gas, the most recently discovered unconventional gas, is growing rapidly as a consequence of new technology and know-how for horizontal drilling and hydraulic fracturing, or fracking.[a] (See Box 1.) An average of more than 2000 new wells per month were drilled from 2005 through 2010 (Figure 3), the majority of which were shale gas wells.

Shale gas accounted for 30 percent of all gas produced in the U.S. in 2011, a share that the U.S. Department of Energy expects will grow significantly in the decades ahead, along with total gas production (Figure 4). Gas prices in the U.S. fell significantly with the growth in shale gas and this has dramatically increased the use of gas for electric-power generation (Figure 5) at the expense of coal-fired power generation. Coal and natural gas provided 37 percent and 30 percent of U.S. electricity in 2012.[6] Only five years earlier, these shares were 49 percent for coal and 22 percent for gas.

Using natural gas in place of coal in electricity generation is widely thought to be an important way to reduce the amount of globe-warming CO_2 emitted into the atmosphere, because combustion of natural gas by itself produces much less CO_2 than the combustion of an energy-equivalent amount of coal (Figure 6, left), and natural gas can be converted much more efficiently into electricity than coal, resulting in an even larger difference between combustion-related emissions per kilowatt-hour of electricity generated (Figure 6, right).

When comparing only combustion emissions, natural gas has a clear greenhouse gas emissions advantage over coal. But emissions are also released during fossil fuel extraction and transportation (these are known as the upstream emissions) and these must also be considered to get an accurate picture of the full greenhouse emissions impact of natural gas compared to coal. The upstream plus combustion emissions when considered together are often called the lifecycle emissions.

*Table 1. Number of years that estimated recoverable resources of natural gas, petroleum, and coal would last if each are used at the rate that they were consumed in 2011.**

	Years left at 2011 rate of use	
	WORLD*	**U.S.****
Conventional Natural Gas	116	42
Unconventional Natural Gas	1021	49
Petroleum	171	36
Coal	2475	140

** Calculated as the average of estimated reserves plus resources from Rogner, et al[1], divided by total global use of gas, petroleum, or coal in 2011 from BP.[2] The consumption rates in 2011 were 122 exajoules for gas, 170 exajoules for oil, and 156 exajoules for coal. One exajoule is 10^{18} joules, or approximately 1 quadrillion BTU (one quad).*
*** Including Alaska. Calculated from resource estimates and consumption data of EIA.[3]*

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[a] Horizontal drilling and hydraulic fracturing are also applied to produce gas from some tight sandstone and tight carbonate formations. A key distinction between the term tight gas and shale gas is that the latter is gas that formed and is stored in the shale formation, whereas the former formed external to the formation and migrated into it over time (millions of years).[4]

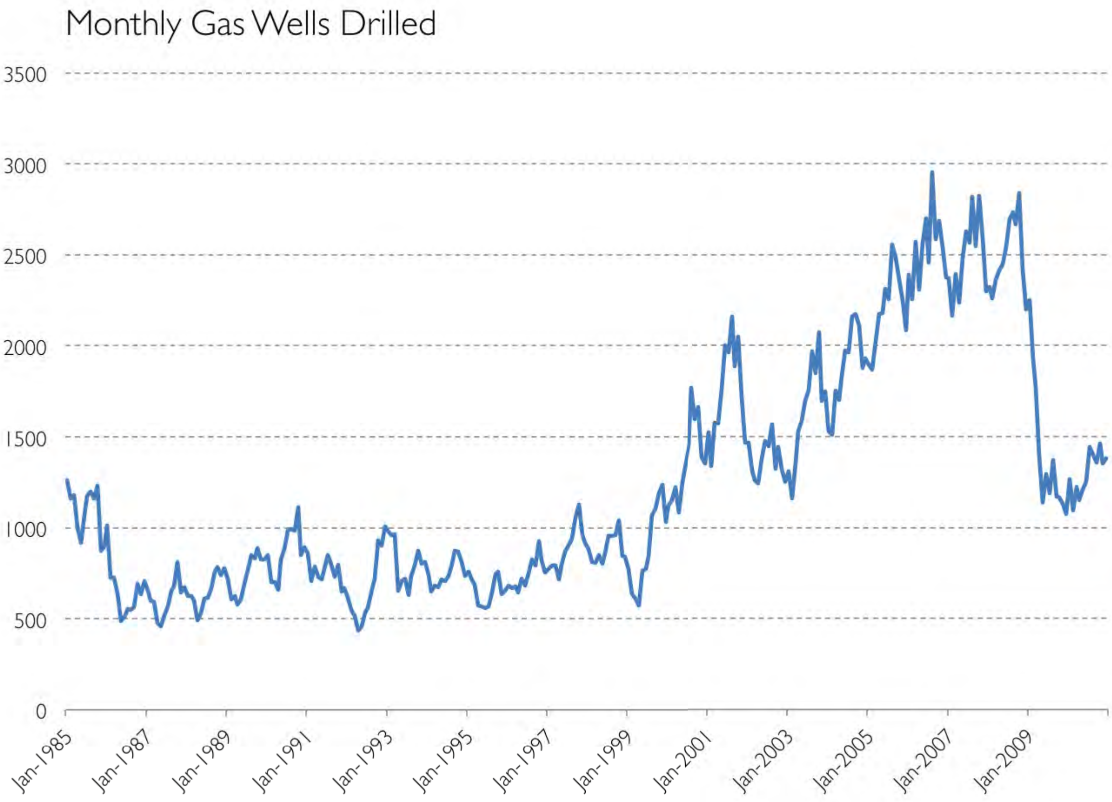


Figure 3. *Number of gas wells drilled per month in the U.S.*[5]

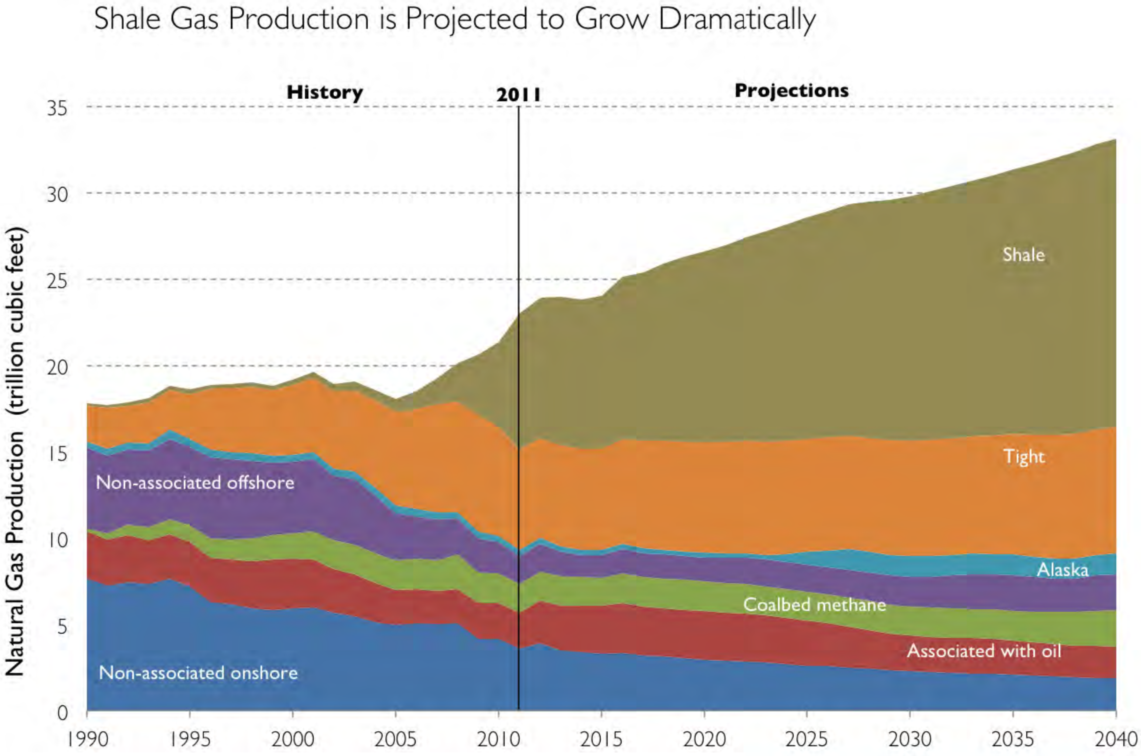


Figure 4. *Past and projected U.S. natural gas production (in trillion cubic feet per year). A trillion cubic feet of natural gas contains about one quadrillion BTU (quad), or equivalently about 1 exajoule (EJ) of energy. Source: EIA.*[7]

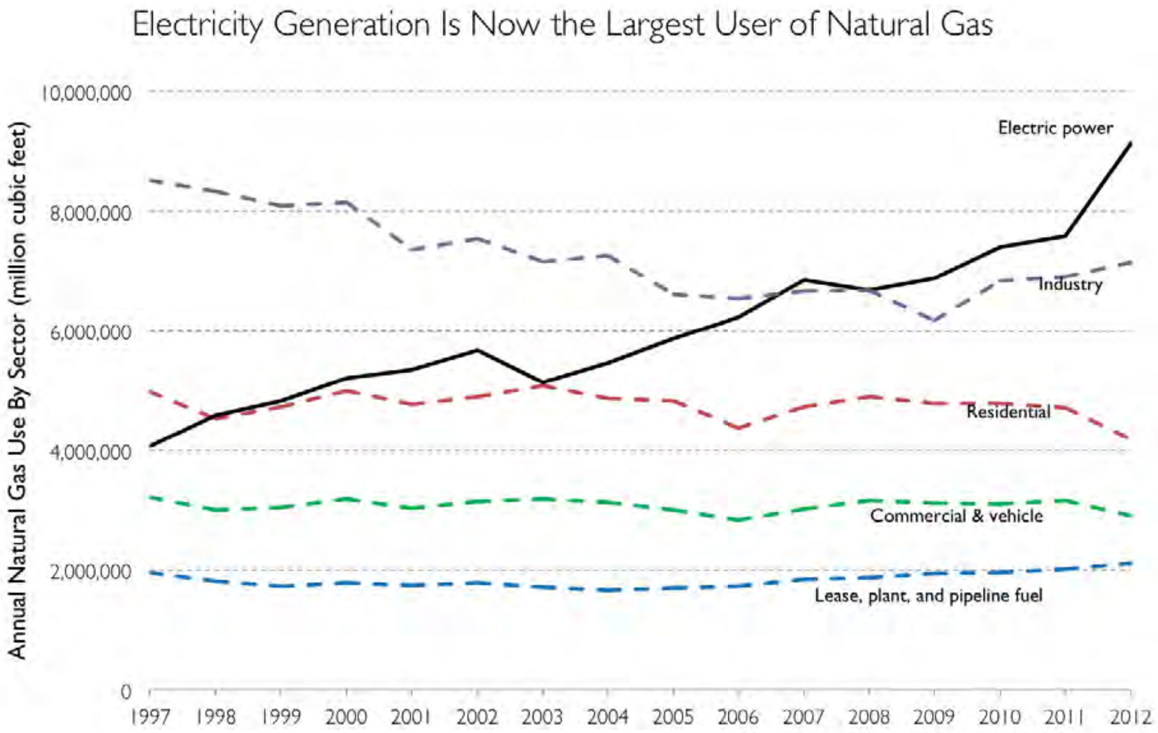

Electricity Generation Is Now the Largest User of Natural Gas

Figure 5. Unlike other sectors, natural gas for electricity generation has been growing since around 1990 and is now the single largest user of natural gas. This graph shows gas use (in million cubic feet per year) by different sectors. Lease, plant, and pipeline fuel refers to natural gas consumed by equipment used to produce and deliver gas to users, such as natural gas engines that drive pipeline compressors. Source: EIA.

Burning Natural Gas Produces Much Less CO_2 Than Burning Coal

Figure 6. Average emissions by fuel type from combustion of fossil fuels in the U.S. in 2011:[7] average emissions per million BTU (higher heating value) of fuel consumed (left) and average emissions per kWh of electricity generated (right).

The recent and dramatic appearance of shale gas on the energy scene has raised questions about whether or not lifecycle greenhouse gas emissions for natural gas are as favorable as suggested by the simple comparison of combustion emissions alone. The main constituent of natural gas, methane (CH_4), is a much more powerful greenhouse gas than CO_2, so small leaks from the natural gas system can have outsized impacts on the overall lifecycle carbon footprint of natural gas. (See Box 2.)

In this report, we review what is known about methane leakage and other greenhouse gas emissions in the full lifecycle of natural gas, including shale gas. The natural gas supply system includes production of raw gas, processing of the raw gas to make it suitable for pipeline transport, transmission of gas in bulk by pipeline (often over long distances), and finally local distribution of the gas to users (Figure 7). The infrastructure is vast, with literally thousands of places where leaks of methane could occur. As of 2011, the U.S. natural gas system included more than half a million producing wells, several hundred gas processing facilities (Figure 8), hundreds of thousands of miles of gas transmission pipelines (Figure 9) and integrated storage reservoirs (Figure 10), more than a million miles of local distribution mains, and more than 60 million service pipe connections from distribution mains to users. The system delivered on average about 70 billion cubic feet of gas each day to users nationwide in 2012.

We discuss GHG emission estimates of the natural gas system made by the U.S. Environmental Production Agency (EPA), which annually produces official and detailed estimates of all U.S. greenhouse gas emissions. We then review other, non-EPA estimates, compare these with EPA's numbers, and highlight where the most significant uncertainties lie. We finish with an analysis that puts in perspective the significance of different methane leak rates for the global warming impact of natural gas substituting coal in electricity generation.

Each Stage in the Natural Gas Supply System is a Vast Infrastructure

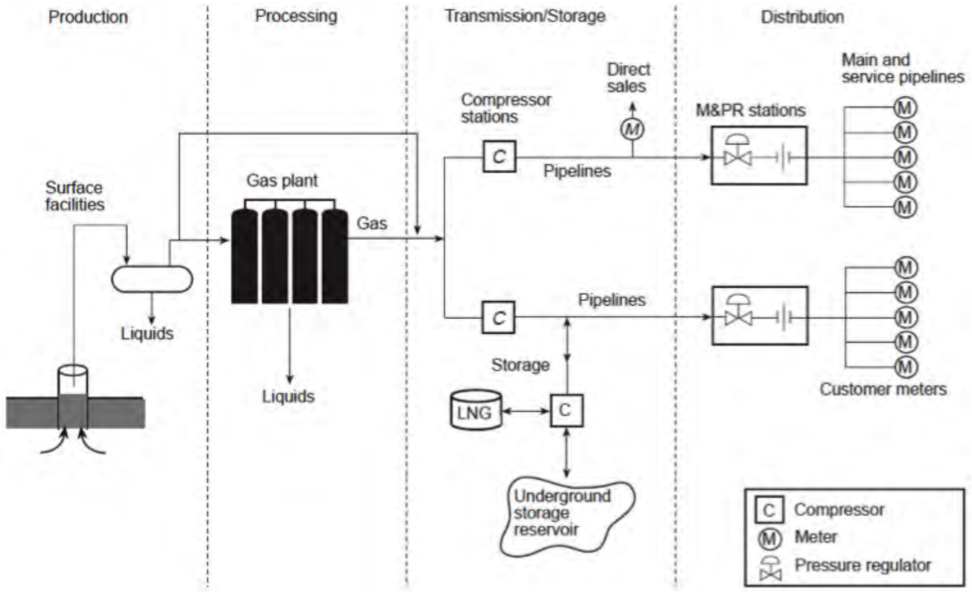


Figure 7. The U.S. natural gas supply system.[8]

There are Hundreds of Natural Gas Processing Plants in the Country

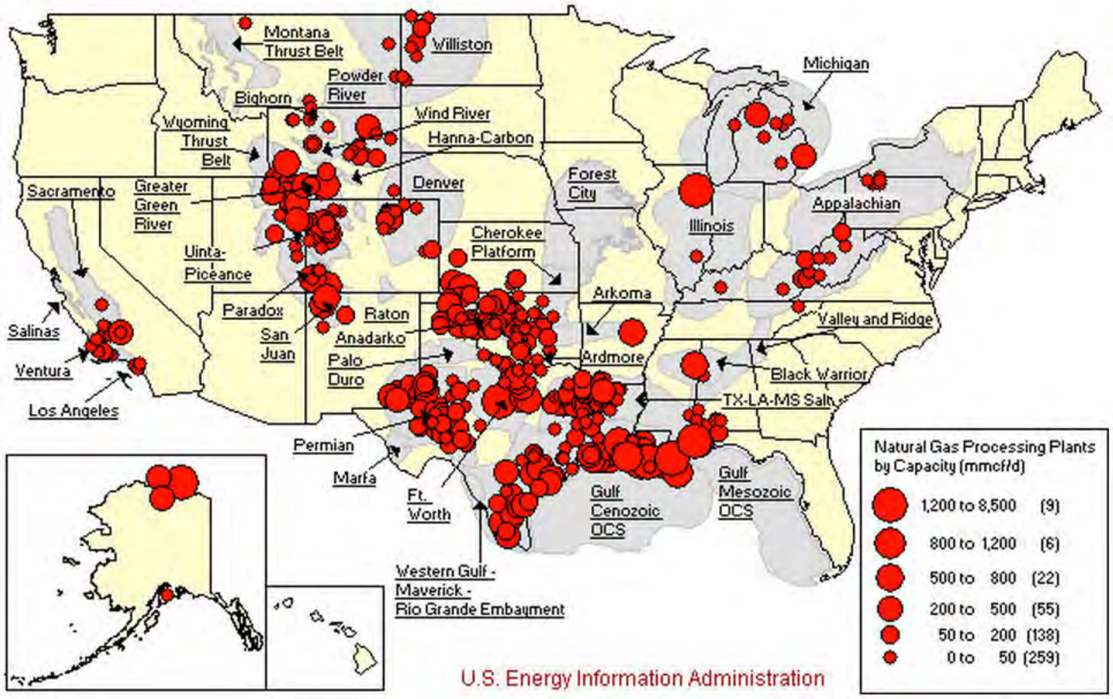


Figure 8. *U.S. natural gas processing plants.* [9]

Hundreds of Thousands of Miles of Gas Transmission Pipelines Cover the U.S.

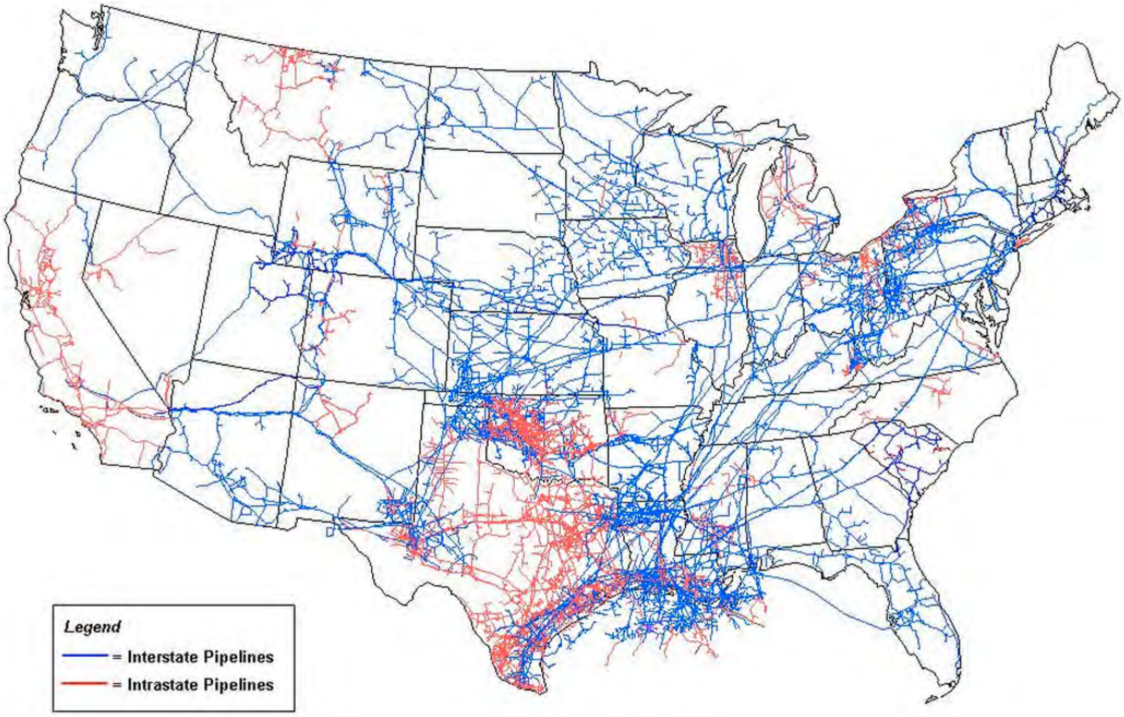
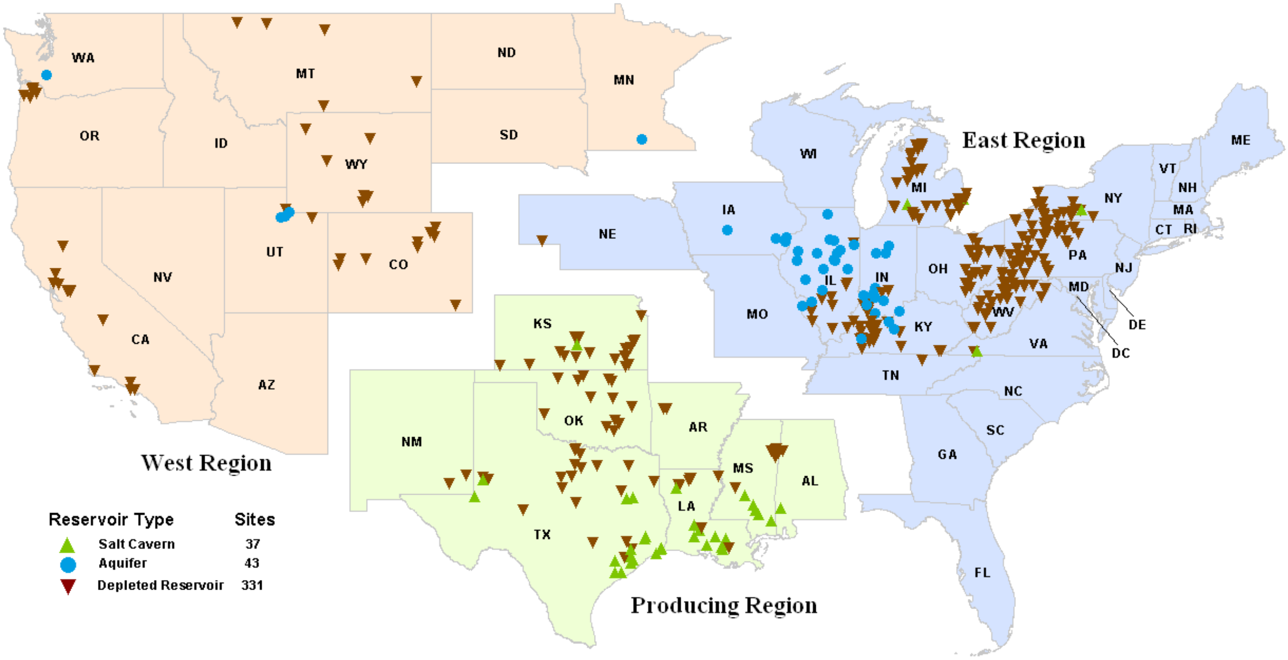


Figure 9. *The U.S. natural gas transmission system (as of 2009).* [10]

Natural Gas Storage Facilities Exist Across the Country



U.S. Lower 48 Underground Natural Gas Storage Facilities, by Type (December 31, 2010)

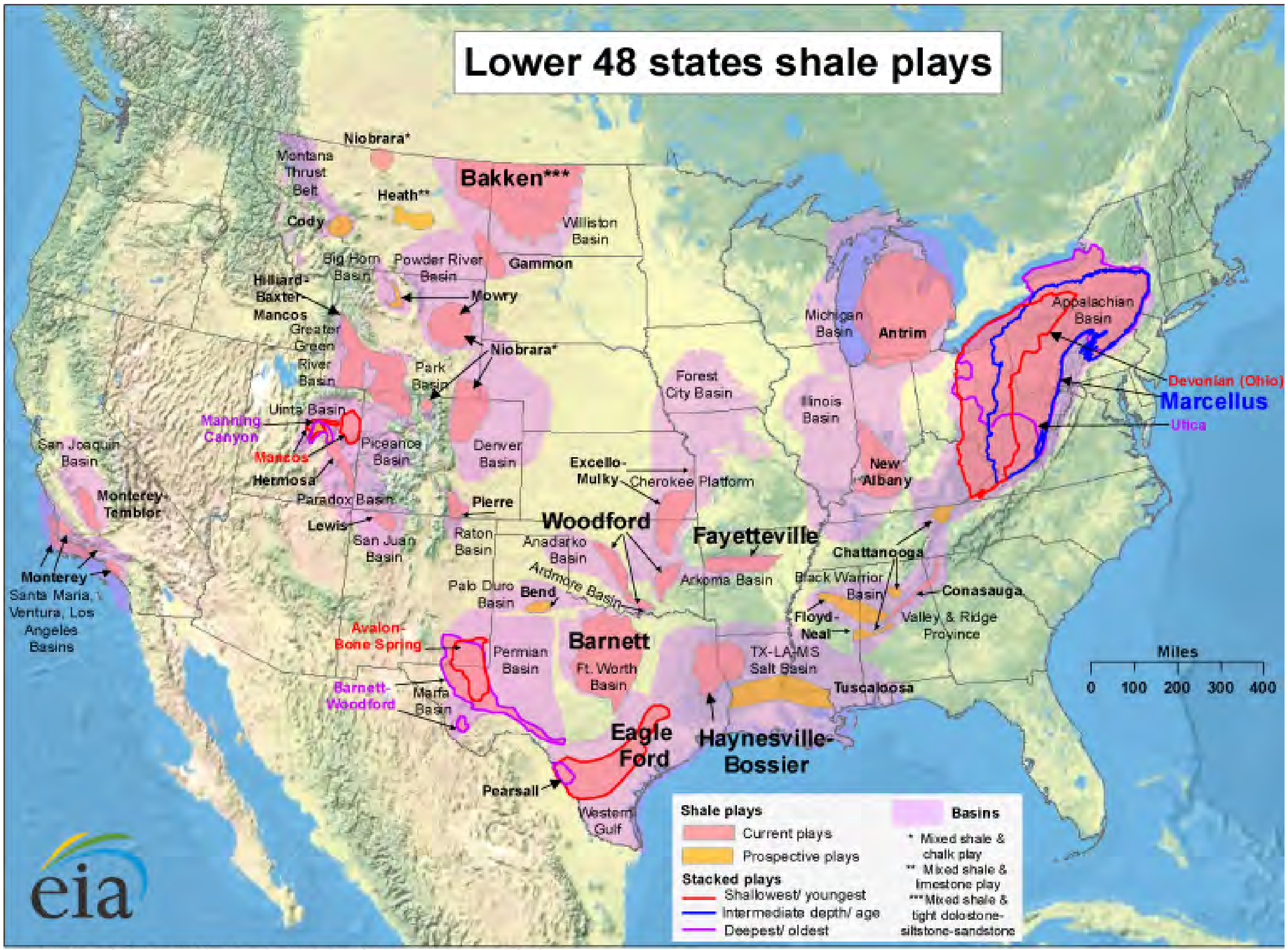
Note: Locations of storage facilities presented in the map are approximate. Some symbols representing storage facilities may overlap.
Source: U.S. Energy Information Administration, Form EIA-191A, "Annual Underground Gas Storage Report"

Figure 10. U.S. natural gas storage facilities.[11]

Box 1: Shale Gas

There are numerous gas-containing shale formations across the lower-48 states (Figure 11) and Alaska, with the largest shale gas reserves estimated to be in the Texas/Gulf Coast and Appalachian regions (Table 2). Alaska's resources are also large, but there are limited means in place today to transport this gas to users elsewhere. Shale gas production in the U.S. quadrupled between 2007 and 2011, with average annual growth of 44 percent. Seven states – Texas, Louisiana, Pennsylvania, Oklahoma, Arkansas, West Virginia and Colorado – accounted for about 90 percent of all shale gas production in 2011 (Figure 12).

Shale gas is formed by decomposition over millennia of organic (carbon-containing) plant and animal matter trapped in geologic sediment layers. Most shale formations are relatively thin and occur thousands of feet below the surface. Marcellus shales are typical, with thicknesses of 50 to 200 feet and occurring at depths of 4,000 to 8,500 feet.[4] The Antrium and New Albany formations (see Figure 11) are unusual in being thinner and shallower than most other U.S. shale deposits. Antrium and New Albany are also differentiated by the presence of water. This leads to the co-production of some water with shale gas from these formations, a complication not present for most wells in other shale formations (but a common occurrence for conventional (non-shale) gas wells – see discussion in Section 2.1 of liquids unloading).



Source: Energy Information Administration based on data from various published studies.
Updated: May 9, 2011

Figure 11. Shale gas formations in the lower-48 states. [12]

Table 2. Mean estimate by the U.S. Geological Survey of undiscovered technically recoverable shale gas resources by basin. [13]

	Trillion cubic feet*
Gulf Coast	**124.896**
Haynesville Sabine	60.734
Eagle Ford	50.219
Maverick Basin Pearsall	8.817
Mid-Bossier Sabine	5.126
Appalachian Basin	**88.146**
Interior Marcelllus	81.374
Northwestern Ohio	2.654
Western Margin Marcellus	2.059
Devonian	1.294
Foldbelt Marcellus	0.765
Alaska North Slope	**40.589**
Shublick	38.405
Brookian	2.184
Permian Basin	**35.130**
Delaware-Pecos Basins Barnett	17.203
Delaware-Pecos Basins Woodford	15.105
Midland Basin Woodword-Barnett	2.822
Arkoma Basin	**26.670**
Woodford	10.678
Fayetteville-High Gamma Ray Depocenter	9.070
Fayetteville Western Arkansas	4.170
Chattanooga	1.617
Caney	1.135
Bend Arch-Forth Worth Basin	**26.229**
Greater Newark East Frac-Barrier	14.659
Extended Continuous Barnett	11.570
Andarko Basin	**22.823**
Woodford	15.973
Thirteen Finger Limestone-Atoka	6.850
Paradox Basin	**11.020**
Gothic, Chimney Rock, Hovenweep	6.490
Cane Creek	4.530
Michigan Basin (Devonian Antrim)	**7.475**
Illinois Basin (Devonian-Mississippian New Albany)	**3.792**
Denver Basin (Niobrara Chalk)	**0.984**
Total	376.734

** One trillion cubic feet of gas contains about one quadrillion BTU (one quad).*

Seven States Accounted for 90 Percent of Shale Gas Production in 2011

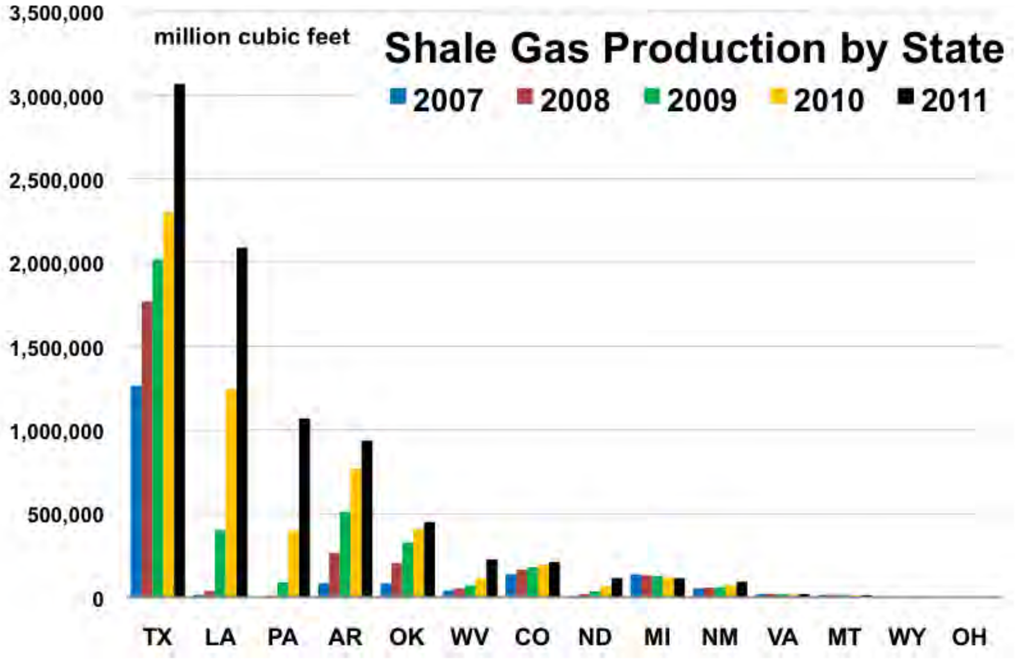


Figure 12. *Shale gas production in the U.S. has grown rapidly.* [14]

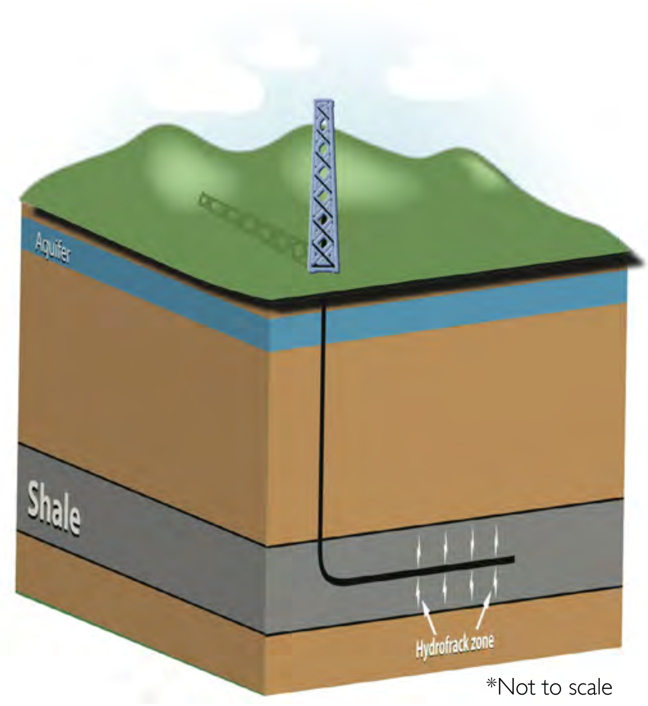


Figure 13. *Hydraulic fracturing combined with horizontal drilling allows accessing more of a thin shale formation.*

The existence of shale gas has been known for decades, but only with the development of hydraulic fracturing and horizontal drilling techniques in the mid-1990s did it become economically viable to produce. Hydraulic fracturing involves injecting a "fracking fluid" (water plus a "proppant" – typically sand – and small amounts of chemicals) at sufficiently high pressure into a well bore to crack the surrounding rock, creating fissures that can extend several hundred feet from the well bore. As the fluid flows back to the surface before the start of gas production, the proppant stays behind and keeps the fissures propped open allowing gas to escape to travel to the well bore.

"Fracking" was originally developed for use in vertically drilled wells, but shale gas production only began in earnest with the development of horizontal drilling, which when combined with fracking, enables access to much more of the volume of the thin, but laterally expansive shale formations (Figure 13). State-of-the art shale gas wells have horizontal holes extending 3000 feet or more from the vertical hole. Additionally, multiple horizontal holes are typically drilled from a single well pad, reducing overall drilling costs and enabling access to much more of a shale formation from a small area on the surface.

Box 2: The Global Warming Potential of Methane

Some molecules in the atmosphere allow solar energy to pass through to the earth's surface, but absorb energy radiated back from the earth and re-radiate that energy back to the surface, thereby making the earth's surface warmer than it would be without these "greenhouse gases" in the atmosphere. Two of the most important global warming molecules are carbon dioxide (CO_2) and methane (CH_4). Each has different global warming behavior and the term "Global Warming Potential" (GWP) is used to characterize their warming power. For convenience, the GWP of one pound (or kilogram) of CO_2 is defined to be equal to one, and GWP's of other gases are defined relative to the warming effect of CO_2.

The GWP of methane is determined by three factors: the warming properties of the methane molecule itself ("direct radiative forcing"), the warming resulting from interactions between methane and other molecules in the atmosphere ("indirect forcing"), and the effective lifetime of methane in the atmosphere. Considering the first two factors, the warming impact of one kilogram of methane is 102 times that of one kilogram of CO_2, according to the Intergovernmental Panel on Climate Change (IPCC). The third factor is relevant because the carbon in a molecule of methane emitted into the atmosphere will eventually react with oxygen and be converted to CO_2. The characteristic lifetime for methane molecules in the atmosphere is 12 years.[15] The lifetime for a CO_2 molecule in the atmosphere is far longer than this.

Because of the different lifetimes of CH_4 and CO_2, the GWP of CH_4 depends on the time period over which the impact is assessed. The longer the time after being emitted, the lower the GWP (Figure 14).

Thus, the timeframe used for any particular analysis is important. A shorter timeframe may be appropriate for evaluating GWP if the focus is on short-term warming effects or if the speed of potential climate change is of more interest than the eventual magnitude of change in the longer term. A longer horizon would be more appropriate when the interest is in changes that will be expressed more in the longer term, such as significant increase in sea level.

GWP values for methane that are considered the consensus of the climate science community are those published in the Assessment Reports of the Intergovernmental Panel on Climate Change (IPCC), Table 3. As understanding of the science of global warming has improved, the estimate of methane's GWP has increased. For example, the IPCC's Second Assessment Report and Third Assessment Report gave a 100-year GWP of 21 for methane, compared with 25 in the Fourth Assessment Report. More recent analysis has suggested that the GWP may be higher still,[16] but pending publication of the IPCC's Fifth Assessment Report (expected in 2013/2014), the scientific consensus GWP values are those in Table 3. Most analysts use the 100-year GWP to convert methane emissions into equivalent CO_2 emissions, since this is the time frame within which significant climate changes are expected to materialize, given current trends in emissions. Some analyses use a 20-year GWP, arguing that short-term effects are significant and demand significant near-term action to reduce emissions.[17] Alvarez et al.[18] suggest that varying time frames for assessing GWP may be useful. The utility of this approach is illustrated in Section 4 of this report.

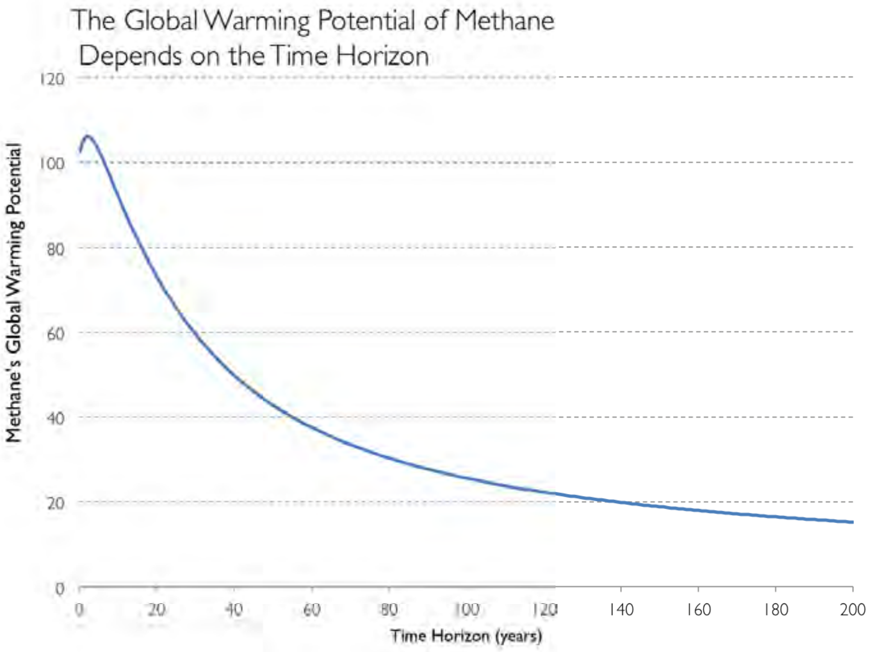


Figure 14. *The global warming potential (GWP) of methane relative to* CO_2 *for a pulse emission at time zero. This assumes a characteristic lifetime in the atmosphere of 12 years for methane and a lifetime for* CO_2 *as predicted by the Bern carbon cycle model.*[15] *(See Alvarez* et al.[18]*)*

Table 3. *The global warming potential for methane falls as the time horizon for its evaluation grows.*[15] *A 20-year GWP of 72 for methane means that 1 kilogram of methane gas in the atmosphere will cause the equivalent warming of 72 kilograms of* CO_2 *over a 20 year period. The GWP values here are consistent with those shown in Figure 14.*

	20-year GWP	100-year GWP	500-year GWP
GWP of CH_4 (methane)	72	25	7.6

2. EPA Estimates of GHG Emissions from the Natural Gas Supply System

Official estimates of U.S. greenhouse gas emissions since 1990 are published each year by the Environmental Protection Agency in its so-called Emissions Inventory[19]. The EPA recently released its 2013 inventory[20], reflecting estimates through 2011. Our discussion here also includes detail drawn from the 2012 inventory[21], reflecting estimates through 2010. We note key changes in methodology and results between the 2012 and 2013 inventories.

The EPA's estimate of total U.S. greenhouse gas (GHG) emissions in the 2012 inventory are shown in Figure 15 in terra-grams (Tg, or millions of metric tons) of CO_2 equivalent per year.[b] Nearly 80 percent of emissions are as CO_2 released from burning fossil fuels.

Methane leakage from the natural gas supply system also contributes[c]. In the 2012 inventory, EPA estimated that 10 percent of all GHG emissions in 2010 (in CO_2-equivalent terms) was methane, with leaks in the natural gas supply system accounting for one third of this, or 215 million metric tons of CO_2-equivalent (Figure 16). These methane emissions from the natural gas supply system correspond to 2.2 percent of methane extracted from the ground (as natural gas) in the U.S. in 2010[d]. The EPA adjusted this estimate significantly downward (to 144 million metric tons of CO_2-equivalent in 2010) in its 2013 inventory, corresponding to an estimated methane leakage rate in 2010 of 1.5 percent. This large adjustment from one EPA inventory to the next hints at the uncertainties involved in estimating the national methane leakage rate.

The EPA develops its emission estimates using a wide variety of data sources and by applying a multitude of assumptions. (See Box 3). EPA's estimated methane emissions in 2010 from the natural gas system are summarized in Table 4, as reported in the 2012 and 2013 inventories.

Methane was an Estimated 10 Percent of U.S. Greenhouse Gas Emissions in 2010

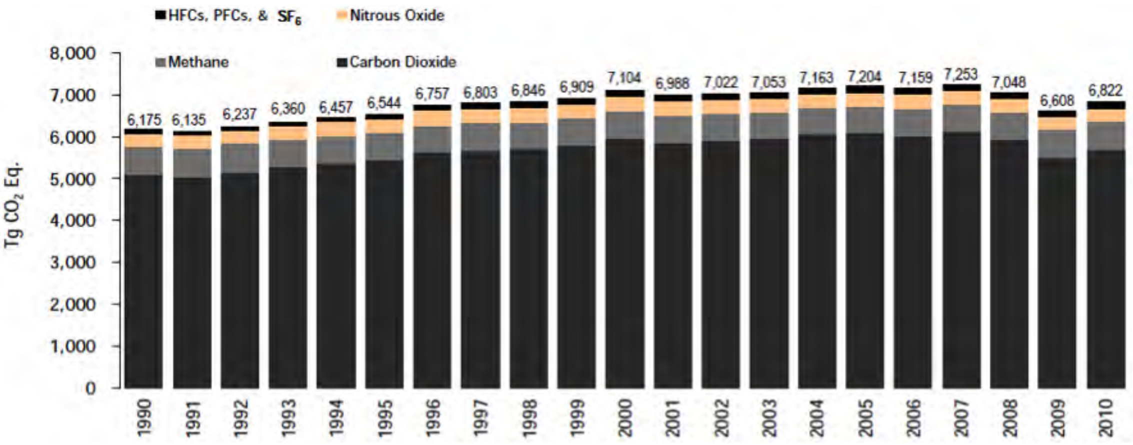


Figure 15. *U.S. greenhouse gas emissions as estimated by the Environmental Protection Agency.*[21]

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[b] The EPA inventories use 100-year global warming potentials (GWPs) for non-CO_2 gases taken from the *Third Assessment Report* (1996) of the Intergovernmental Panel on Climate Change (IPCC), not from the most recent (2007) IPCC Assessment. The methane GWP value used by EPA in this inventory is 21. See Box 2 for discussion of GWP.

[c] Some naturally-occurring underground CO_2 is also vented to the atmosphere in the course of producing, processing, and transporting natural gas. EPA estimates these are much less one-tenth of one percent of the CO_2-equivalent emissions of methane.[23]

[d] U.S. natural gas consumption in 2010 was 24.1 trillion standard cubic feet according to the U.S. Energy Information Administration. Assuming the methane fraction in this gas was 93.4 percent, the value assumed by EPA in its emissions inventory,[23] and taking into account the fact that one standard cubic foot (scf) of methane contains 20.23 grams (or 20.23 metric tons per million scf), the total methane consumed (as natural gas) was 455 million metric tons. Considering a GWP of 21 for methane (as the EPA does), this is 9,556 million metric tons of CO_2-equivalent. The ratio of 215 (Table 4) to 9,556 gives a leakage estimate of 2.25 percent of methane consumed. The leakage as a fraction of methane extracted from the ground is $L = 1 - \frac{1}{(1+x)}$ where x is the leakage expressed as a fraction of methane consumption. For $x = 0.0225$, or $L = 0.0220$, or 2.2%.

Leaks in the Natural Gas System are Estimated to be One Third of Methane Emissions

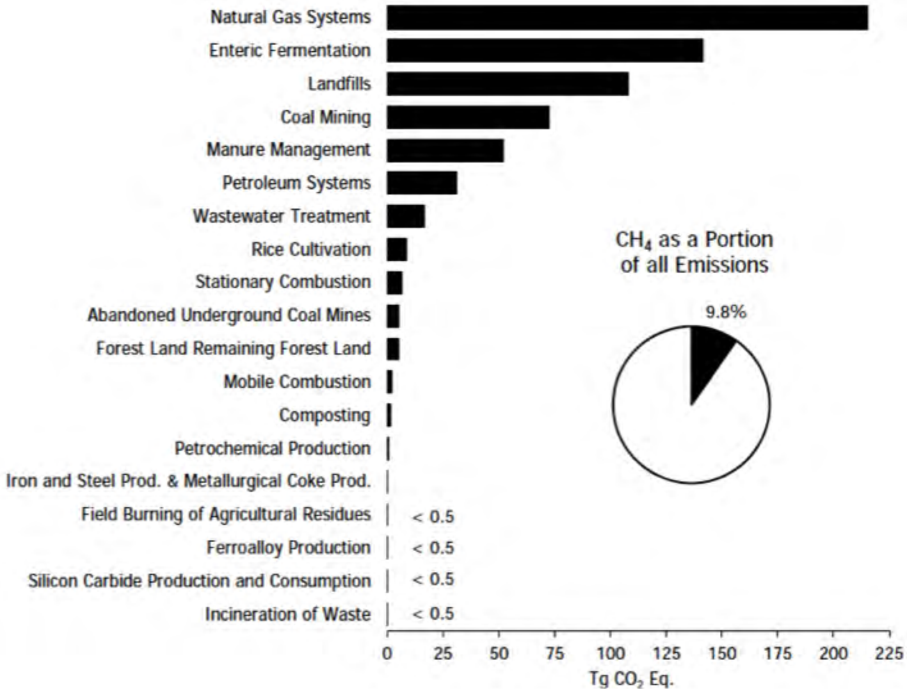

Figure 16. *U.S. methane emissions in 2010 (in million metric tons of CO_2 equivalents) as estimated by the Environmental Protection Agency.*[21]

Table 4. *EPA estimates of methane emissions in 2010 from the natural gas system in units of million metric tons of CO_2-equivalent (for a methane GWP of 21). Figures are from the 2012*[22] *inventory and the 2013 inventory.*[20]

	2012 Inventory	**2013 Inventory**
	million metric tons of CO_2-equivalent	
Natural Gas Production	**126.0**	**57.2**
Liquids unloading	85.7	5.4
Pneumatic device vents	12.8	
Gas engines	5.6	
Shallow water gas platforms	5.6	
Completions and workovers with hydraulic fracturing	3.8	16.7
Other production sources	12.5	
Natural Gas Processing	**17.1**	**16.5**
Reciprocating compressors	8.3	
Centrifugal compressors (wet seals)	4.9	
Gas engines	3.5	
Other processing sources	0.3	
Natural Gas Transmission and Storage	**43.8**	**41.6**
Centrifugal compressors (wet seals) (transmission)	15.7	
Reciprocating compressors (transmission)	12.8	
Engines (transmission)	4.7	
Reciprocating compressors (storage)	3.7	
Liquefied natural gas (LNG) systems	1.9	
Other transmission and storage sources	5.0	
Natural Gas Distribution	**28.5**	**28.3**
Meter/regulator (at city gates)	12.5	
Leaks from main distribution pipelines	9.3	
Leaks from service pipelines connected mains and users' meters	4.3	
Other distribution sources	2.4	
Total Natural Gas System (excluding end-use combustion)	**215.4**	**143.6**

Box 3: EPA's Methodologies for Estimating Methane Leakage from the Natural Gas Supply System

EPA arrives at most of the numbers in Table 4 using a "bottom-up" approach, which refers to estimating the emissions for a piece of equipment or process in the natural gas system as the product of an "emissions factor" and the estimated number of times this activity is repeated across the country each year. This is done for many different activities and the results are added up.[23] As an example, for reciprocating compressors used at gas processing plants (see Table 4), EPA estimated (for the 2012 inventory) that the total number of compressors was 5,028 in 2010 and that on average each compressor had an emission factor (leakage of natural gas to the atmosphere) of 15,205 cubic feet per day. Actual emissions per day will vary from one compressor to another[24], but the objective of the EPA inventory is to estimate emissions at a national level so an average emission factor is adopted. Multiplying the activity level (e.g., number of compressors) by the emission factor, by 365 days per year, and by the assumed methane fraction in the natural gas (which varies by region in the production and processing steps) gives the total annual estimated cubic feet of methane leaked from reciprocating compressors at gas processing plants in 2010. The EPA converts cubic feet per year to grams per year for purposes of reporting in the inventory. (A standard cubic foot of methane contains 20.2 grams.)

Many of EPA's emission factors were developed from a large measurement-based study of the natural gas system done in the mid-1990s.[25] Some of the factors have been updated since then.

For some activities, EPA adjusts its emissions estimates to account for various factors that lead to lower estimated emissions than when using default emission factors. For example, industry partners in EPA's Natural Gas STAR Program[26] use various technologies to lower emissions. In its 2012 inventory, EPA adjusted its national estimate of emissions to account for reductions by the STAR Program partners. As another example, some state regulations require the use of certain technologies to avoid venting of methane in parts of the natural gas system. The EPA adjusts

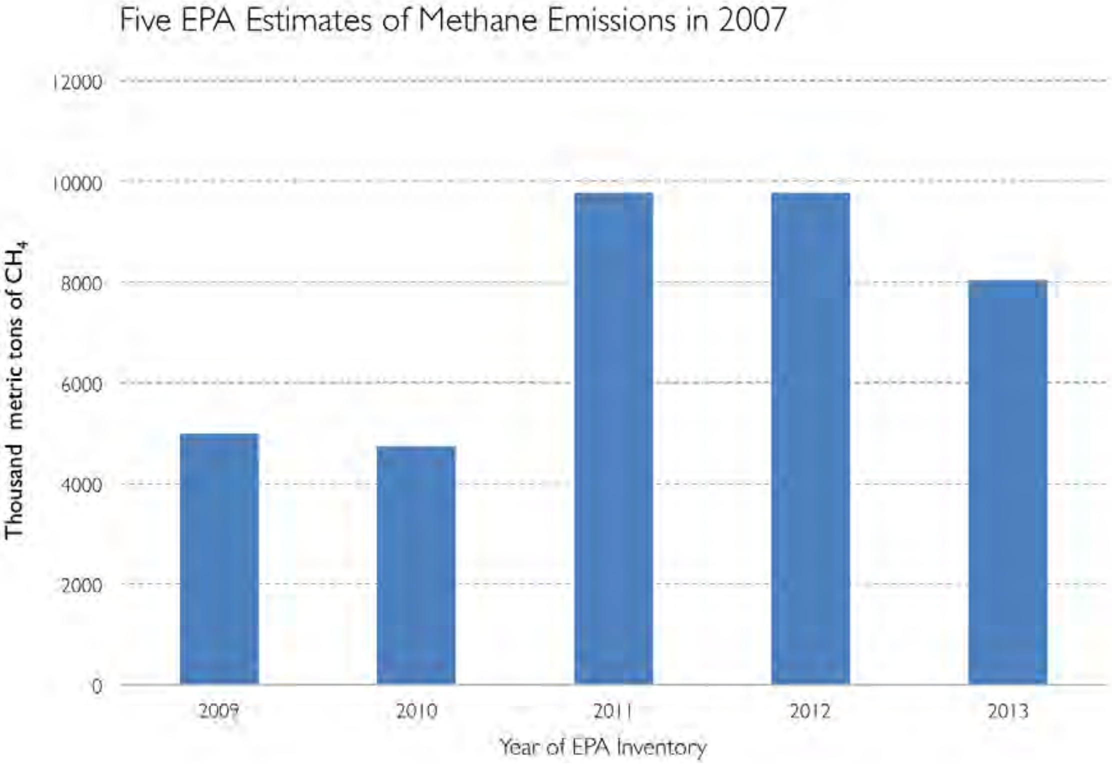

Figure 17. *Methane emissions from the natural gas supply system for 2007, as estimated in five different EPA Emission Inventories. Differences in data sources and methodologies account for the differences in estimated emissions.* [27]

its national estimates to account for the reduced emissions that are assumed to have been achieved in such states. For example, some states require gas wells created by hydraulic fracturing to use technology that eliminates venting of methane during well drilling and fracturing. In its 2012 inventory the EPA cites the example of Wyoming as having such regulations.[23] For its 2012 inventory, EPA estimated that in 2010 approximately 51 percent of all gas wells that were hydraulically fractured in the U.S. were in Wyoming. Accordingly, the 2012 inventory assumes that 51 percent of the estimated total number of hydraulically fractured gas wells in the U.S. had essentially no emissions associated with hydraulic fracturing. The 2013 inventory includes major changes in these assumptions, contributing to a significant increase in estimated emissions associated with hydraulically fractured wells (Table 4).

Completing the emissions inventory involves a massive effort on EPA's part, but is not without uncertainties. To help address these, EPA is continually evaluating and modifying its sources and assumptions in an effort to improve the accuracy of its estimates. When modifications are introduced into the estimation methodology, emissions estimates for all prior years (back to 1990) are revised to maintain a consistent set of estimates over time. These modifications sometimes result in large revisions in prior estimates. This is illustrated in Figure 17, which shows estimates of emissions from the natural gas system for a single year (2007) as made in five successive inventories. In its 2011 inventory, EPA made major adjustments in its data and methodologies from the prior year, resulting in a doubling in the estimate of methane emissions. No changes were made in the methodology for the inventory published in 2012, but changes in the 2013 inventory then resulted in a drop in emissions of nearly 20 percent.

2.1 Gas Production

Among the four stages that constitute the natural gas supply system (Figure 7), the production phase contributes the largest fraction of emissions in EPA's inventory (Table 4). It is also the stage for which the largest changes were made from the 2012 inventory to the 2013 inventory. Within the production phase, "liquids unloading" was the largest contributor in the 2012 inventory, but shrank by more than 90 percent in the 2013 inventory (Table 4). The category "completions and workovers with hydraulic fracturing" was the smallest contributor to production emissions in the 2012 inventory, but was more than quadrupled into the largest contributor in the 2013 inventory.

Liquids unloading refers to the removal of fluids (largely water) that accumulate in the well bore over time at a gas producing well. The fluids must be removed to maintain gas flow, and during this process, methane entrained with the fluids can be released to the atmosphere. Conventional gas wells tend to require more liquids unloading than shale gas wells due to differences in underground geology. From the 2012 to 2013 inventory EPA adjusted many of the assumptions used to estimate liquids unloading, including both the number of wells that use liquids unloading and the amount of methane emitted per unloading. Important considerations in the latter include the number of times each year that the average well is unloaded, the average volume of gas that is entrained with the liquids upon unloading (which varies by region), and the extent to which the entrained gas is captured for flaring (burning)[e] or for sale.[28]

A shale gas operation in Greene County, PA. (Nov 2010). Credit: Mark Schmerling via FracTracker.org.

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[e] One pound of methane vented to the atmosphere has a GWP of 25, considering a 100-yr time horizon (see Box 2). If instead the 1 lb of methane were burned, 2.75 lbs of CO_2 would be produced. This amount of CO_2 has a GWP of 2.75. In this comparison, flaring methane instead of venting it reduces the global warming impact of the emission by a factor of 9.

Well completion refers to the process of finishing the creating of a shale gas well (including hydraulic fracturing) such that it can begin producing saleable gas. A workover is the re-fracturing of a shale gas well to maintain its productivity at an acceptable level. Different wells require different numbers of workovers during their producing life, with some wells not requiring any workovers. With hydraulic fracturing, before gas can flow freely to the surface, there is a fracking fluid flowback period (typically lasting several days) during which a substantial portion of the injected fluid returns to the surface, bringing some amount of gas with it. During the flowback period, if gas that surfaces with the returning fluid is not captured (for flaring or for sale) methane is released to the atmosphere. In the 2013 inventory, well completion and workover emissions more than quadrupled from the 2012 inventory primarily because of an increase in the estimate of the number of wells that were hydraulically fractured and a decrease in the assumed percentage of wells using "green completions" – technology that is employed at some wells to eliminate most well-completion emissions.

2.2 Gas Processing

About 60 percent of all natural gas withdrawn from the ground in the U.S. each year undergoes processing[f] to make it suitable for entry into the gas transmission system.[29] Processing is estimated to account for the smallest contribution to methane emissions among the four stages of the natural gas system (Table 4). Some 97 percent of methane emissions estimated to occur during gas processing are the result of leaks from compressors and gas-fired engines. (Gas-fired engines are used to drive reciprocating compressors. Incomplete combustion of gas in engines results in methane emissions.) The EPA estimates emissions based on the number of compressors and engines in use and an emissions factor (scf methane per day) for each. The 1990s EPA-sponsored study mentioned earlier[25] determined the emission factors and the number of compressors and engines operating in 1992. EPA's inventories for subsequent years use the same emission factors, and the number of compressors and engines is estimated by scaling the 1992 counts of these by the ratio of gas produced in the inventory year to the gas produced in 1992.

Natural gas processing plant

2.3 Gas Transmission and Storage

The natural gas pipeline transmission system in the U.S. includes more than 305,000 miles of pipe, some 400 storage reservoirs, over 1400 compressor stations (Figure 18) each usually with multiple compressors, and thousands of inter-connections to bulk gas users (such as power plants) and to distribution pipeline

Natural gas transmisison lines

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[f] Processing typically removes "condensates" (water and hydrocarbon liquids), "acid gases" (H_2S, CO_2, and others), and sometimes nitrogen. On average the volume of gas after processing is 7 percent or 8 percent less than before processing.

Compression Stations Exist Throughout the Natural Gas Transmission System

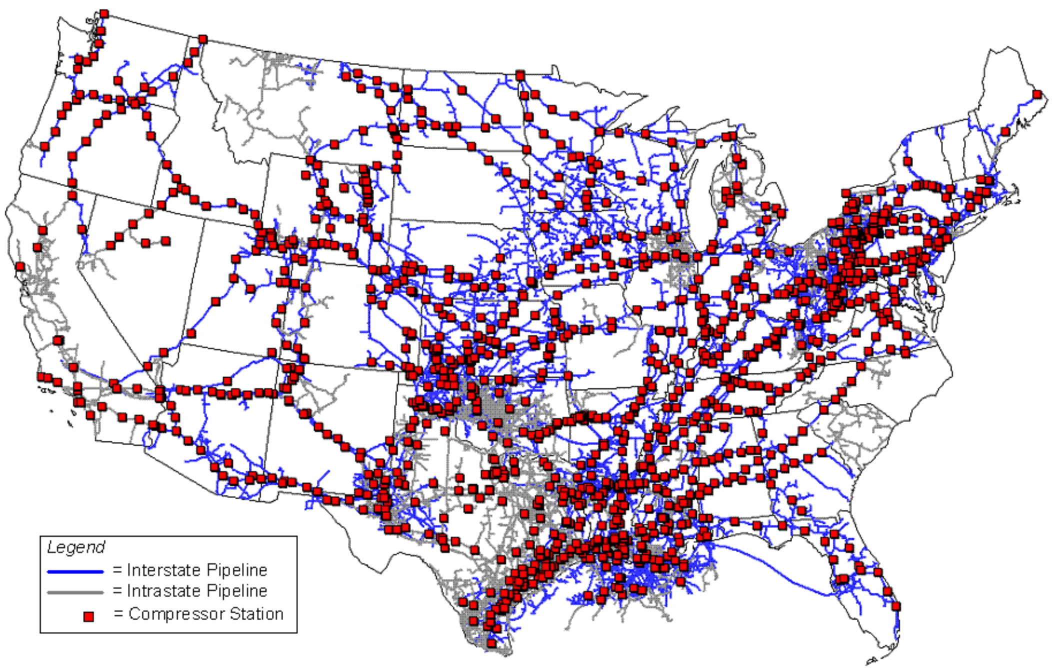
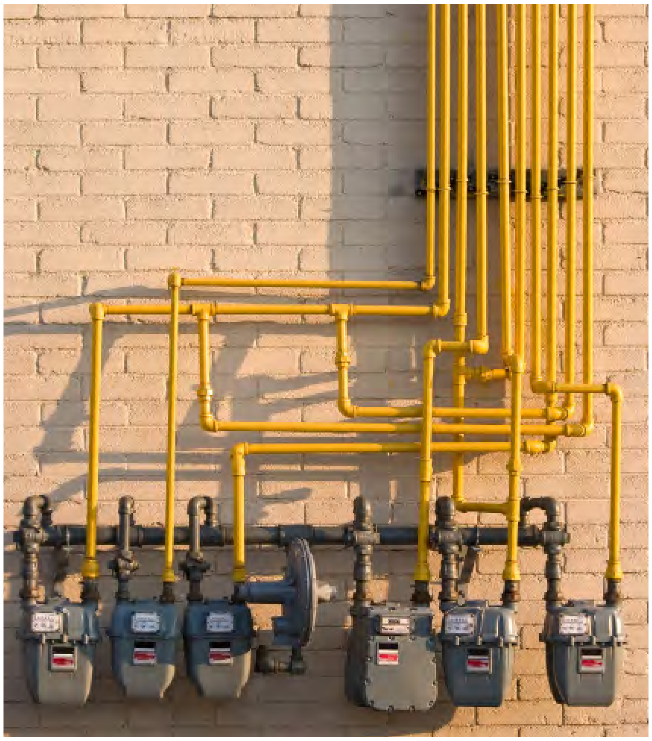


Figure 18. *There are more than 1400 compressor stations in the U.S. natural gas transmission pipeline system.* [30]

systems. The EPA estimates that most emissions from the transmission and storage stage come from compressors and engines, with only a small contribution from pipeline leakage (Table 4). Emissions are estimated using emission factors (e.g., scf/mile/yr for pipeline leaks or scf/day for compressor leaks), pipeline mileage, and equipment counts based largely on measurements made in the 1990s.[25] Variations in leakage associated with the large seasonal movements of gas in and out of storage reservoirs were not considered when measurements were made, and this may introduce some uncertainty.

2.4 Gas Distribution

More than 1,500 companies manage the distribution of natural gas to about 70 million customers.[31] The EPA's estimate of methane emissions from gas distribution are for local pipeline distribution systems (an estimated 1.2 million miles of pipe) that are fed by the main transmission pipelines and through which the majority of customers receive their gas. (This excludes most electric power plants and about half of large industrial customers, which are connected directly to a main transmission pipeline and account for perhaps half of all gas used.[g]) A gas-distribution system includes stations where gas is metered and pressure-regulated



Natural gas meters in the distribution system.

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[g] In 2012, 36 percent of all gas used for energy was used in electric power generation and 33 percent was used in industry. Assuming all of the gas used for electric power and half of the gas used by industry was delivered via transmission pipelines, then approximately half of all gas used in the U.S. was delivered to users via transmission pipeline.

as it is transferred from a transmission line into a distribution network. It also includes the distribution pipelines, "services" (the pipe connecting a customer to a distribution main), and customer meters. The EPA estimates there are more than 63 million service connections in total, and it assumes no leakage occurs after the customer meter.

In the EPA 2012 inventory, the most significant leakage of methane is at the metering/regulating stations (Table 4). The EPA differentiates ten different station types according to function (metering and/or regulating) and the pressure of gas they each handle, and assigns a different emissions factor to each (ranging from 0.09 to 179.8 scf per station per year, based on measurements made in the 1990s[25]). The emissions factor for each type of station is multiplied by the estimated number of that type of station in operation in that year.

Leakage from distribution and service pipelines account for most of the rest of the estimated methane emissions from the distribution system. This leakage is calculated according to pipe type – cast iron, unprotected steel, protected steel, plastic, and copper – using a different emission factor for each type (in scf per mile per year) and service line (in scf per service per year). In the EPA inventory, cast-iron and unprotected steel pipes are assumed to have high leak rates, based on measurements made in the 1990s (Table 5)[h]. The inventory also estimates the number of miles of each type of pipe in the distribution system and the number of each type of service connection to customers based on data from the Pipeline and Hazardous Materials Safety Association (PHMSA)[32].

Table 5. Pipeline methane emission factors and pipeline mileage in EPA's 2013 inventory.[20]

	Annual Leak Rate (scf/mile)	**Miles of Pipe**
Distribution mains		
Cast iron	239,000	33,586
Unprotected Steel	110,000	64,092
Plastic	9,910	645,102
Protected steel	3,070	488,265
Transmission pipelines	**566**	**304,606**

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[h] Protected steel refers to carbon steel pipes equipped with a special material coating or with cathodic protection to limit corrosion that can lead to leakage. (Cathodic protection involves the use of electrochemistry principles.) The use of cast iron and unprotected steel pipes, which are susceptible to corrosion, is declining. Nevertheless, there are still an estimated 100,000 miles of distribution pipe made of cast iron or unprotected steel and more than 4.2 million unprotected steel service lines still in use.[23]

3. Other Estimates of GHG Emissions from the Natural Gas Supply System

When the EPA made relatively large methodology adjustments in its 2011 inventory (Figure 17), they included a provision to separately calculate emissions from the production of shale gas and conventional gas. This adjustment, together with the growing importance of shale gas in the U.S. supply (Box 1), led others to develop greenhouse gas emission estimates for natural gas. Many technical reports[33-42] and peer-reviewed journal papers[17,43-51] have appeared, with emissions estimates varying from one to the next.

All of the published analyses have been made using methodologies similar to the bottom-up approach used in the EPA inventory calculations, but each study varies in its input assumptions. Because of the diversity of natural gas basin geologies, the many steps involved in the natural gas system, the variety of technologies and industry practices used, and, perhaps most importantly, the lack of measured emissions data, a large number of assumptions must be made to estimate overall emissions. As a consequence, different authors come to different conclusions about the magnitude of upstream GHG emissions. For example, some conclude that upstream emissions per unit energy for shale gas are higher than for conventional gas[17,46] and others conclude the opposite.[33,43,49] Many of the authors rely on the same two information sources for many of their input assumptions,[52,53] leaving just a few key assumptions mainly responsible for differences among results.

Table 6. Estimates of upstream methane and CO_2 emissions for conventional gas and shale gas, with comparison to EPA estimates for the natural gas supply system as a whole.* (Emissions from gas distribution are not included here.)

UPSTREAM EMISSIONS	**Jiang[47]**		**NETL[33]**		**Hultman[46]**	**Stephenson[48]**		**Burnham[43]**		**Howarth[17]**		**Best[49]**		**EPA**
Methane, $kgCO_2e/GJ(LHV)$	Conv	Shale	Conv	Shale	Shale	Conv	Shale	Conv	Shale	Conv	Shale	Conv	Shale	All
Well pad construction		0.1	0.2	0.1				1.6	1.0		1.5	0.16	0.16	
Well drilling		0.2				0.3	0.3					0.23	0.2	
Hydraulic fracturing water		0.3					0.3						0.26	
Chemicals for hydraulic fracturing		0.1											0.07	
Well completion		1.0		1.3	4.7	0.4	1.6		0.8		8.6	0.18	1.2	
Fugitive well emissions	3.4	3.4	1.8	1.8	2.1	0.9	0.9	3.6	3.6	5.0	5.0	2.70	2.70	
Workovers				4.6	4.7				1.5				1.20	
Liquids unloading	2.5		6.6					5.9		0.6		3.80		
Production emissions	5.9	5.1	8.6	7.8	11.5	1.6	3.1	11.1	6.9	5.6	15.1	7.1	5.8	6.8
Processing emissions	1.5	1.5	1.2	1.2	0.6	0.5	0.5	0.8	0.8	0.4	0.4	1.8	1.8	0.9
Transmission emissions	1.9	1.9	2.3	2.3	1.8	1.7	1.7	0.9	0.9	6.8	6.8	1.9	1.9	2.4
Total upstream methane emissions	**9.3**	**8.5**	**12.1**	**11.3**	**13.9**	**3.8**	**5.3**	**12.8**	**8.6**	**12.8**	**22.3**	**10.8**	**9.5**	**10.0**
Carbon dioxide, $kgCO_2/GJ(LHV)$														
Flaring	0.4	0.4	1.8	2.0		2.8	2.8	0.4	0.4	4.1	4.1	0.6	0.6	
Lease/plant energy	3.7	3.7						4.3	4.1			3.2	3.2	
Vented at processing plant	1.0	1.0	0.2	0.2				0.8	0.8			1.2	1.2	
Transmission compressor fuel	0.4	0.4	0.4	0.4		0.2	0.2	0.3	0.3	0.6	0.6	0.4	0.4	
Total upstream CO_2 emissions	**5.5**	**5.5**	**2.4**	**2.6**		**3.0**	**3.0**	**5.8**	**5.6**	**4.7**	**4.7**	**5.4**	**5.4**	**4.6**
TOTAL UPSTREAM, $kgCO_2e/GJ(LHV)$	**14.8**	**14.0**	**14.5**	**13.9**	**13.9**	**6.8**	**8.3**	**18.6**	**14.2**	**17.5**	**27.0**	**16.2**	**14.9**	**14.6**

** Methane leakage has been converted to $kgCO_2e$ using a GWP of 25. Numbers in all but the EPA column are taken from Table SI-5 in the supplemental information for the paper by Weber and Clavin.[49] Numbers in the EPA column are my estimates based on the 2012 inventory (Table 4, but adjusted to GWP of 25) and total 2010 U.S. natural gas end-use consumption for energy.[54] CO_2 emissions in the EPA column include estimates from the EPA 2012 inventory[23] plus emissions from complete combustion of lease and plant fuel in 2010 that I have estimated based on EIA data.[55]*

3.1 Leakage During Gas Production, Processing, and Transmission

A careful analysis by Weber and Clavin[49] encapsulates well the diversity of estimates of upstream emissions that have been published relating to the gas production, processing, and transmission stages. They analyzed in detail the assumptions made in six different studies and took care to normalize estimates from each study to eliminate differences arising from inconsistent assumptions between studies, such as different values for methane GWP, methane fraction in natural gas, and other variables. Weber and Clavin excluded distribution emissions estimates from their comparisons.

Table 6 shows their normalized estimates in units of grams of CO$_2$-equivalent per megajoule of lower heating value (MJ$_{LHV}$) natural gas energy,[i] assuming a methane GWP of 25. "Best" refers to what Weber and Clavin consider their best estimate based on their analysis, including a Monte Carlo uncertainty analysis, of all of the studies. For comparison, I have added estimates of emissions based on the EPA 2012 inventory (year 2010 values, adjusted for a methane GWP of 25).

Figure 19, taken from Weber and Clavin, graphs numbers from Table 6, and shows estimated uncertainty ranges.[j] For shale gas five of the seven estimates are similar (13.9 to 14.9 gCO$_2$e/MJ$_{LHV}$), with estimates based on Howarth[17] and Stephenson[48] being markedly higher and lower, respectively. Uncertainty ranges in most cases overlap each other. For conventional gas, the estimates based on Burnham and Stephenson represent the highest and lowest estimates, with the others falling in the range 14.5 to 17.5 gCO$_2$e/MJ$_{LHV}$.

As seen from Table 6, the largest upstream CO$_2$ emissions are due to combustion of natural gas used for energy in processing and transmission stages (lease and plant fuel plus transmission compressor fuel). The numbers in Table 6 suggest that the global warming impact of upstream CO$_2$ emissions accounts for about one third of the combined impact of CO$_2$ plus methane,

Estimates of Upstream Emissions in the Natural Gas System Vary Widely

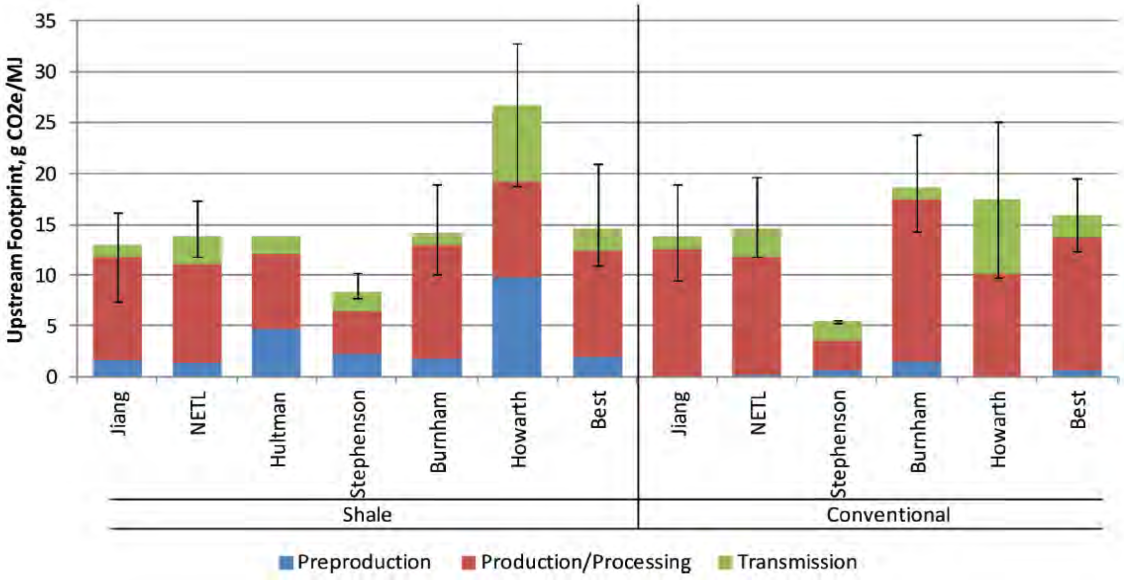

Figure 19. A diversity of estimates exist in the literature for GHG emissions associated with natural gas production, processing, and delivery. This graph, from Weber and Clavin[49] (and consistent with numbers in Table 6, but using different sub-groupings) shows upstream emissions in units of grams of CO$_2$e/MJ$_{LHV}$ of natural gas, excluding emissions associated with natural gas distribution. Ranges of uncertainty are also indicated. "Best" refers to Weber and Clavin's own estimates.

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[i] The energy content of a fuel can be expressed on the basis of its lower heating value (LHV) or its higher heating value (HHV). The difference between the LHV and HHV of a fuel depends on the amount of hydrogen it contains. The heating value of a fuel is determined by burning it completely under standardized conditions and measuring the amount of heat released. Complete combustion means that all carbon in the fuel is converted to CO$_2$ and all hydrogen is converted to water vapor (H$_2$O). The heat released as a result of these oxidation processes represents the LHV of the fuel. If the water vapor in the combustion products is condensed, additional heat is released and the sum of this and the LHV represents the HHV of the fuel. For fuels with low hydrogen content, like coal, relatively little water vapor forms during combustion, so the difference between LHV and HHV is not especially large. The high hydrogen content of methane, CH$_4$, means the difference between LHV and HHV is more significant. Delivered natural gas, which is mostly methane, has an HHV that is about 11 percent higher than its LHV.

[j] Category groupings in Figure 19 are different from those in Table 6, but overall totals are the same.

a not insignificant fraction. However, this is based on assuming a methane GWP of 25 (100-year time frame). Were a higher GWP value (shorter time frame) to be considered, methane would have a higher impact, and the impact of CO_2 would be correspondingly reduced.[k]

Leaving aside the upstream CO_2 emissions for the moment, it is possible to remove the complication introduced by the choice of GWP value by expressing the methane emissions in physical terms as a percent of total methane extracted from the ground. This total methane leakage during production, processing, and transmission, as estimated in the various studies, ranges from an average of under 1 percent to 2.6 percent for conventional gas and from 1 percent to 4.5 percent for shale gas (Table 7). The EPA 2012 inventory estimate corresponds to a leakage of 2 percent (which increases to 2.2 percent if leakage from the distribution system is included). The methane leak rates corresponding to the lower and upper ends of the uncertainty ranges for the "Best" case in Figure 19 are 0.9 percent to 3.4 percent for conventional gas and 0.7 percent to 3.8 percent for shale gas. The uncertainty range for shale gas in the highest emissions case (Howarth) corresponds to leakage of 3.3 percent to 7.0 percent[l] (not shown in Table 7). Notably, the lower bound of this range is nearly as high as the upper end of the uncertainty ranges for any of the other shale gas results shown in Figure 19. (Howarth's range for conventional gas is 1.6 percent to 3.8 percent.)

Some perspective on the estimates in Table 7 is provided by O'Sullivan and Paltsev,[50] who estimate leakage during completion (including hydraulic fracturing) of shale gas wells in the same shale basins (Barnett and Haynesville) as considered by Howarth.[m] O'Sullivan and Paltsev drew on gas production data for 1785 shale gas wells that were completed in 2010 in the Barnett formation and 509 in the Haynesville formation. They estimated well completion emissions by assuming that for each well the "flowback" of hydraulic fracking fluid (see Section 2.1) occurs over a 9 day period and that the amount of gas brought to the surface with the fluid during this period rises linearly from zero at start to a maximum at the end of the period equal to the peak gas production rate reported for the well. They further assume that current field practice for gas handling is represented by an assumption that, on average, 70 percent of the flowback gas is captured for sale, 15 percent is flared at the wellhead (converted to CO_2), and 15 percent is vented without flaring. They acknowledge the uncertainties in this latter assumption, stating that "significant opaqueness surrounds real world gas handling practices in the field, and what proportion of gas produced during well completions is subject to which handling techniques." Their estimate of average per-well emissions in the Barnett formation is 7 times less than the estimate of Howarth *et al.*,[17] who assume that all flowback gas is vented. For the Haynesville formation, the difference between the estimates in the two studies is a factor of 30.

*Table 7. Upstream methane leakage (excluding leakage in distribution systems) as a percentage of methane production for the studies shown in Table 6 and Figure 19.**

	Jiang		**NETL**		**Hultman**	**Stephenson**		**Burnham**		**Howarth**		**Best**		**EPA**
	Conv	Shale	Conv	Shale	Shale	Conv	Shale	Conv	Shale	Conv	Shale	Conv	Shale	All
	Methane leakage (percentage of methane production)													
Production	1.2	1.0	1.7	1.5	2.2	0.3	0.6	2.2	1.3	1.1	3.0	1.4	1.1	1.37
Processing	0.3	0.3	0.2	0.2	0.1	0.1	0.1	0.2	0.2	0.1	0.1	0.4	0.4	0.19
Transmission	0.4	0.4	0.4	0.4	0.3	0.3	0.3	0.2	0.2	1.4	1.4	0.4	0.4	0.48
TOTAL	**1.9**	**1.7**	**2.4**	**2.2**	**2.7**	**0.7**	**1.0**	**2.5**	**1.7**	**2.6**	**4.5**	**2.1**	**1.9**	**2.02**

** Based on Table 6 and (for all but the EPA numbers) energy contents of produced gas per kg of contained methane reported by Weber and Clavin:[49] Jiang (50 $MJ_{LHV}/kgCH_4$), NETL (48.8), Hultman (48.2), Stephenson (47.3), Burnham (48.6), Howarth (50.0), and Best (48.8). The EPA estimate assumes a gas energy content of 51.5 $MJ_{LHV}/kgCH_4$ for consistency with EPA numbers in Table 6.*

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[k] For example, with GWP = 72 (20-year time frame), CO_2 emissions would be less than 15 percent of total CO_2-equivalent emissions in most cases.

[l] The paper by Howarth, *et al.*[17] gives total estimated system leakage fractions (including leakage in distribution), of 3.6 percent to 7.9 percent. I have estimated the range for distribution leakage, based on discussion in that paper, to be 0.35 percent to 0.9 percent and removed this from the original Howarth *et al.* estimates to provide a consistent figure for comparison with the others' results.

[m] O'Sullivan and Paltsev also made estimates for wells in the Fayetteville, Marcellus, and Woodford formations.

Table 8. Comparison of estimates for methane leakage during completion of shale gas wells in two different formations.

	O'Sullivan[50] kgCH$_4$ per well completion	Howarth[17](as quoted by O'Sullivan[50]) kgCH$_4$ per well completion
Barnett formation	35.1	252
Haynesville formation	151.3	4638

O'Sullivan and Paltsev report an estimate of total methane emissions from all U.S. shale well completions in 2010 of 216,000 metric tons of methane. EPA's estimate for 2010 using its 2012 inventory methodology was close to this value (181,000 tons), but using the methodology reported in its 2013 inventory, the emissions are more than triple this value (795,000 tons). (See Table 4.) Thus, there continues to be significant uncertainty about what average well completion emissions are.

Uncertainties may be reduced in the future when a new EPA rule takes effect starting in 2015. The rule requires all new hydraulically fractured shale gas wells to use commercially-established "green completion" technologies to capture, rather than vent or flare, methane. The EPA estimates that 95 percent or more of the methane that might otherwise be vented or flared during well completion will be captured for sale. Wyoming and Colorado already require green completions on all shale wells.

The new EPA rule is significant because there is general agreement that methane leakage in the gas production phase is among the most significant leakages in the entire natural gas system, a conclusion supported by some recent measurements of the concentrations of methane in the air above gas wells,[56,57,58] including a reported leakage rate of 9 percent from oil and gas production and processing operations in the Uinta Basin of Utah,[59] and 17 percent of production in the Los Angeles Basin.[60] Such estimates, based on "top-down" measurements, involve large uncertainties, but draw attention to the need for more and better measurements that can help reduce the uncertainty of estimated leakage from natural gas production. Some such measurements are underway.[61]

Well completion emissions are only one of several important leakage components in gas production. In Weber and Clavin's review, they identified six assumptions that contribute most significantly to variations in overall estimates from one study to another: *i)* the number of workovers per shale-gas well, *ii)* the well completion and workover emissions factor, *iii)* the liquids unloading emissions factor (for conventional gas wells), *iv)* the rate of fugitive emissions at the wellhead, *v)* the fugitive emissions during gas processing, *vi)* and the EUR.

The last of these requires some explanation. Emissions that occur only once over the lifetime of a well (e.g., well completion emissions) or only a limited number of times (e.g, liquids unloading) are converted into an estimate of emissions per unit of gas produced by dividing the estimated emission by the total gas production from the well over its full lifetime – the well's estimated ultimate recovery (EUR). Because the shale gas industry is still young, there is a limited production history with wells on which to base EUR estimates. O'Sullivan and Paltsev[50] have noted that there is "appreciable uncertainty regarding the level of ultimate recovery that can be expected from shale wells." The challenge of determining what EUR to use to accurately represent leakage per unit of gas production is compounded by the large and inherent variability in EUR across different wells. Mean EUR values estimated by the U.S. Geological Survey[62] for wells in different shale formations (based on decline-curve analysis using a limited amount of monthly production data), vary by a factor of 60 from largest to smallest. Within a given formation, the maximum estimated EUR can be up to 1,000 times larger than the estimated minimum EUR. In Weber and Clavin's "Best" estimate in Figure 19, the uncertainty range in emissions results in part from assumed average EUR values from a low of 0.5 to a high of 5.3 billion cubic feet per well. (The authors state that an EUR of 2 bcf is the "most likely" value.) This order-of-magnitude range in EUR highlights the (significant) uncertainty introduced in using EUR to estimate leakage fractions.

3.2 Leakage from Gas Distribution Systems

Studies reviewed in the previous section were concerned primarily with gas leakage in connection with power generation. Leakage from gas distribution systems was excluded in those studies because most gas-fired power plants receive gas directly from the gas transmission system. But gas used in residential and commercial buildings and smaller industrial facilities – about half of all gas used – passes through the distribution system before reaching a user. The EPA 2012 inventory estimates that leaks in the distribution system account for 13 percent of all upstream methane leakage (Table 4), or less than 0.3 percent of methane produced. But the sheer size and diversity of the gas distribution infrastructure – over a million miles of varying-vintage distribution mains, more than 60 million service pipelines connecting the mains to users, the large number of metering and pressure-regulating stations found at the interface of transmission and distribution systems and elsewhere – and the limited number of leakage measurements that have been made suggest that there could be large uncertainties in the EPA estimate.

One study[63] in Sao Paulo, Brazil, which measured leakage from cast-iron distribution mains, highlights the uncertainties. In the 1950s, cast-iron was the standard material used for distribution mains in the U.S. Sao Paulo has a cast-iron distribution network comparable to or younger than the U.S. cast-iron network. Much of the cast iron in the U.S. has been replaced with less-leaky steel or plastic in recent decades, but there are still an estimated 35,000 miles of cast-iron pipe still in everyday use in the U.S. When cast-iron pipes leak it is typically at the joints where 12-foot long pipe sections are fitted together in "bell and spigot" arrangements. The jute fiber that was routinely used as the sealant dries out over time, leading to leakage. There are about 15 million such joints in the U.S. distribution system today. Comgas, the natural gas utility in Sao Paulo, measured leak rates in over 900 pipe sections in their network. Based on these measurements, they conservatively estimated an average annual leak rate of 803,548 scf per mile of pipe, more than triple the emission factor used in the 2012 EPA inventory (Table 5).[n] In some 15 percent of the Comgas measurements, emissions were two million scf per mile or higher.

New "top-down" measurement approaches are being pursued to try to improve estimates of leakage from the distribution system. These involve measuring methane concentrations in the air above a defined region and analyzing these in conjunction with wind patterns and other variables to try to estimate what leakage originated from the natural gas system. Recent measurements have identified elevated methane concentrations above urban streets in Boston,[64] San Francisco,[65] and Los Angeles.[66] Work is ongoing in acquiring more measurements to help estimate associated leak rates.[61,65]

4. Natural Gas vs. Coal in Electricity Generation

The growing use of natural gas for power generation in place of coal makes it particularly important to understand methane leakage and its global warming implications. This issue has been discussed by others[17,33,43,46,47,49] with varying conclusions due in large part to different methane leakage rate assumptions (as discussed in Section 3.1). In the absence of greater certainty about actual methane leakage rates, it is especially informative to understand the prospective global warming impact of different overall leakage rates when natural gas electricity displaces coal electricity.

Figure 20 shows total lifecycle greenhouse gas emissions associated with natural gas (independent of end use) per unit of energy for different assumed total system leakage rates. The red portion of each bar

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[n] Comgas subsequently implemented an effort to place plastic inserts in their cast-iron distribution mains to reduce leakage. The extent to which such leak mitigation measures have been applied in the U.S. is difficult to determine. Some U.S. gas utilities utilize pipe-crawling CISBOTs (cast-iron joint sealing robot) that add sealant to jute-packed joints by self-navigating through distribution mains, thereby reducing the need for more costly excavation to repair or replace pipes.[65,63]

Even Small Methane Leaks Can Have a Large Global Warming Impact in the Short Term

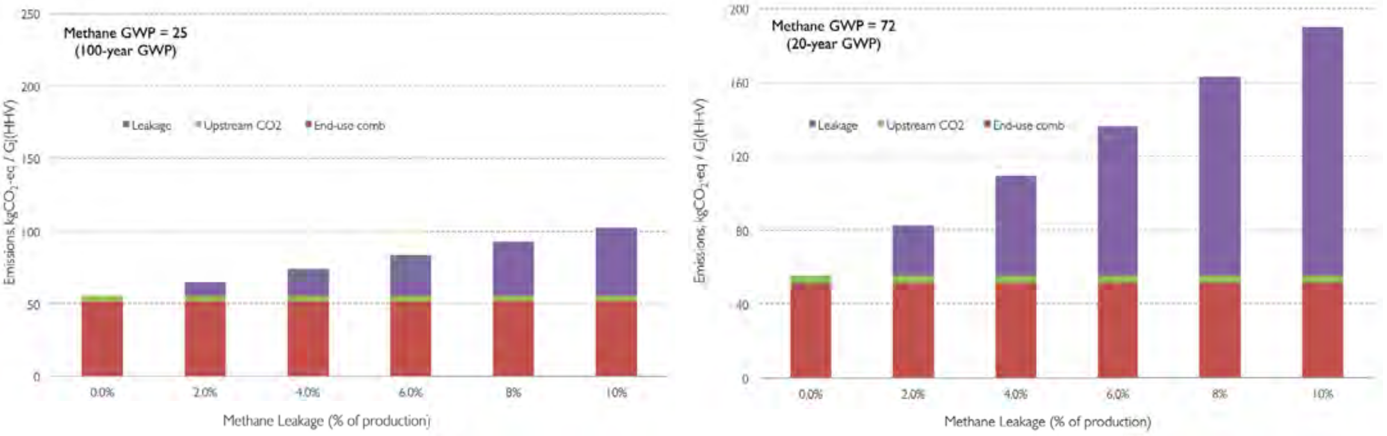

Figure 20. *Estimates of greenhouse gas emissions from natural gas production, processing, delivery, and end-use for different assumed rates of upstream methane leakage.*

represents end-use combustion emissions.[o] Purple is the contribution from methane leakage corresponding to leakage fractions on the x-axis.[p] Green represents the comparatively small direct "upstream" CO_2 emissions. (The latter result from combustion of natural gas used as fuel at gas processing plants and in the gas transmission system and from CO_2 that originated underground and was removed from the natural gas during gas processing.[q])

The left and right graphs include the same physical emissions, but represent these using 100-year and 20-year GWPs for methane, respectively. When there is leakage the choice of time horizon affects the global warming impact estimate tremendously, since the GWP for a 20-year time horizon is nearly triple the GWP for a 100 year horizon (Table 3).

As a point of reference, the EPA's 2012 inventory estimate of GHG emissions from the natural gas system is approximated by the 2 percent leakage case in the left panel (100-yr GWP). Also, as a reminder, other leakage estimates discussed in Section 3.1 ranged from 1 percent to 7 percent (excluding any gas distribution leakage).

With 2 percent leakage and a 100-yr GWP (left-panel), emissions of CO_2 from end-use combustion dominate total emissions. Methane leakage contributes only about 15 percent to the total global warming impact. Only if methane leakage is at the high end in this graph (10 percent leakage) does the global warming impact of leakage approach the level of combustion emissions. When a 20-year GWP is considered instead (right panel), leakage of only 4 percent is sufficient to cause a global warming impact equal to that from gas combustion alone. With 10 percent leakage, the impact of methane leakage is triple the impact from combustion alone.

Going a step further, we can calculate emissions per kilowatt-hour of electricity from natural gas and compare this with those for coal electricity. As noted earlier, natural gas contains much less carbon per unit of energy than coal and can be converted more efficiently into electricity. Power plant efficiencies for both coal and gas are well known. A representative efficiency for a modern natural gas combined cycle power plant is 50 percent (higher heating value basis).[67] Representative efficiencies for plants using pulverized bituminous coal are 31 percent for a "sub-critical" plant[68] and 36

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[o] Assuming complete combustion of natural gas containing 14 kg of carbon per GJ_{HHV}. This corresponds to an assumed natural gas composition by volume of 97.01percent methane, 1.76 percent ethane, 0.47 percent nitrogen, 0.38 percent CO_2, 0.26 percent propane, and 0.11 percent n-butane and an elemental composition by weight of 74.0 percent C, 24.4 percent H, 0.8 percent N, and 0.7 percent O. The average molecular weight is 16.57 g/mol, and the LHV and HHV are 47.76 MJ/kg and 52.97 MJ/kg, respectively.

[p] The methane leakage (in $kgCO_2e/GJ_{HHV}$) as a function of the percentage of production leaked is calculated, using the natural gas characteristics in footnote o, as follows: $\frac{kgCO_2e}{GJ_{HHV}} = GWP * \frac{\%\ leaked}{100} * 14 \frac{kgC}{GJ_{HHV}} * \frac{16gCH_4}{molC} * \frac{1molC}{12gC}$

[q] Upstream CO_2 emissions include those reported by the EPA for the natural gas system[23] plus emissions from combustion of "lease and plant fuel" (which EPA excludes from its inventory for the natural gas system to avoid double counting). Lease and plant fuel emissions are estimated by assuming complete combustion of lease and plant fuel energy used in 2010 as reported by the Energy Information Administration.[54]

With Methane Leakage Natural Gas Power Generation Can Have a Similar or Higher Global Warming Impact as Coal Power Generation

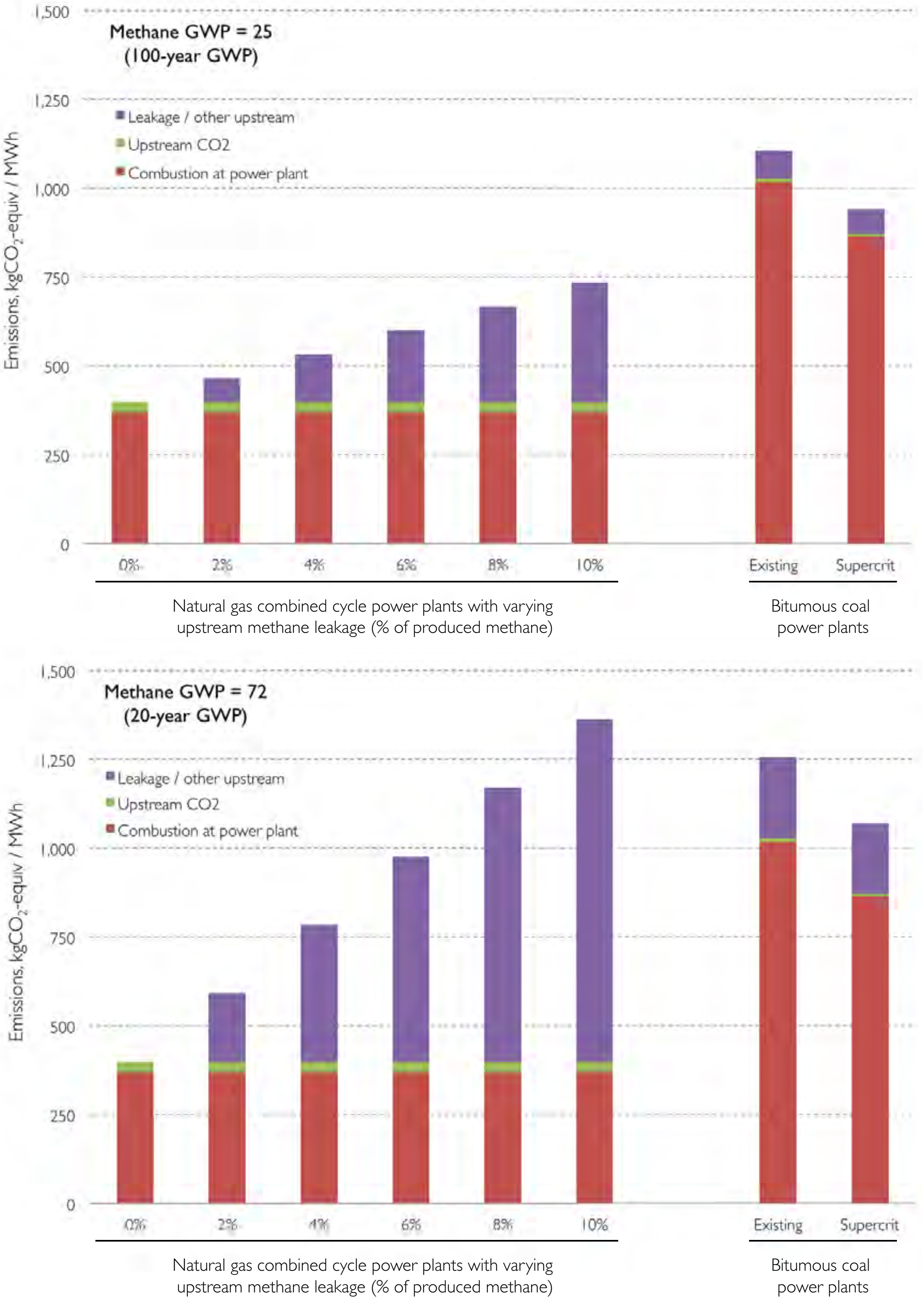

Figure 21. *Estimates of greenhouse gas emissions from electricity production from natural gas for different assumed rates of upstream methane leakage and from bituminous coal for typical existing coal plants and for a more efficient variant.*[r]

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[r] Based on emissions shown in Figure 20 and power plant fuel consumption of 7172 GJ_{HHV}/kWh a natural gas combined cycle (corresponding to 50.2 percent efficiency),[67] 11736 GJ_{HHV}/kWh (30.7 percent efficiency) for an existing subcritical coal-fired power plant[67] and 10019 GJ_{HHV}/kWh for a supercritical coal plant (35.9 percent efficiency)[68] Upstream CO_2 emissions for the subcritical and supercritical coal plants are 8.34 kg/MWh and 7.48 kg/MWh, respectively, and upstream methane emissions are 3.20 $kgCH_4$/MWh and 2.76 $kgCH_4$/MWh, respectively.[68,69]

percent for a "super-critical" plant.[69] (Most existing coal power plants use sub-critical steam pressures. Newer plants use super-critical pressures.)

With these efficiencies, Figure 21 shows our estimates of GHG emissions per kWh of electricity generated from natural gas (with different methane leakage rates) and from bituminous coal, assuming methane GWP time horizons of 100 years (top panel) and 20 years (bottom panel). These calculations include estimates of the "upstream" emissions associated with coal electricity, including estimated methane emissions that accompany mining of bituminous coal.[68,69]

With the 100-yr time horizon (top panel), the GHG emissions for a kwh of electricity from a natural gas plant are half the emissions from a kwh from an existing coal plant if methane leakage is under about 5 percent. Even with leakage as high as 10 percent, the natural gas kwh still has a lower global warming impact than the coal kwh – about one-third less.

In contrast, when the 20-yr time horizon is considered (bottom panel), leakage must be limited to about 2 percent for the natural gas kwh to have half the global warming impact of an existing coal plant's kwh. If leakage is about 8 percent, the natural gas kwh is no better for the climate than the kwh from an existing coal plant.

The comparisons in Figure 21 do not address the question of what is the "correct" GWP value to use in comparing the global warming impact of electricity from gas and coal. Alvarez *et al.*[18] have proposed a method for assessing the climate impact of a switch from one technology to another (such as coal to gas electricity generation) that involves more than one type of greenhouse gas emission, for example methane and CO$_2$. They define a technology warming potential (TWP) that represents the ratio of the time-dependent global warming potential of technology "A" divided by the time-dependent global warming potential of technology "B" that it replaces. By explicitly including the different atmospheric lifetimes of methane and CO$_2$, this method yields a ratio, for any time horizon of interest, that represents the relative global warming potential of switching from technology

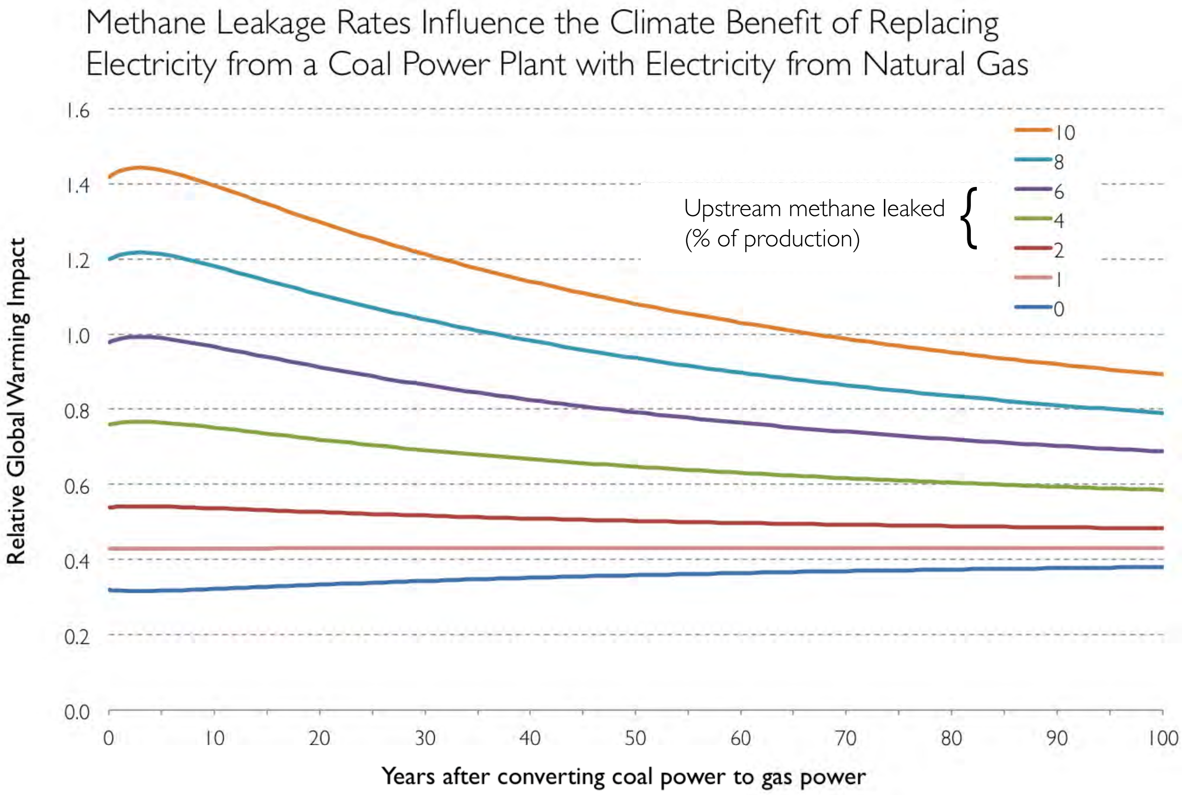

Figure 22. *Global warming impact of shifting electricity generation from a coal power plant to a natural gas power plant in year zero and continuing that generation from gas each year thereafter, assuming different methane leakage rates in the natural gas system. Natural gas is friendlier for the climate for values less than 1.0.*[s]

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[s] Assumed heat rates for electricity generation are 7172 kJ_{HHV}/kWh (6798 BTU/kWh) for NGCC and 10550 kJ_{HHV}/kWh (10000 BTU/kWh) for existing coal plants. Upstream emissions for coal are as described for subcritical coal in footnote r.

"A" to technology "B". The ratio varies with the time horizon due to the different atmospheric lifetimes of methane and CO_2. A ratio less than one at a particular point in time after a switch is made from "A" to "B" means that technology "A" has a lower global warming potential than technology "B" over that time frame.

Combining the TWP methodology of Alvarez *et al.* with our leakage assumptions, Figure 22 shows the global warming impact of replacing the electricity from a coal-fired power plant with natural gas electricity and then maintaining that natural gas generation for every subsequent year thereafter. Results are shown for different assumed total methane leakage rates expressed as a fraction of gas produced. For a time-frame of interest (x-axis), if the corresponding value on the y-axis is less than one, then the switch from coal to gas produces some level of climate benefit relative to maintaining electricity generation using coal. For example, if the y-axis value is 0.5 at some point in time, NGCC electricity has half as much global warming potential as coal over that time period.

Many authors have suggested that switching from coal to gas electricity halves the global warming impact of electricity generation. Figure 22 indicates that this is true if methane leakage is about 1.5 percent of production. If leakage were as high as 6 percent, the switch to gas would still be better for the climate than coal over any time period considered, although barely so in the earlier years after the switch. If leakage were 8 percent, switching from coal to gas would require 37 years before any climate benefit is achieved. With 10 percent leakage it takes 67 years. At these higher leak rates, a 50 percent climate benefit would not be realized for well over a century.

Figure 22 represents the impact of shifting one power plant worth of electricity generation from coal to gas. An important follow-on question is what woud be the global warming impact of shifting over time the whole fleet of coal power plants to gas. To provide some context in answering the question, it is helpful to know that the average rate at which coal electricity generation decreased over the decade from 2002 to 2012 in the U.S. was 2.4 percent per year. The annual percentage rate of reduction has been rising in recent years (Table 9). The decreased generation from coal has been predominantly replaced by increased generation from natural gas. (The combined electricity generation from gas plus coal grew an average of less than half of one percent per year during the past decade, Table 9.)

We extend the method presented by Alvarez *et al.* to analyze shifting of the whole coal fleet to gas over time. We assume an average annual percentage reduction in electricity generated from coal and a corresponding increase in electricity generated from gas,[t] with total electricity production from coal plus gas remaining the same each year.[u] If we assume a methane leakage rate of 2 percent of production, then Figure 23 shows the prospective global warming impact of switching from coal to natural gas electricity at different annual rates (compared to not replacing any coal electricity). With a 10 percent per year switching rate, it would take 29 years to replace 95 percent of coal generation. For the other cases, 95 percent coal replacement would be reached in 39 years (7.5 percent per year), 59 years (5 percent per year), 118 years (2.5 percent per year), or more than 200 years (1 percent per year).

As full replacement of coal is approached, the impact on global warming reaches a limiting value. Over a long enough time horizon, all of the cases will approach the same relative impact level of around 0.5 (for an assumed 2 percent leakage) but, importantly, this impact level is reached more slowly when coal replacement occurs more slowly. The slower the approach to the 0.5 level, the more rapid the rate of warming. Considering an often-used target year of 2050, 37 years from today, we see that the higher replacement rates (5, 7.5, and 10 percent per year) each achieves 40 percent or more reduction in global warming potential – approaching the maximum level reachable in the longer term. At the 2.5 percent per year replacement rate (roughly the average actual rate over the past decade), only a 29 percent reduction in warming potential is achieved by 2050.

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[t] For a constant annual percentage conversion of coal electricity to gas electricity, the fraction of original coal electricity converted to gas each year is $[r * (1 - r)^{(t-1)}]$ where r is the annual percentage reduction in coal electricity and t is the number of years from the start of the conversion process. (Conversion begins in year $t = 1$.)

[u] The Technology Warming Potential (TWP) defined by Alvarez *et al.*[18] (Equation 2 in their paper, with $L/L_{ref} = 1$) is used here to calculate the reduction in Global Warming Potential from substituting a unit amount of coal-generated electricity with gas-generated electricity in a given year and continuing to produce that unit amount of electricity from gas in subsequent years. (Figure 22 shows the result of this calculation.) When the amount of electricity made from natural gas is not constant every year but increases year to year (as coal electricity generation decrseases year to year) the climate impact of each new annual increment of gas electricity is assessed using the TWP. Then, the climate impact of the electricity generated from coal and gas in total in any year is the sum of climate impacts caused that year by each new increment of gas-generated electricity added from the start of the counting period up to that year plus the impact of the reduced amount of coal-generated electricity being produced in that year. Mathematically, the climate impact in total from the start of a shift from coal to gas over some number of years, N, is calculated as: $\int_{t=1}^{N}[r\,(1 - r)^{(t-1)} * \mathrm{TWP}(N + 1 - t)]dt + \{1 - \int_{t=1}^{N} [r\,(1 - r)^{(t-1)}]dt\}$ where r is the annual percentage reduction in coal electricity and $\mathrm{TWP}(N + 1 - t)$ is given by Equation 2 in Alvarez *et al.*

Table 9. U.S. coal and natural gas electricity generation 2002-2012 (left)[6] *and annual percentage reduction in coal electricity generation when averaged over different time periods (right).*

	Electricity Generated (1000 MWh per year)		
	Coal	Natural Gas	Coal + Gas
2002	1,933,130	691,006	2,624,136
2003	1,973,737	649,908	2,623,645
2004	1,978,301	710,100	2,688,401
2005	2,012,873	760,960	2,773,833
2006	1,990,511	816,441	2,806,952
2007	2,016,456	896,590	2,913,046
2008	1,985,801	882,981	2,868,782
2009	1,755,904	920,979	2,676,883
2010	1,847,290	987,697	2,834,987
2011	1,733,430	1,013,689	2,747,119
2012	1,517,203	1,230,708	2,747,911

Time Period	Average Annual Reduction in Coal Electricity
2002 - 2012	2.4 percent
2003 - 2012	2.9 percent
2004 - 2012	3.3 percent
2005 - 2012	4.0 percent
2006 - 2012	4.4 percent
2007 - 2012	5.5 percent
2008 - 2012	6.5 percent
2009 - 2012	4.8 percent
2010 - 2012	9.4 percent
2011 -2012	12.5 percent
-	-

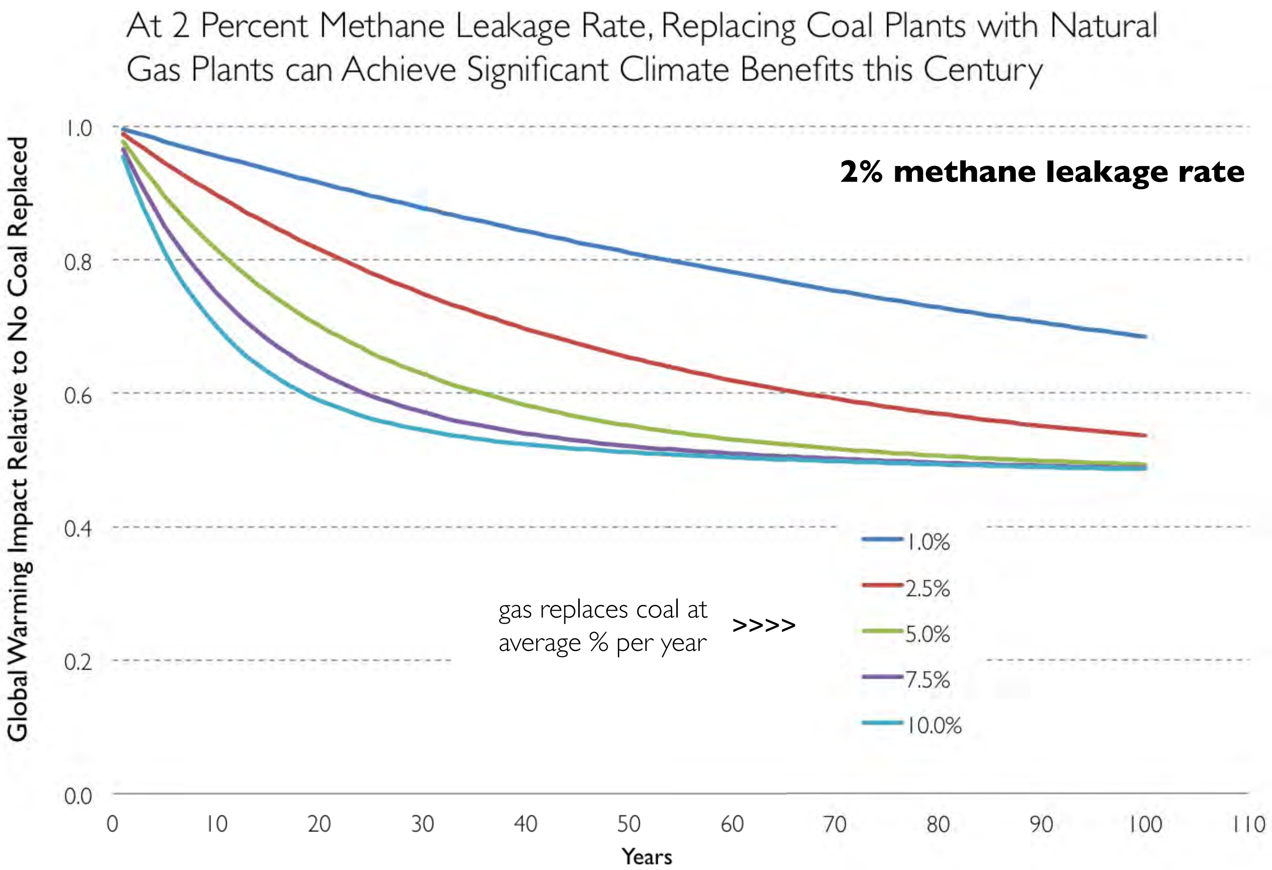

Figure 23. Relative global warming impact of natural gas combined cycle power replacing existing coal-fired power generation at different annual rates. In all cases the assumed methane leakage is 2 percent of production.

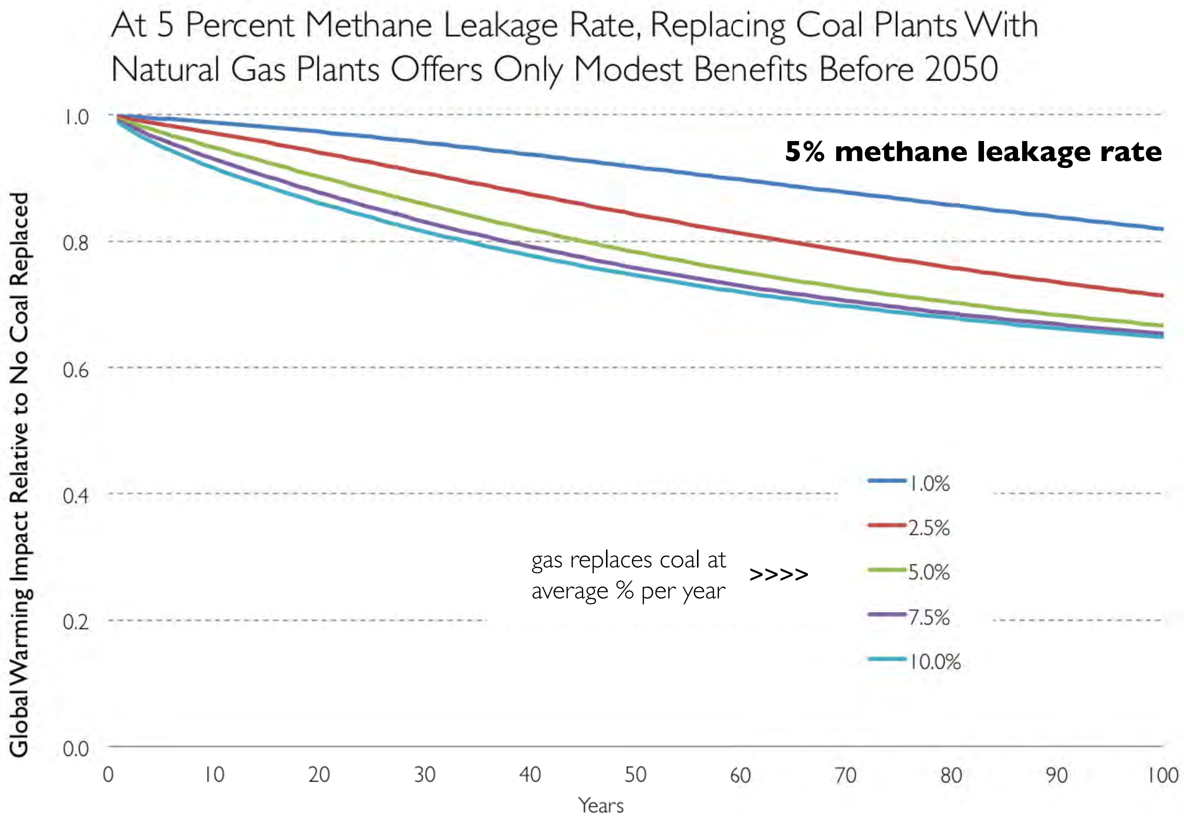
At 5 Percent Methane Leakage Rate, Replacing Coal Plants With Natural Gas Plants Offers Only Modest Benefits Before 2050

Figure 24. Relative global warming impact of natural gas combined cycle power replacing existing coal-fired power generation at different annual rates. In all cases the assumed methane leakage is 5 percent of production.

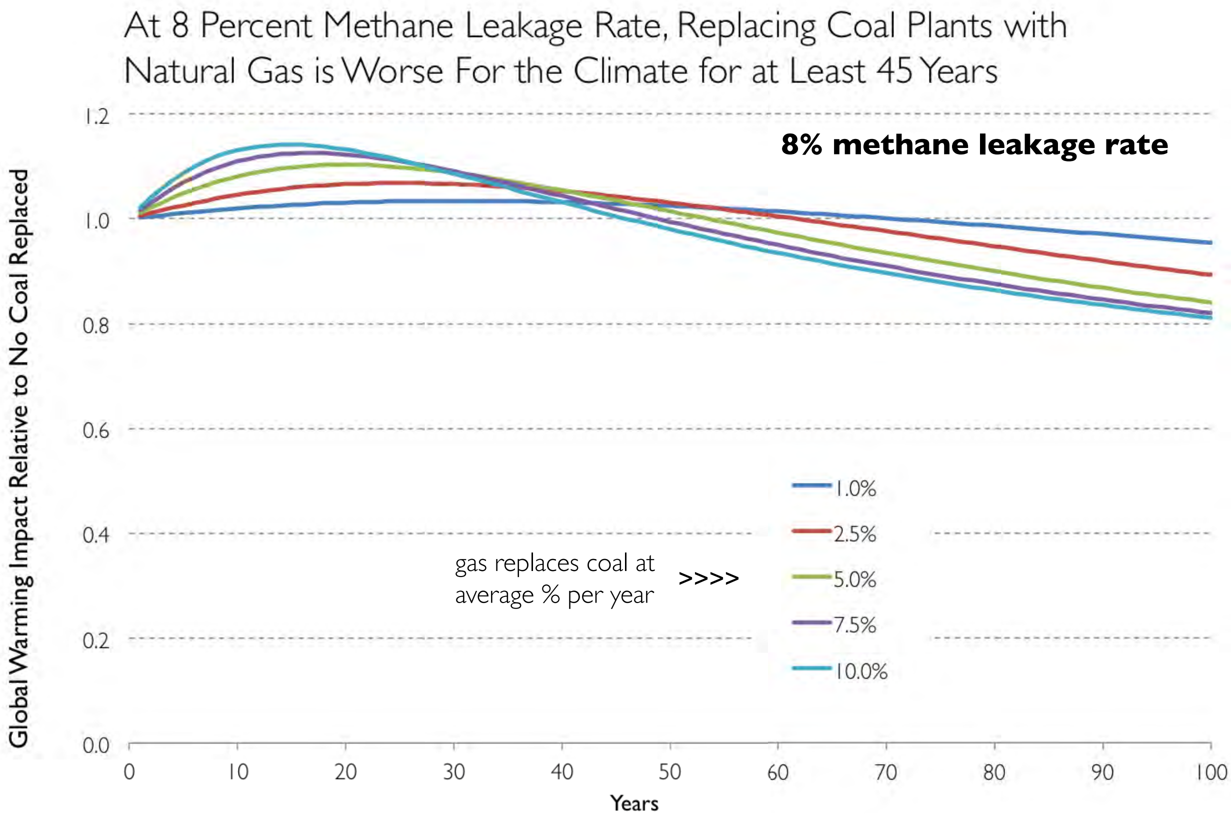
At 8 Percent Methane Leakage Rate, Replacing Coal Plants with Natural Gas is Worse For the Climate for at Least 45 Years

Figure 25. Relative global warming impact of natural gas combined cycle power replacing existing coal-fired power generation at different annual rates. In all cases the assumed methane leakage is 8 percent of production.

The same analysis can be carried out for a different assumed methane leakage rate. Figure 24 shows results for 5 percent leakage. Because of the higher methane leakage, the impact of switching from coal to gas is not as substantial as with lower leakage. In fact, by 2050, even the highest coal replacement rate of 10 percent/year achieves only about a 20 percent reduction in warming potential. The 2.5 percent replacement rate achieves only a 12 percent reduction compared with no coal-to-gas conversion.

As expected based on Figure 22, if leakage exceeds 6 percent, there would initially be negative impacts of switching from coal to gas nationally. With 8 percent leakage, a global warming benefit of switching from coal to gas is reached only after 45 years or more (Figure 25).

Finally, the different coal-to-gas substitution rates in Figure 23 and Figure 24 would have different gas supply requirements. If we consider 2013 as year 1 in these graphs, then the amount of additional gas supplies required in the U.S. to sustain the different rates of coal-to-gas substitution are as shown in Figure 26. Shown for comparison are the Energy Information Administration (EIA) projections of new gas supplies (for all end-uses of gas). New gas supplies could be higher than EIA projects, but the higher coal substitution rates (5 to 10

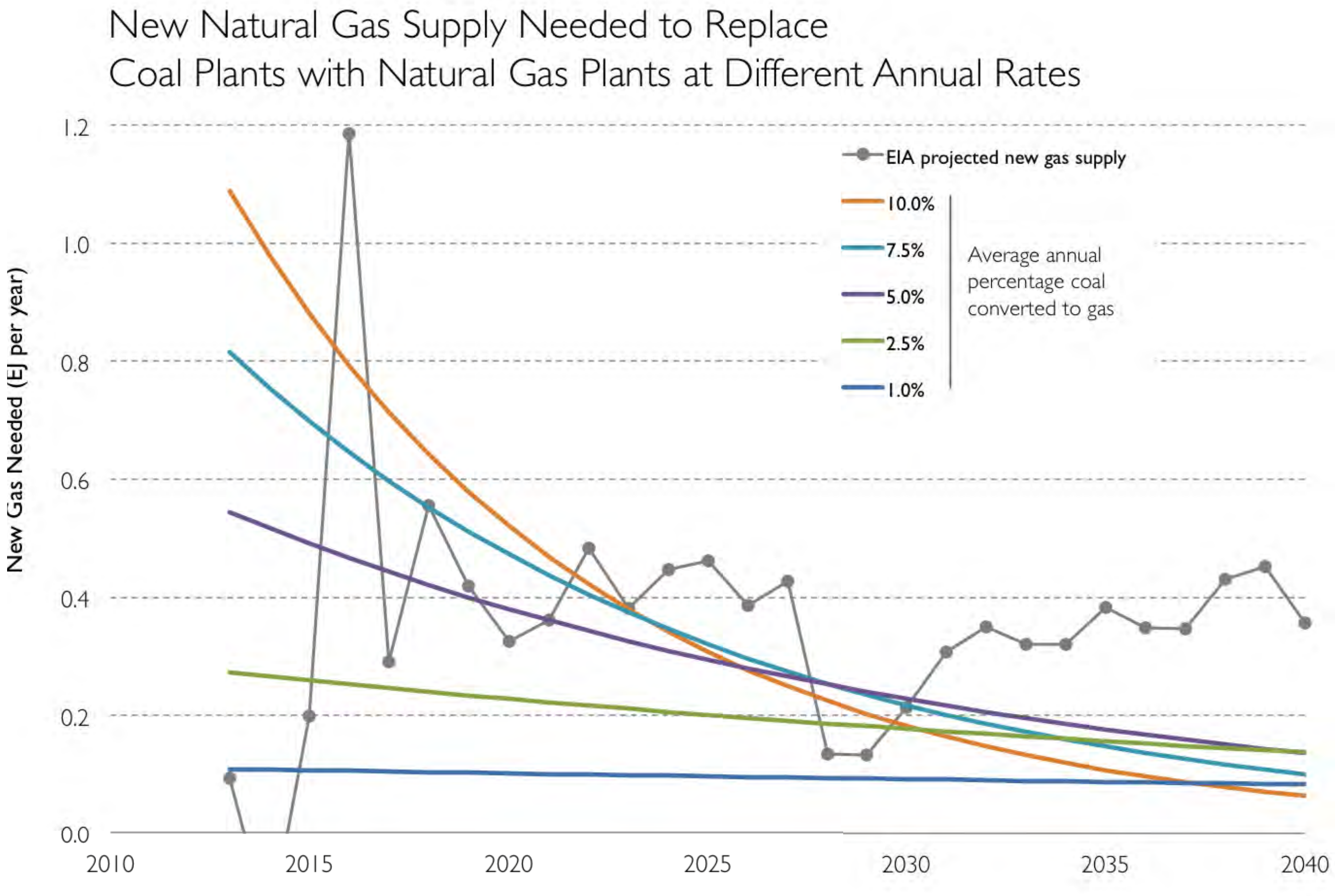

Figure 26. Additional gas required each year (compared to preceding year) under different scenarios. The solid lines represent the new gas required for electricity generation to replace coal-fired generation in the U.S. at the annual percentage rates indicated. (Coal-fired generation in 2012 was 1517 TWh. Gas generation that replaces coal is assumed to require 7,172 kJ of gas per kWh generated, corresponding to a heat rate of 6,798 BTU/kWh.) The black line is the new gas supply (for all gas uses) projected by the Energy Information Administration in its 2013 Annual Energy Outlook (Early Release) Reference Scenario[7] (There are approximately 1.1 EJ per trillion cubic feet (TCF) of gas.)

percent/year) would be difficult to achieve in the early years with the gas supply levels currently projected by the EIA, considering demands for gas from users other than electric power plants are also projected by EIA to grow during the projection period. In this context, the 2.5 percent per year rate may be an achievable average coal-to-gas shifting rate over the next several decades. In that case, the achievable reduction in global warming impact from substituting gas for coal out to 2050 would be 12 percent to 29 percent, considering methane leakage of 2 percent to 5 percent (Figure 23 and Figure 24). To achieve better than this would require other lower-carbon options, such as reduced electricity consumption and/or increased electricity supply from nuclear, wind, solar, or fossil fuel systems with CO_2 capture and storage to provide some of the substitution in lieu of gas over this time frame.

This analysis considered no change in leakage rate or in the efficiencies of power generation over time. The benefit of a switch from coal to gas would obviously increase if leakage were reduced and/or natural gas power generating efficiency increased over time.

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Methane Emissions from Natural Gas Systems

Background Paper Prepared for the National Climate Assessment
Reference number 2011-0003

Robert Howarth (Cornell University), Drew Shindell (NASA Goddard Space Institute), Renee Santoro (Cornell University), Anthony Ingraffea (Cornell University), Nathan Phillips (Boston University), and Amy Townsend-Small (University of Cincinnati)

February 25, 2012

The past few years have seen major changes both in our understanding of the importance of methane as a driver of global climate change and in the importance of natural gas systems as a source of atmospheric methane. Here, we summarize the current state of knowledge, relying on peer-reviewed literature.

Methane is the second largest contributor to human-caused global warming after carbon dioxide. Hansen and Sato (2004) and Hansen et al. (2007) suggested that a warming of the Earth to 1.8° C above the 1890-1910 baseline may trigger a large and rapid increase in the release of methane from the arctic due to melting of permafrost. While there is a wide range in both the magnitude and timing of projected carbon release from thawing permafrost in the literature (e.g. Schaefer et al., 2011), warming consistently leads to greater release. This release will therefore in turn cause a positive feedback of accelerated global warming (Zimov et al. 2006).

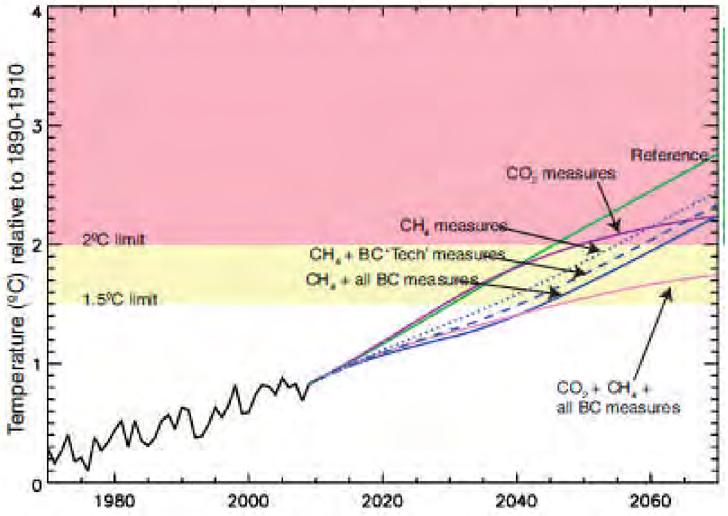

Fig. 1. Observed global mean temperature from 1900 to 2009 and projected future temperature under various scenarios of controlling methane + black carbon (BC) and carbon dioxide, alone and in combination. An increase to 1.5° to 2.0 ° C above the 1890-1910 baseline (illustrated by the yellow bar) poses high risk of passing a tipping point and moving the Earth into an alternate state for the climate system. Reprinted from Shindell et al. (2012).

Shindell et al. (2012) noted that the climate system is more immediately responsive to changes in methane (and black carbon) emissions than carbon dioxide emissions (Fig. 1). They predicted that unless emissions of methane and black carbon are reduced immediately, the Earth will warm to 1.5° C by 2030 and to 2.0° C by 2045 to 2050 whether or not carbon dioxide emissions are reduced. Reducing methane and black carbon emissions, even if carbon dioxide is not controlled, would significantly slow the rate of global warming and postpone reaching the 1.5° C and 2.0° C marks by 12 to 15 years. Controlling carbon dioxide as well as methane and black carbon emissions further slows the rate of global warming after 2045, through at least 2070.

Natural gas systems are the single largest source of anthropogenic methane emissions in the United States (Fig. 2), representing almost 40% of the total flux according to the most recent estimates from the U.S. Environmental Protection Agency (EPA) as compiled by Howarth et al. (2012). Note that through the summer of 2010, the EPA used emission factors from a 1996 study to estimate the contribution of natural gas systems to the U.S. greenhouse gas (GHG) inventory. Increasing evidence over the past 16 years has indicated these emission factors were probably too low, and in November 2010 EPA began to release updated factors. The estimates for natural gas systems in Fig. 2 are based on these updated emission factors and information released through 2011 in two additional EPA reports, as presented in Howarth et al. (2012). Note that the use of these new methane emission factors resulted in a doubling in the estimate of methane emissions from the natural gas industry. Note also that, to date, EPA has only increased emission factors for “upstream” and “midstream” portions of the natural gas industry (leaks and emissions at the well site and in processing gas). Factors for “downstream” emissions (storage systems and transmission and distribution pipelines) are still from the 1996 report, although EPA is considering also modifying these (Howarth et al. 2012). The natural-gas-system emissions in Fig. 2 are based on an average emission of 2.6% of the methane produced from natural gas wells over their production lifetime, with 1.7% from upstream and midstream emissions (for the national mix of conventional and unconventional gas in 2009) and 0.9% from downstream emissions (Howarth et al. 2012). As discussed below, these methane emission estimates from natural gas systems are based on limited data and remain uncertain.

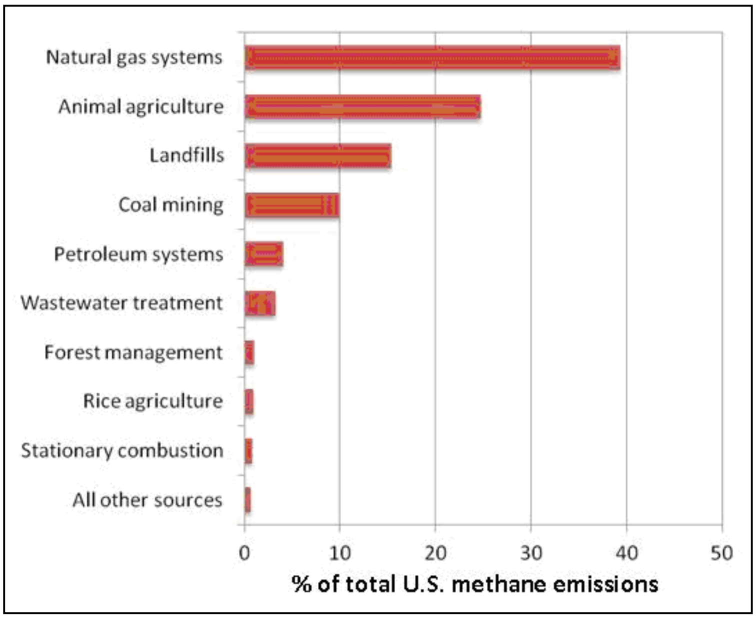

Fig. 2. Human-controlled sources of atmospheric methane from the United States for 2009, based on emission estimates from the U.S. Environmental Protection Agency in 2011. Reprinted from Howarth et al. (2012).

Recent estimates in the peer-reviewed literature for downstream emissions of methane from natural gas systems range from 0.07% to 10% of the methane produced over the lifetime of a well (Table 1). It is important to note that only Lelieveld et al. (2005) presented actual data on emissions, in their case leakage from high-pressure transmission pipelines. Other estimates are based on emission factors from the 1996 EPA study, on emission factors from a more recent report from the American Petroleum Institute, or on reports of "lost and unaccounted for gas" to governmental agencies, leading to high uncertainty. Lelieveld et al. reported a leakage rate from high-pressure transmission pipelines of 0.4% to 1.6%, with a "best estimate" of 0.7%; they used the 1996 EPA emission factors to estimate emissions from storage and distribution systems, yielding an estimate for total downstream emissions of 1.4% (or twice their measured value for just transmission). Howarth et al. (2011) took the "best estimate" of 1.4% from Lelieveld et al. (2005) as their low-end estimate, arguing that the 1996 EPA emission factors were probably low. For their high-end estimate, Howarth et al. (2011) used data on "missing and unaccounted for gas" from Texas. Their mean estimate of 2.5% is identical to the "best estimate" from Hayhoe et al.

Table 1. Estimates of methane emission from downstream emissions (transmission pipelines and storage and distribution systems) expressed as the percentage of methane produced over the lifecycle of a well. Studies are listed chronologically by date of publication. Modified from Howarth et al. (2012).

Hayhoe et al. (2002)	2.5 %	("best estimate;" range = 0.2% – 10%)
Lelieveld et al. (2005)	1.4 %	("best estimate;" range = 1.0% – 2.5%)
Howarth et al. (2011)	2.5 %	(mean; range = 1.4% – 3.6%)
EPA (2011)*	0.9 %	
Jiang et al. (2011)	0.4 %	
Hultman et al. (2011)	0.9 %	
Ventakesh et al. (2011)	0.4 %	
Burnham et al. (2011)	0.6 %	
Stephenson et al. (2011)	0.07 %	
Cathles et al. (2012)	0.7 %	

* The EPA (2011) estimate is as calculated in Howarth et al. (2012), using national emissions from EPA reports and national gas production data from US Department of Energy reports.

(2002). The estimates of Jiang et al. (2011), Hultman et al. (2011), Ventakesh et al. (2011), Burnham et al. (2011), and Cathles et al. (2012) are all based on various permutations of the 1996 EPA emission factors, factors that were developed before the measurements of Lelieveld et al. (2005). The "best estimate" of measured emissions from transmission pipelines of 0.7% by Lelieveld et al. (2005) is similar to or greater than the estimates for all downstream emissions (including storage and distribution) from these studies that used the 1996 EPA emission factors. The estimate of Stephenson et al. (2011) includes only transmission pipelines, is based on emission factors reported by the American Petroleum Institute in 2009 (which in turn are derived from the EPA 1996 emission factors), and is far lower than any other estimate. Comparisons of predicted and observed methane concentrations in Los Angeles have indicated that emissions factors for leakage from natural gas systems may be underestimated (Wunch et al. 2009; Hsu et al. 2010). A new study using stable isotopic and radiocarbon signatures of methane confirms that emission from natural gas systems is likely the dominant source of methane in Los Angeles (Townsend-Small et al. 2012).

Most recent estimates for upstream emissions (those that occur during well completion and production at the well site) and midstream emissions (those that occur during gas processing) for conventional natural

Table 2. Conventional natural gas, estimates of methane emissions from upstream (at the well site) plus midstream (at gas processing plants), expressed as the percentage of methane produced over the lifecycle of a well. Studies are listed chronologically by date of publication. Modified from Howarth et al. (2012).

Hayhoe et al. (2002)	1.2 %	("best estimate")
Howarth et al. (2011)	1.4 %	(mean; range = 0.2% to 2.4%)
EPA (2011)*	1.6 %	
Hultman et al. (2011)	1.3 %	
Venkatesh et al. (2011)	1.8 %	
Burnham et al. (2011)	2.0 %	
Stephenson et al. (2011)	0.4 %	
Cathles et al. (2012)	0.9 %	

* The EPA (2011) estimate is as calculated in Howarth et al. (2012), using national emissions from EPA reports and national gas production data from US Department of Energy reports.

gas cluster fairly closely to the new EPA estimate of 1.6% (Table 2). The mean estimate from Howarth et al. (2011) is 1.4%; the Howarth et al. (2011) low-end value of 0.2% is an estimate of what is possible using best technologies, while 2.4% reflects emissions using poor technologies. Other estimates range from 0.4% to 2.0% (Table 2). As for the downstream emissions, the lowest number (0.4%) comes from Stephenson et al. (2011).

Table 3. Unconventional gas (shale gas and gas from tight sands), estimates of methane emissions from upstream (at the well site) plus midstream (at gas processing plants), expressed as the percentage of methane produced over the lifecycle of a well. Studies are listed chronologically by date of publication. Modified from Howarth et al. (2012).

Howarth et al. (2011)	3.3 %	(mean; range = 2.2% to 4.3%)
EPA (2011)*	3.0 %	
Jiang et al. (2011)	2.0 %	
Hultman et al. (2011)	2.8 %	
Burnham et al. (2011)	1.3 %	
Stephenson et al. (2011)	0.6 %	
Cathles et al. (2012)	0.9 %	
Petron et al. (2012)	4.0 %	("best estimate;" range = 2.3 to 7.7%)

* The EPA (2011) estimate is as calculated in Howarth et al. (2012), using national emissions from EPA reports and national gas production data from US Department of Energy reports.

Estimates for upstream plus midstream methane emissions from unconventional gas (obtained from shales and tight-sands) vary from 0.6% to 4.0% for mean or "best" estimates (Table 3). The US EPA 2011 data indicate an estimated loss of 3.0% for upstream plus midstream emissions from unconventional gas (Howarth et al. 2012).

With the exception of the estimate by Petron et al. (2012), all of these upstream emissions for unconventional gas are based on sparse and poorly documented data (Howarth et al. 2011, 2012). The study by Petron et al. (2012) measured fluxes from an unconventional gas field – at the landscape scale – over the course of a year, and is a robust estimate. Although it represents only one field (the Piceance tight-sands basin in Colorado), emissions during the flowback period following hydraulic fracturing for unconventional gas are similar in this basin to other unconventional gas basins for which data are available (Howarth et al. 2011).

The Petron et al. (2012) study should be repeated in other unconventional gas fields, but it nonetheless suggests that most of the estimates in Table 3 are likely to be too low.

The methane emissions during flowback of fracking fluids, which occur during a 1-2 week period following hydraulic fracturing, are the major difference in emissions between unconventional and conventional gas. Flowback emissions are estimated as 1.9% of the lifetime production of an unconventional gas well according to Howarth et al. (2011), although the data of Petron et al. (2012) suggest the flux may in fact be greater. Flowback does not occur when a conventional gas well is completed, and the methane emissions at the time of well completion are far less (Howarth et al. 2011, 2012). Howarth et al. (2012), which was published before the Petron et al. (2012) study was released, concluded that shale gas emissions are 40% to 60% greater than emissions from conventional natural gas, when both upstream and downstream emissions are considered.

The US Department of Energy predicts that the major use of shale gas over the next 23 years will be to replace conventional reserves of natural gas as these become depleted. To the extent that methane emissions associated with shale gas and other unconventional gas are greater than for conventional gas, this will increase the methane emissions from the US from the natural gas industry beyond those indicated in Fig. 2. An increase of 40% to 60% in methane emissions is likely, based on the majority of studies summarized in Howarth et al. (2012), possibly more in light of the new field-based measurements by Petron et al. (2012). Note further that to the extent the US EPA is underestimating emissions from downstream sources (storage, transmission, and distribution), methane emissions from natural gas systems may already be substantially greater than shown in Fig. 2.

Global warming potentials provide a relatively simple approach for comparing the influence of methane and carbon dioxide on climate change. In the national GHG inventory, the US EPA uses a global warming potential of 21 over an integrated 100-year time frame, based on the 1995 report from the Intergovernmental Panel on Climate Change (IPCC) and the Kyoto protocol. However, the latest IPCC Assessment from 2007 used a value of 25, while more recent research that better accounts for the interaction of methane with other radiatively active materials in the atmosphere suggests a mean value for the global warming potential of 33 for the 100-year integrated time frame (Shindell et al. 2009). Using this value and the methane emission estimates based on EPA data shown in Fig. 2, Howarth et al. (2012) calculated that methane contributes 19% of the entire GHG inventory of the U.S., including carbon dioxide and all other gases from all human activities. The methane from natural gas systems alone contributes over 7% of the entire GHG inventory of the U.S. Note that the variation in the global warming potential estimates between 21 and 33 is substantially less than the variation among the methane emission estimates.

The global warming potentials of 21, 25 and 33 are all for an integrated 100-year time frame following emission of methane to the atmosphere. The choice of 100 years is arbitrary, and one can also consider the global warming potentials at longer or shorter time scales. To date, estimates have typically been provided at time scales of 20 years and 500 years, in addition to the 100-year time frame. An emphasis on the 20-year time frame in addition to the widely-used 100-year timeframe is important, given the urgency of reducing methane emissions and the evidence that if measures are not taken to rapidly reduce the rate of warming, the Earth will continue to warm so quickly that risk of dangerous consequences will grow markedly. We may reach critical tipping points in the climate system, on the time scale of 18 to 38 years (Figure 1).

For the 20-year time frame, Shindell et al. (2009) provide a mean estimate of 105 for the global warming potential. Using this value, Howarth et al. (2012) calculated that methane contributes 44% of the entire GHG inventory of the U.S., including carbon dioxide and all other gases from all human activities. Hence while methane is only causing about 1/5 of the century-scale warming due to US emissions, it is responsible for nearly half the warming impact of current US emissions over the next 20 years. At this time scale, the methane emissions from natural gas systems contribute 17% of the entire GHG inventory of the U.S., for all gases from all sources. We repeat that these estimates may be low, and that the gradual replacement of conventional natural gas by shale gas is predicted to increase these methane fluxes by 40% to 60% or more (Howarth et al. 2012).

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LETTER

Methane and the greenhouse-gas footprint of natural gas from shale formations

A letter

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Abstract We evaluate the greenhouse gas footprint of natural gas obtained by high-volume hydraulic fracturing from shale formations, focusing on methane emissions. Natural gas is composed largely of methane, and 3.6% to 7.9% of the methane from shale-gas production escapes to the atmosphere in venting and leaks over the lifetime of a well. These methane emissions are at least 30% more than and perhaps more than twice as great as those from conventional gas. The higher emissions from shale gas occur at the time wells are hydraulically fractured—as methane escapes from flow-back return fluids—and during drill out following the fracturing. Methane is a powerful greenhouse gas, with a global warming potential that is far greater than that of carbon dioxide, particularly over the time horizon of the first few decades following emission. Methane contributes substantially to the greenhouse gas footprint of shale gas on shorter time scales, dominating it on a 20-year time horizon. The footprint for shale gas is greater than that for conventional gas or oil when viewed on any time horizon, but particularly so over 20 years. Compared to coal, the footprint of shale gas is at least 20% greater and perhaps more than twice as great on the 20-year horizon and is comparable when compared over 100 years.

Keywords Methane · Greenhouse gases · Global warming · Natural gas · Shale gas · Unconventional gas · Fugitive emissions · Lifecycle analysis · LCA · Bridge fuel · Transitional fuel · Global warming potential · GWP

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Many view natural gas as a transitional fuel, allowing continued dependence on fossil fuels yet reducing greenhouse gas (GHG) emissions compared to oil or coal over coming decades (Pacala and Socolow 2004). Development of "unconventional" gas dispersed in shale is part of this vision, as the potential resource may be large, and in many regions conventional reserves are becoming depleted (Wood et al. 2011). Domestic production in the U.S. was predominantly from conventional reservoirs through the 1990s, but by 2009 U.S. unconventional production exceeded that of conventional gas. The Department of Energy predicts that by 2035 total domestic production will grow by 20%, with unconventional gas providing 75% of the total (EIA 2010a). The greatest growth is predicted for shale gas, increasing from 16% of total production in 2009 to an expected 45% in 2035.

Although natural gas is promoted as a bridge fuel over the coming few decades, in part because of its presumed benefit for global warming compared to other fossil fuels, very little is known about the GHG footprint of unconventional gas. Here, we define the GHG footprint as the total GHG emissions from developing and using the gas, expressed as equivalents of carbon dioxide, per unit of energy obtained during combustion. The GHG footprint of shale gas has received little study or scrutiny, although many have voiced concern. The National Research Council (2009) noted emissions from shale-gas extraction may be greater than from conventional gas. The Council of Scientific Society Presidents (2010) wrote to President Obama, warning that some potential energy bridges such as shale gas have received insufficient analysis and may aggravate rather than mitigate global warming. And in late 2010, the U.S. Environmental Protection Agency issued a report concluding that fugitive emissions of methane from unconventional gas may be far greater than for conventional gas (EPA 2010).

Fugitive emissions of methane are of particular concern. Methane is the major component of natural gas and a powerful greenhouse gas. As such, small leakages are important. Recent modeling indicates methane has an even greater global warming potential than previously believed, when the indirect effects of methane on atmospheric aerosols are considered (Shindell et al. 2009). The global methane budget is poorly constrained, with multiple sources and sinks all having large uncertainties. The radiocarbon content of atmospheric methane suggests fossil fuels may be a far larger source of atmospheric methane than generally thought (Lassey et al. 2007).

The GHG footprint of shale gas consists of the direct emissions of CO_2 from end-use consumption, indirect emissions of CO_2 from fossil fuels used to extract, develop, and transport the gas, and methane fugitive emissions and venting. Despite the high level of industrial activity involved in developing shale gas, the indirect emissions of CO_2 are relatively small compared to those from the direct combustion of the fuel: 1 to 1.5 g C MJ^{-1} (Santoro et al. 2011) vs 15 g C MJ^{-1} for direct emissions (Hayhoe et al. 2002). Indirect emissions from shale gas are estimated to be only 0.04 to 0.45 g C MJ^{-1} greater than those for conventional gas (Wood et al. 2011). Thus, for both conventional and shale gas, the GHG footprint is dominated by the direct CO_2 emissions and fugitive methane emissions. Here we present estimates for methane emissions as contributors to the GHG footprint of shale gas compared to conventional gas.

Our analysis uses the most recently available data, relying particularly on a technical background document on GHG emissions from the oil and gas industry (EPA 2010) and materials discussed in that report, and a report on natural gas losses on federal lands from the General Accountability Office (GAO 2010). The

EPA (2010) report is the first update on emission factors by the agency since 1996 (Harrison et al. 1996). The earlier report served as the basis for the national GHG inventory for the past decade. However, that study was not based on random sampling or a comprehensive assessment of actual industry practices, but rather only analyzed facilities of companies that voluntarily participated (Kirchgessner et al. 1997). The new EPA (2010) report notes that the 1996 "study was conducted at a time when methane emissions were not a significant concern in the discussion about GHG emissions" and that emission factors from the 1996 report "are outdated and potentially understated for some emissions sources." Indeed, emission factors presented in EPA (2010) are much higher, by orders of magnitude for some sources.

1 Fugitive methane emissions during well completion

Shale gas is extracted by high-volume hydraulic fracturing. Large volumes of water are forced under pressure into the shale to fracture and re-fracture the rock to boost gas flow. A significant amount of this water returns to the surface as flow-back within the first few days to weeks after injection and is accompanied by large quantities of methane (EPA 2010). The amount of methane is far more than could be dissolved in the flow-back fluids, reflecting a mixture of fracture-return fluids and methane gas. We have compiled data from 2 shale gas formations and 3 tight-sand gas formations in the U.S. Between 0.6% and 3.2% of the life-time production of gas from wells is emitted as methane during the flow-back period (Table 1). We include tight-sand formations since flow-back emissions and the patterns of gas production over time are similar to those for shale (EPA 2010). Note that the rate of methane emitted during flow-back (column B in Table 1) correlates well to the initial production rate for the well following completion (column C in Table 1). Although the data are limited, the variation across the basins seems reasonable: the highest methane emissions during flow-back were in the Haynesville, where initial pressures and initial production were very high, and the lowest emissions were in the Uinta, where the flow-back period was the shortest and initial production following well completion was low. However, we note that the data used in Table 1 are not well documented, with many values based on PowerPoint slides from EPA-sponsored workshops. For this paper, we therefore choose to represent gas losses from flow-back fluids as the mean value from Table 1: 1.6%.

More methane is emitted during "drill-out," the stage in developing unconventional gas in which the plugs set to separate fracturing stages are drilled out to release gas for production. EPA (2007) estimates drill-out emissions at 142 × 10^3 to 425 × 10^3 m^3 per well. Using the mean drill-out emissions estimate of 280 × 10^3 m^3 (EPA 2007) and the mean life-time gas production for the 5 formations in Table 1 (85 × 10^6 m^3), we estimate that 0.33% of the total life-time production of wells is emitted as methane during the drill-out stage. If we instead use the average life-time production for a larger set of data on 12 formations (Wood et al. 2011), 45 × 10^6 m^3, we estimate a percentage emission of 0.62%. More effort is needed to determine drill-out emissions on individual formation. Meanwhile, in this paper we use the conservative estimate of 0.33% for drill-out emissions.

Combining losses associated with flow-back fluids (1.6%) and drill out (0.33%), we estimate that 1.9% of the total production of gas from an unconventional shale-gas

Table 1 Methane emissions during the flow-back period following hydraulic fracturing, initial gas production rates following well completion, life-time gas production of wells, and the methane emitted during flow-back expressed as a percentage of the life-time production for five unconventional wells in the United States

	(A) Methane emitted during flow-back (10^3 m^3)[a]	(B) Methane emitted per day during flow-back (10^3 m^3 day^{-1})[b]	(C) Initial gas production at well completion (10^3 m^3 day^{-1})[c]	(D) Life-time production of well (10^6 m^3)[d]	(E) Methane emitted during flow-back as % of life-time production[e]
Haynesville (Louisiana, shale)	6,800	680	640	210	3.2
Barnett (Texas, shale)	370	41	37	35	1.1
Piceance (Colorado, tight sand)	710	79	57	55	1.3
Uinta (Utah, tight sand)	255	51	42	40	0.6
Den-Jules (Colorado, tight sand)	140	12	11	?	?

Flow-back is the return of hydraulic fracturing fluids to the surface immediately after fracturing and before well completion. For these wells, the flow-back period ranged from 5 to 12 days

[a] Haynesville: average from Eckhardt et al. (2009); Piceance: EPA (2007); Barnett: EPA (2004); Uinta: Samuels (2010); Denver-Julesburg: Bracken (2008)

[b] Calculated by dividing the total methane emitted during flow-back (column A) by the duration of flow-back. Flow-back durations were 9 days for Barnett (EPA 2004), 8 days for Piceance (EPA 2007), 5 days for Uinta (Samuels 2010), and 12 days for Denver-Julesburg (Bracken 2008); median value of 10 days for flow-back was assumed for Haynesville

[c] Haynesville: http://shale.typepad.com/haynesvilleshale/2009/07/chesapeake-energy-haynesville-shale-decline-curve.html1/7/2011 and http://oilshalegas.com/haynesvilleshalestocks.html; Barnett: http://oilshalegas.com/barnettshale.html; Piceance: Kruuskraa (2004) and Henke (2010); Uinta: http://www.epmag.com/archives/newsComments/6242.htm; Denver-Julesburg: http://www.businesswire.com/news/home/20100924005169/en/Synergy-Resources-Corporation-Reports-Initial-Production-Rates

[d] Based on averages for these basins. Haynesville: http://shale.typepad.com/haynesvilleshale/decline-curve/); Barnett: http://www.aapg.org/explorer/2002/07jul/barnett_shale.cfm and Wood et al. (2011); Piceance: Kruuskraa (2004); Uinta: http://www.epmag.com/archives/newsComments/6242.htm

[e] Calculated by dividing column (A) by column (D)

Table 2 Fugitive methane emissions associated with development of natural gas from conventional wells and from shale formations (expressed as the percentage of methane produced over the lifecycle of a well)

	Conventional gas	Shale gas
Emissions during well completion	0.01%	1.9%
Routine venting and equipment leaks at well site	0.3 to 1.9%	0.3 to 1.9%
Emissions during liquid unloading	0 to 0.26%	0 to 0.26%
Emissions during gas processing	0 to 0.19%	0 to 0.19%
Emissions during transport, storage, and distribution	1.4 to 3.6%	1.4 to 3.6%
Total emissions	1.7 to 6.0%	3.6 to 7.9%

See text for derivation of estimates and supporting information

well is emitted as methane during well completion (Table 2). Again, this estimate is uncertain but conservative.

Emissions are far lower for conventional natural gas wells during completion, since conventional wells have no flow-back and no drill out. An average of 1.04×10^3 m^3 of methane is released per well completed for conventional gas (EPA 2010), corresponding to 1.32×10^3 m^3 natural gas (assuming 78.8% methane content of the gas). In 2007, 19,819 conventional wells were completed in the US (EPA 2010), so we estimate a total national emission of 26×10^6 m^3 natural gas. The total national production of onshore conventional gas in 2007 was 384×10^9 m^3 (EIA 2010b). Therefore, we estimate the average fugitive emissions at well completion for conventional gas as 0.01% of the life-time production of a well (Table 2), three orders of magnitude less than for shale gas.

2 Routine venting and equipment leaks

After completion, some fugitive emissions continue at the well site over its lifetime. A typical well has 55 to 150 connections to equipment such as heaters, meters, dehydrators, compressors, and vapor-recovery apparatus. Many of these potentially leak, and many pressure relief valves are designed to purposefully vent gas. Emissions from pneumatic pumps and dehydrators are a major part of the leakage (GAO 2010). Once a well is completed and connected to a pipeline, the same technologies are used for both conventional and shale gas; we assume that these post-completion fugitive emissions are the same for shale and conventional gas. GAO (2010) concluded that 0.3% to 1.9% of the life-time production of a well is lost due to routine venting and equipment leaks (Table 2). Previous studies have estimated routine well-site fugitive emissions as approximately 0.5% or less (Hayhoe et al. 2002; Armendariz 2009) and 0.95% (Shires et al. 2009). Note that none of these estimates include accidents or emergency vents. Data on emissions during emergencies are not available and have never, as far as we can determine, been used in any estimate of emissions from natural gas production. Thus, our estimate of 0.3% to 1.9% leakage is conservative. As we discuss below, the 0.3% reflects use of best available technology.

Additional venting occurs during "liquid unloading." Conventional wells frequently require multiple liquid-unloading events as they mature to mitigate water intrusion as reservoir pressure drops. Though not as common, some unconventional wells may also require unloading. Empirical data from 4 gas basins indicate that 0.02

to 0.26% of total life-time production of a well is vented as methane during liquid unloading (GAO 2010). Since not all wells require unloading, we set the range at 0 to 0.26% (Table 2).

3 Processing losses

Some natural gas, whether conventional or from shale, is of sufficient quality to be "pipeline ready" without further processing. Other gas contains sufficient amounts of heavy hydrocarbons and impurities such as sulfur gases to require removal through processing before the gas is piped. Note that the quality of gas can vary even within a formation. For example, gas from the Marcellus shale in northeastern Pennsylvania needs little or no processing, while gas from southwestern Pennsylvania must be processed (NYDEC 2009). Some methane is emitted during this processing. The default EPA facility-level fugitive emission factor for gas processing indicates a loss of 0.19% of production (Shires et al. 2009). We therefore give a range of 0% (i.e. no processing, for wells that produce "pipeline ready" gas) to 0.19% of gas produced as our estimate of processing losses (Table 2). Actual measurements of processing plant emissions in Canada showed fourfold greater leakage than standard emission factors of the sort used by Shires et al. (2009) would indicate (Chambers 2004), so again, our estimates are very conservative.

4 Transport, storage, and distribution losses

Further fugitive emissions occur during transport, storage, and distribution of natural gas. Direct measurements of leakage from transmission are limited, but two studies give similar leakage rates in both the U.S. (as part of the 1996 EPA emission factor study; mean value of 0.53%; Harrison et al. 1996; Kirchgessner et al. 1997) and in Russia (0.7% mean estimate, with a range of 0.4% to 1.6%; Lelieveld et al. 2005). Direct estimates of distribution losses are even more limited, but the 1996 EPA study estimates losses at 0.35% of production (Harrison et al. 1996; Kirchgessner et al. 1997). Lelieveld et al. (2005) used the 1996 emission factors for natural gas storage and distribution together with their transmission estimates to suggest an overall average loss rate of 1.4% (range of 1.0% to 2.5%). We use this 1.4% leakage as the likely lower limit (Table 2). As noted above, the EPA 1996 emission estimates are based on limited data, and Revkin and Krauss (2009) reported "government scientists and industry officials caution that the real figure is almost certainly higher." Furthermore, the IPCC (2007) cautions that these "bottom-up" approaches for methane inventories often underestimate fluxes.

Another way to estimate pipeline leakage is to examine "lost and unaccounted for gas," e.g. the difference between the measured volume of gas at the wellhead and that actually purchased and used by consumers. At the global scale, this method has estimated pipeline leakage at 2.5% to 10% (Crutzen 1987; Cicerone and Oremland 1988; Hayhoe et al. 2002), although the higher value reflects poorly maintained pipelines in Russia during the Soviet collapse, and leakages in Russia are now far less (Lelieveld et al. 2005; Reshetnikov et al. 2000). Kirchgessner et al. (1997) argue against this approach, stating it is "subject to numerous errors including gas theft, variations in temperature and pressure, billing cycle differences, and meter inaccuracies.” With the exception of theft, however, errors should be randomly distributed and should not bias the leakage estimate high or low. Few recent data on lost and unaccounted gas are publicly available, but statewide data for Texas averaged 2.3% in 2000 and 4.9% in 2007 (Percival 2010). In 2007, the State of Texas passed new legislation to regulate lost and unaccounted for gas; the legislation originally proposed a 5% hard cap which was dropped in the face of industry opposition (Liu 2008; Percival 2010). We take the mean of the 2000 and 2007 Texas data for missing and unaccounted gas (3.6%) as the upper limit of downstream losses (Table 2), assuming that the higher value for 2007 and lower value for 2000 may potentially reflect random variation in billing cycle differences. We believe this is a conservative upper limit, particularly given the industry resistance to a 5% hard cap.

Our conservative estimate of 1.4% to 3.6% leakage of gas during transmission, storage, and distribution is remarkably similar to the 2.5% “best estimate” used by Hayhoe et al. (2002). They considered the possible range as 0.2% and 10%.

5 Contribution of methane emissions to the GHG footprints of shale gas and conventional gas

Summing all estimated losses, we calculate that during the life cycle of an average shale-gas well, 3.6 to 7.9% of the total production of the well is emitted to the atmosphere as methane (Table 2). This is at least 30% more and perhaps more than twice as great as the life-cycle methane emissions we estimate for conventional gas, 1.7% to 6%. Methane is a far more potent GHG than is CO_2, but methane also has a tenfold shorter residence time in the atmosphere, so its effect on global warming attenuates more rapidly (IPCC 2007). Consequently, to compare the global warming potential of methane and CO_2 requires a specific time horizon. We follow Lelieveld et al. (2005) and present analyses for both 20-year and 100-year time horizons. Though the 100-year horizon is commonly used, we agree with Nisbet et al. (2000) that the 20-year horizon is critical, given the need to reduce global warming in coming decades (IPCC 2007). We use recently modeled values for the global warming potential of methane compared to CO_2: 105 and 33 on a mass-to-mass basis for 20 and 100 years, respectively, with an uncertainty of plus or minus 23% (Shindell et al. 2009). These are somewhat higher than those presented in the 4th assessment report of the IPCC (2007), but better account for the interaction of methane with aerosols. Note that carbon-trading markets use a lower global-warming potential yet of only 21 on the 100-year horizon, but this is based on the 2nd IPCC (1995) assessment, which is clearly out of date on this topic. See Electronic Supplemental Materials for the methodology for calculating the effect of methane on GHG in terms of CO_2 equivalents.

Methane dominates the GHG footprint for shale gas on the 20-year time horizon, contributing 1.4- to 3-times more than does direct CO_2 emission (Fig. 1a). At this time scale, the GHG footprint for shale gas is 22% to 43% greater than that for conventional gas. When viewed at a time 100 years after the emissions, methane emissions still contribute significantly to the GHG footprints, but the effect is diminished by the relatively short residence time of methane in the atmosphere. On this time frame, the GHG footprint for shale gas is 14% to 19% greater than that for conventional gas (Fig. 1b).

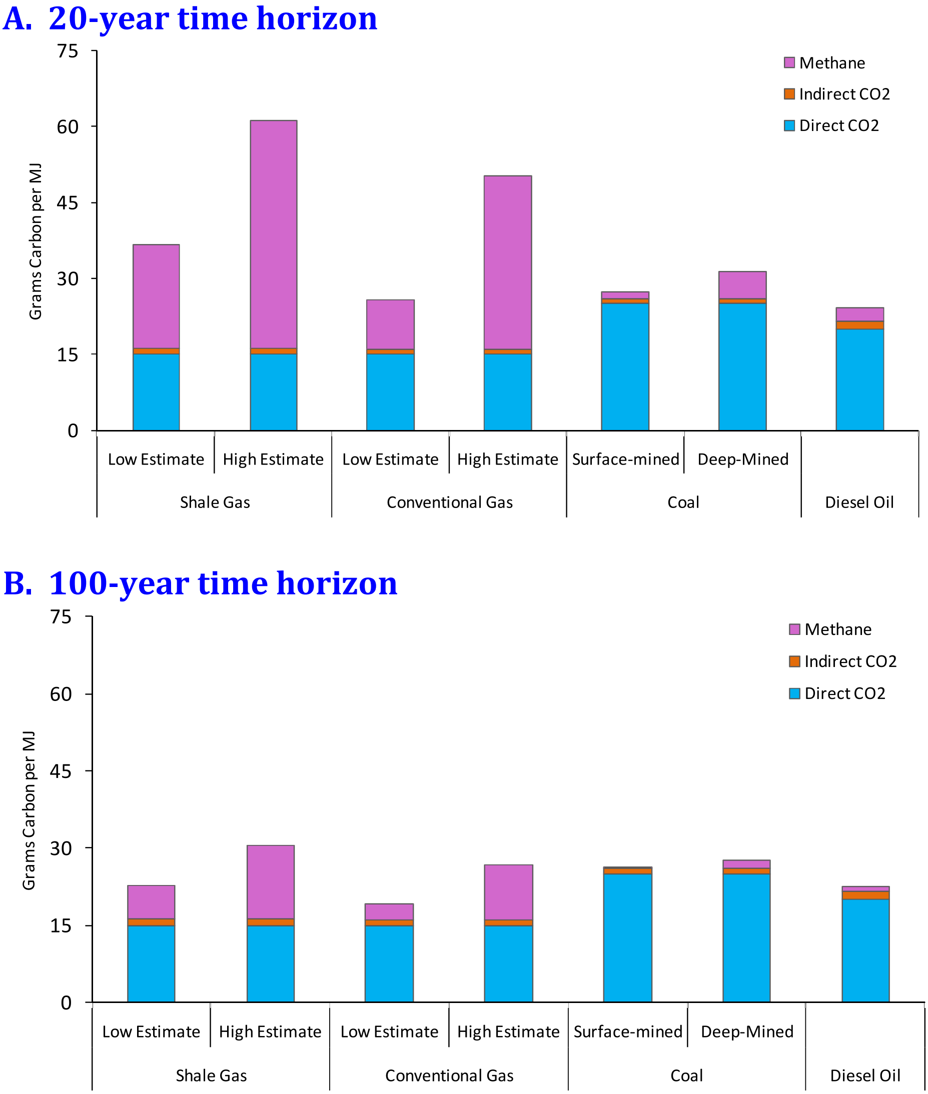

Fig. 1 Comparison of greenhouse gas emissions from shale gas with low and high estimates of fugitive methane emissions, conventional natural gas with low and high estimates of fugitive methane emissions, surface-mined coal, deep-mined coal, and diesel oil. **a** is for a 20-year time horizon, and **b** is for a 100-year time horizon. Estimates include direct emissions of CO_2 during combustion (*blue bars*), indirect emissions of CO_2 necessary to develop and use the energy source (*red bars*), and fugitive emissions of methane, converted to equivalent value of CO_2 as described in the text (*pink bars*). Emissions are normalized to the quantity of energy released at the time of combustion. The conversion of methane to CO_2 equivalents is based on global warming potentials from Shindell et al. (2009) that include both direct and indirect influences of methane on aerosols. Mean values from Shindell et al. (2009) are used here. Shindell et al. (2009) present an uncertainty in these mean values of plus or minus 23%, which is not included in this figure

6 Shale gas versus other fossil fuels

Considering the 20-year horizon, the GHG footprint for shale gas is at least 20% greater than and perhaps more than twice as great as that for coal when expressed per quantity of energy available during combustion (Fig. 1a; see Electronic Supplemental Materials for derivation of the estimates for diesel oil and coal). Over the 100-year frame, the GHG footprint is comparable to that for coal: the low-end shale-gas emissions are 18% lower than deep-mined coal, and the high-end shale-gas emissions are 15% greater than surface-mined coal emissions (Fig. 1b). For the 20 year horizon, the GHG footprint of shale gas is at least 50% greater than for oil, and perhaps 2.5-times greater. At the 100-year time scale, the footprint for shale gas is similar to or 35% greater than for oil.

We know of no other estimates for the GHG footprint of shale gas in the peer-reviewed literature. However, we can compare our estimates for conventional gas with three previous peer-reviewed studies on the GHG emissions of conventional natural gas and coal: Hayhoe et al. (2002), Lelieveld et al. (2005), and Jamarillo et al. (2007). All concluded that GHG emissions for conventional gas are less than for coal, when considering the contribution of methane over 100 years. In contrast, our analysis indicates that conventional gas has little or no advantage over coal even over the 100-year time period (Fig. 1b). Our estimates for conventional-gas methane emissions are in the range of those in Hayhoe et al. (2002) but are higher than those in Lelieveld et al. (2005) and Jamarillo et al. (2007) who used 1996 EPA emission factors now known to be too low (EPA 2010). To evaluate the effect of methane, all three of these studies also used global warming potentials now believed to be too low (Shindell et al. 2009). Still, Hayhoe et al. (2002) concluded that under many of the scenarios evaluated, a switch from coal to conventional natural gas could aggravate global warming on time scales of up to several decades. Even with the lower global warming potential value, Lelieveld et al. (2005) concluded that natural gas has a greater GHG footprint than oil if methane emissions exceeded 3.1% and worse than coal if the emissions exceeded 5.6% on the 20-year time scale. They used a methane global warming potential value for methane from IPCC (1995) that is only 57% of the new value from Shindell et al. (2009), suggesting that in fact methane emissions of only 2% to 3% make the GHG footprint of conventional gas worse than oil and coal. Our estimates for fugitive shale-gas emissions are 3.6 to 7.9%.

Our analysis does not consider the efficiency of final use. If fuels are used to generate electricity, natural gas gains some advantage over coal because of greater efficiencies of generation (see Electronic Supplemental Materials). However, this does not greatly affect our overall conclusion: the GHG footprint of shale gas approaches or exceeds coal even when used to generate electricity (Table in Electronic Supplemental Materials). Further, shale-gas is promoted for other uses, including as a heating and transportation fuel, where there is little evidence that efficiencies are superior to diesel oil.

7 Can methane emissions be reduced?

The EPA estimates that ’green’ technologies can reduce gas-industry methane emissions by 40% (GAO 2010). For instance, liquid-unloading emissions can be greatly reduced with plunger lifts (EPA 2006; GAO 2010); industry reports a 99% venting reduction in the San Juan basin with the use of smart-automated plunger lifts (GAO 2010). Use of flash-tank separators or vapor recovery units can reduce dehydrator emissions by 90% (Fernandez et al. 2005). Note, however, that our lower range of estimates for 3 out of the 5 sources as shown in Table 2 already reflect the use of best technology: 0.3% lower-end estimate for routine venting and leaks at well sites (GAO 2010), 0% lower-end estimate for emissions during liquid unloading, and 0% during processing.

Methane emissions during the flow-back period in theory can be reduced by up to 90% through Reduced Emission Completions technologies, or REC (EPA 2010). However, REC technologies require that pipelines to the well are in place prior to completion, which is not always possible in emerging development areas. In any event, these technologies are currently not in wide use (EPA 2010).

If emissions during transmission, storage, and distribution are at the high end of our estimate (3.6%; Table 2), these could probably be reduced through use of better storage tanks and compressors and through improved monitoring for leaks. Industry has shown little interest in making the investments needed to reduce these emission sources, however (Percival 2010).

Better regulation can help push industry towards reduced emissions. In reconciling a wide range of emissions, the GAO (2010) noted that lower emissions in the Piceance basin in Colorado relative to the Uinta basin in Utah are largely due to a higher use of low-bleed pneumatics in the former due to stricter state regulations.

8 Conclusions and implications

The GHG footprint of shale gas is significantly larger than that from conventional gas, due to methane emissions with flow-back fluids and from drill out of wells during well completion. Routine production and downstream methane emissions are also large, but are the same for conventional and shale gas. Our estimates for these routine and downstream methane emission sources are within the range of those reported by most other peer-reviewed publications inventories (Hayhoe et al. 2002; Lelieveld et al. 2005). Despite this broad agreement, the uncertainty in the magnitude of fugitive emissions is large. Given the importance of methane in global warming, these emissions deserve far greater study than has occurred in the past. We urge both more direct measurements and refined accounting to better quantify lost and unaccounted for gas.

The large GHG footprint of shale gas undercuts the logic of its use as a bridging fuel over coming decades, if the goal is to reduce global warming. We do not intend that our study be used to justify the continued use of either oil or coal, but rather to demonstrate that substituting shale gas for these other fossil fuels may not have the desired effect of mitigating climate warming.

Finally, we note that carbon-trading markets at present under-value the greenhouse warming consequences of methane, by focusing on a 100-year time horizon and by using out-of-date global warming potentials for methane. This should be corrected, and the full GHG footprint of unconventional gas should be used in planning for alternative energy futures that adequately consider global climate change.

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COMMENT

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CULTURE Martin Kemp muses on 15 years of artists in lab schemes **p.278**

CORRESPONDENCE Pros and cons of 24/7 working stir up debate **p.280**

D. ACKER/BLOOMBERG VIA GETTY

A drilling operation in Bradford County, Pennsylvania: one of the many places where shale rocks are fractured to release oil and gas.

Should fracking stop?

Extracting gas from shale increases the availability of this resource, but the health and environmental risks may be too high.

POINT

Yes, it's too high risk

Natural gas extracted from shale comes at too great a cost to the environment, say Robert W. Howarth *and* Anthony Ingraffea.

Natural gas from shale is widely promoted as clean compared with oil and coal, a 'win–win' fuel that can lessen emissions while still supplying abundant fossil energy over coming decades until a switch to renewable energy sources is made. But shale gas isn't clean, and shouldn't be used as a bridge fuel.

Shale rock formations can contain vast amounts of natural gas (which is mostly methane). Until quite recently, most of

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COUNTERPOINT

No, it's too valuable

Fracking is crucial to global economic stability; the economic benefits outweigh the environmental risks, says Terry Engelder.

After a career in geological research on one of the world's largest gas supplies, I am a born-again 'cornucopian'. I believe that there is enough domestic gas to meet our needs for the foreseeable future thanks to technological advances in hydraulic fracturing. According to IHS, a business-information company in Douglas County, Colorado, the estimated recoverable gas from US shale source rocks using fracking is about 42 trillion cubic metres, almost

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POINT: FRACKING: TOO HIGH RISK ▶

this gas was not economically obtainable, because shale is far less permeable than the rock formations exploited for conventional gas. Over the past decade or so, two new technologies have combined to allow extraction of shale gas: 'high-volume, slick-water hydraulic fracturing' (also known as 'fracking'), in which high-pressure water with additives is used to increase fissures in the rock; and precision drilling of wells that can follow the contour of a shale layer closely for 3 kilometres or more at depths of more than 2 kilometres (see 'Fracking for fuel'). Industry first experimented with these two technologies in Texas about 15 years ago. Significant shale-gas production in other states, including Arkansas, Pennsylvania and Louisiana, began only in 2007–09. Outside North America, only a handful of shale-gas wells have been drilled.

Industry sources claim that they have used fracking to produce more than 1 million oil and natural gas wells since the late 1940s. However, less than 2% of the well fractures since the 1940s have used the high-volume technology necessary to get gas from shale, almost all of these in the past ten years. This approach is far bigger and riskier than the conventional fracking of earlier years. An average of 20 million litres of water are forced under pressure into each well, combined with large volumes of sand or other materials to help keep the fissures open, and 200,000 litres of acids, biocides, scale inhibitors, friction reducers and surfactants. The fracking of a conventional well uses at most 1–2% of the volume of water used to extract shale gas[1].

Many of the fracking additives are toxic, carcinogenic or mutagenic. Many are kept secret. In the United States, such secrecy has been abetted by the 2005 'Halliburton loophole' (named after an energy company headquartered in Houston, Texas), which exempts fracking from many of the nation's major federal environmental-protection laws, including the Safe Drinking Water Act. In a 2-hectare site, up to 16 wells can be drilled, cumulatively servicing an area of up to 1.5 square kilometres, and using 300 million litres or more of water and additives. Around one-fifth of the fracking fluid flows back up the well to the surface in the first two weeks, with more continuing to flow out over the lifetime of the well. Fracking also extracts natural salts, heavy metals, hydrocarbons and radioactive materials from the shale, posing risks to ecosystems and public health when these return to the surface. This flowback is collected in open pits or large tanks until treated, recycled or disposed of.

Because shale-gas development is so new, scientific information on the environmental costs is scarce. Only this year have studies begun to appear in peer-reviewed journals, and these give reason for pause. We call for a moratorium on shale-gas development to allow for better study of the cumulative risks to water quality, air quality and global climate. Only with such comprehensive knowledge can appropriate regulatory frameworks be developed.

We have analysed the well-to-consumer lifecycle greenhouse-gas footprint of shale gas when used for heat generation (its main use), compared with conventional gas and other fossil fuels — the first estimate in the peer-reviewed literature[2]. Methane is a major component of this footprint, and we estimate that 3.6–7.9% of the lifetime production of a shale gas well (compared with 1.7–6% for conventional gas wells) is vented or leaked to the atmosphere from the well head, pipelines and storage facilities. In addition, carbon dioxide is released both directly through the burning of the gas for heat, and to a lesser extent indirectly through the process of developing the resource.

Methane is a potent greenhouse gas, so even small emissions matter. Over a 20-year time period, the greenhouse-gas footprint of shale gas is worse than that for coal or oil (see 'A daunting climate footprint'). The influence of methane is lessened over longer time scales, because methane does not stay in the atmosphere as long as carbon dioxide. Still, over 100 years, the footprint of shale gas remains comparable to that of oil or coal.

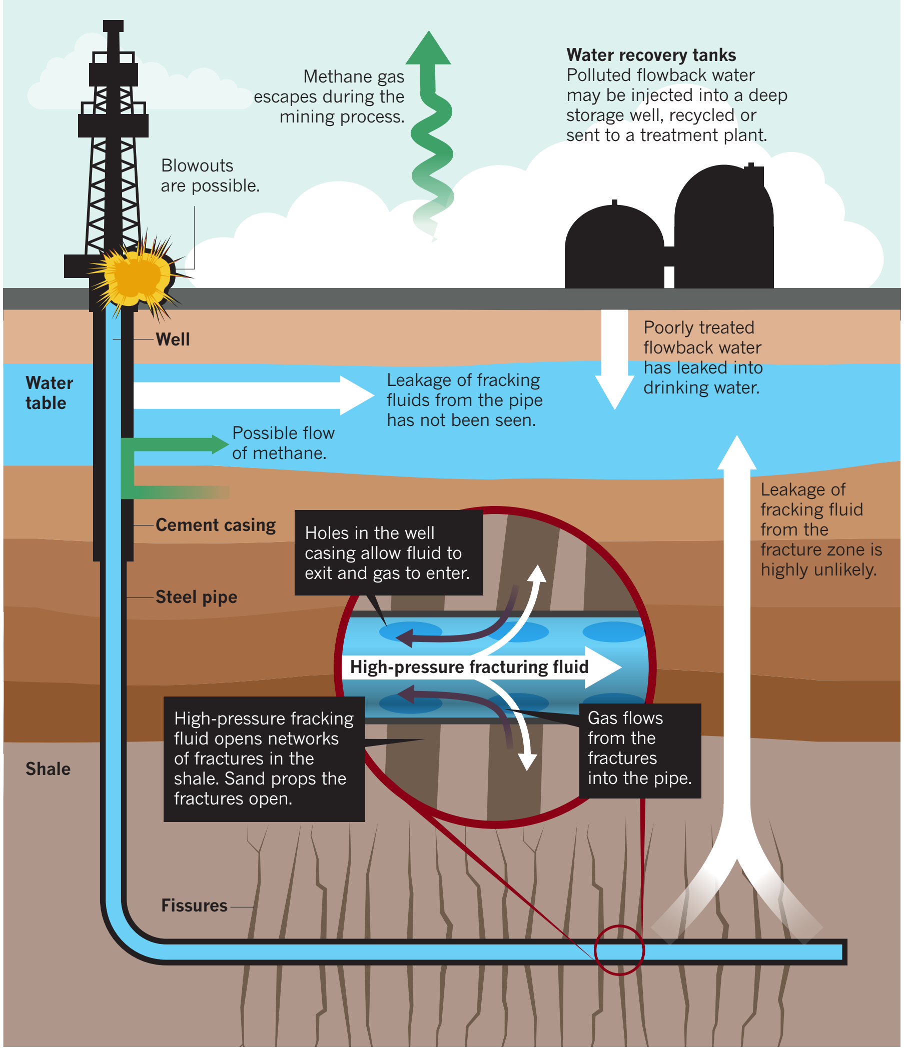
When used to produce electricity rather than heat, the greater efficiency of gas plants compared with coal plants slightly lessens the footprint of shale gas[3]. Even then, the total greenhouse-gas footprint from shale gas exceed those of coal at timescales of less than about 50 years.

Methane venting and leakage can be decreased by upgrading old pipelines and storage systems, and by applying better technology for capturing gas in the 2-week flowback period after fracking. But current economic incentives are not sufficient to drive such improvements; stringent regulation will be required. In July, the US Environmental Protection Agency released a draft rule that would push industry to reduce at least some methane emissions, in part focusing on post-fracking flowback. Nonetheless, our analysis[2] indicates that the greenhouse-gas footprint of shale gas is likely to remain large.

Another peer-reviewed study looked at

FRACKING FOR FUEL

Hydraulic fracturing is used to access oil and gas resources that are locked in non-porous rocks.



private water wells near fracking sites[4]. It found that about 75% of wells sampled within 1 kilometre of gas drilling in the Marcellus shale in Pennsylvania were contaminated with methane from the deep shale formations. Isotopic fingerprinting of the methane indicated that deep shale was the source of contamination, rather than biologically derived methane, which was present at much lower concentrations in water wells at greater distances from gas wells. The study found no fracking fluids in any of the drinking-water wells examined. This is good news, because these fluids contain hazardous materials, and methane itself is not toxic. However, methane poses a high risk of explosion at the levels found, and it suggests a potential for other gaseous substances in the shale to migrate with the methane and contaminate water wells over time.

Have fracking-return fluids contaminated drinking water? Yes, although the evidence is not as strong as for methane contamination, and none of the data has yet appeared in the peer-reviewed literature (although a series of articles in *The New York Times* documents the problem, for example go.nature.com/58hxot and go.nature.com/58koj3). Contamination can happen through blowouts, surface spills from storage facilities, or improper disposal of fracking fluids. In Texas, flowback fluids are disposed of through deep injection into abandoned gas or oil wells. But such wells are not available everywhere. In New York and Pennsylvania, some of the waste is treated in municipal sewage plants that weren't designed to handle these toxic and radioactive wastes. Subsequently, there has been contamination of tributaries of the Ohio River with barium, strontium and bromides from municipal wastewater treatment plants receiving fracking wastes[5]. This contamination apparently led to the formation of dangerous brominated hydrocarbons in municipal drinking-water supplies that relied on these surface waters, owing to interaction of the contaminants with organic matter during the chlorination process.

Shale-gas development — which uses huge diesel pumps to inject the water — also creates local air pollution, often at dangerous levels. Volatile hydrocarbons such as benzene (which occurs naturally in shale, and is a commonly used fracking additive) are one major concern. The state of Texas reports benzene concentrations in air in the Barnett shale area that sometimes exceed acute toxicity standards[6], and although the concentrations observed in the Marcellus shale area in Pennsylvania are lower[7] (with only 2,349 wells drilled at the time these air contaminants were reported, out of an expected total of 100,000), they are high enough to pose a risk of cancer from chronic exposure[8]. Emissions from drills, compressors, trucks and other machinery can lead to very high levels of ground-level ozone, as documented in parts of Colorado that had not experienced severe air pollution before shale-gas development[9].

> ***"Have fracking-return fluids contaminated drinking water? Yes."***

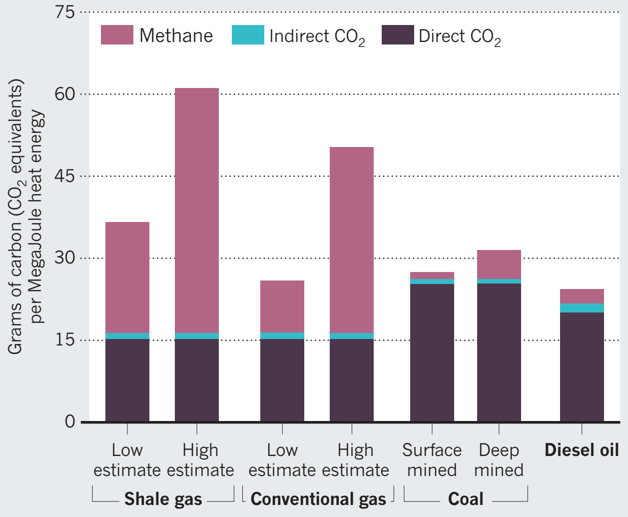
UNPROFITABLE PROGRESS

The argument for continuing shale-gas exploitation often hinges on the presumed gigantic size of the resource. But this may be exaggerated. The Energy Information Administration of the US Department of Energy estimates that 45% of US gas supply will come from shale gas by 2035 (with the vast majority of this replacing conventional gas, which has a lower greenhouse-gas footprint). Other gas industry observers are even more bullish. However, David Hughes, a geoscientist with more than 30 years experience with the Canadian Geological Survey, concludes in his report for the Post Carbon Institute, a non-profit group headquartered in Santa Rosa, California, that forecasts are likely to be overstated, perhaps greatly so[3]. Last month, the US Geological Survey released a new estimate of the amount of gas in the Marcellus shale formation (the largest shale-gas formation in the United States), concluding that the Department of Energy has over-estimated the resource by some five-fold[10].

Shale gas may not be profitable at current prices, in part because production rates for shale-gas wells decline far more quickly than for conventional wells. Although very large resources undoubtedly exist in shale reservoirs, an unprecedented rate of well drilling and fracking would be required to meet the Department of Energy's projections, which might not be economic[3]. If so, the recent enthusiasm over shale gas could soon collapse, like the dot-com bubble.

A DAUNTING CLIMATE FOOTPRINT

Over 20 years, shale gas is likely to have a greater greenhouse effect than conventional gas or other fossil fuels.



SOURCE: REF. 2

Meanwhile, shale gas competes for investment with green energy technologies, slowing their development and distracting politicians and the public from developing a long-term sustainable energy policy.

With time, perhaps engineers can develop more appropriate ways to handle fracking-fluid return wastes, and perhaps the technology can be made more sustainable and less polluting in other ways. Meanwhile, the gas should remain safely in the shale, while society uses energy more efficiently and develops renewable energy sources more aggressively. ■

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COUNTERPOINT: FRACKING: TOO VALUABLE ▶

equal to the total conventional gas discovered in the United States over the past 150 years, and equivalent to about 65 times the current US annual consumption. During the past three years, about 50 billion barrels of additional recoverable oil have been found in shale oil deposits — more than 20% of the total conventional recoverable US oil resource. These ‘tight’ oil resources, which also require fracking to access, could generate 3 million barrels a day by 2020, offsetting one-third of current oil imports. International data aren’t as well known, but the effect of fracking on global energy production will be huge (see ‘Global gas reserves’).

Global warming is a serious issue that fracking-related gas production can help to alleviate. In a world in which productivity is closely linked to energy expenditure, fracking will be vital to global economic stability until renewable or nuclear energy carry more of the workload. But these technologies face persistent problems of intermittency and lack of power density or waste disposal. Mankind’s inexorable march towards 9 billion people will require a broad portfolio of energy resources, which can be gained only with breakthroughs such as fracking. Such breakthroughs should be promoted by policy that benefits the economy yet reduces overall greenhouse-gas emissions. Replacing coal with natural gas in power plants, for example, reduces the plants’ greenhouse emissions by up to 50% (ref. 1).

At present, fracking accounts for 50% of locally produced natural gas (see ‘US natural-gas production set to explode’) and 33% of local petroleum. The gas industry in America accounts for US$385 billion in direct economic activity and nearly 3 million jobs. Because gas wells have notoriously steep production declines, stable supplies depend on a steady rate of new well completions. A moratorium on new wells would have an immediate and harsh effect on the US economy that would trigger a global ripple.

Global warming aside, there is no compelling environmental reason to ban hydraulic fracturing. There are environmental risks, but these can be managed through existing, and rapidly improving, technologies and regulations. It might be nice to have moratoria after each breakthrough to study the consequences (including the disposal of old batteries or radioactive waste), but because energy expenditure and economic health are so closely linked, global moratoria are not practical.

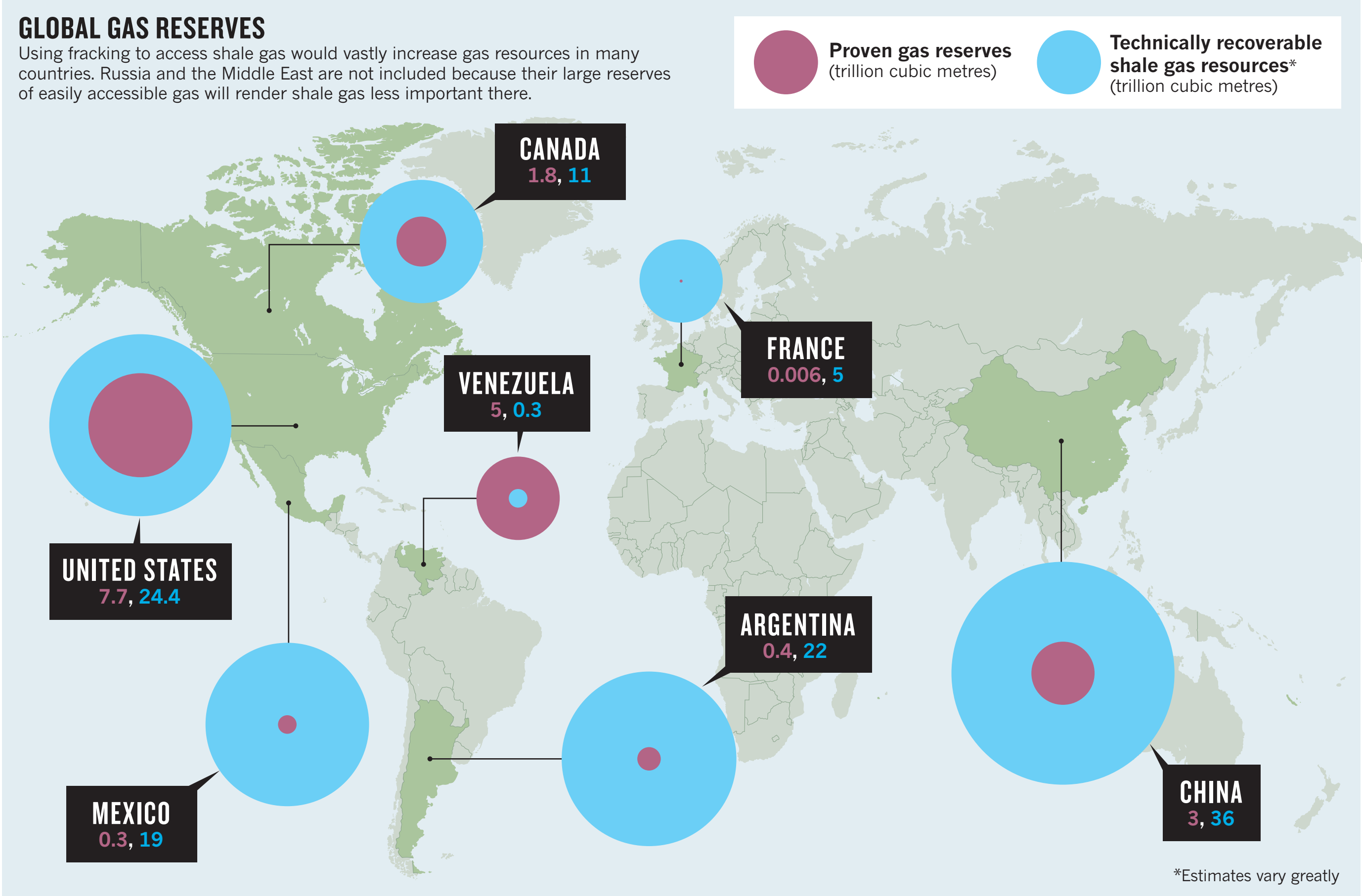
The gains in employment, economics and national security, combined with the potential to reduce global greenhouse-gas emissions if natural gas is managed properly, make a compelling case.

NO NEED FOR PANIC

I grew up with the sights, sounds and smells of the Bradford oil fields in New York state. My parents’ small farm was over a small oil pool, with fumes from unplugged wells in the air and small oil seeps coating still waters. Before college, I worked these oil fields as a roustabout, mainly cleaning pipes and casings. Like me, most people living in such areas are not opposed to drilling, it seems. In my experience, such as during the recent hearings for the Pennsylvania Governor’s Marcellus Shale Advisory Commission, activists from non-drilling regions outnumber those from drilling regions by approximately two to one.

Modern, massive hydraulic fracturing is very different from that used decades ago. Larger pads are required to accommodate larger drill rigs, pumps and water supplies. People usually infer from this that modern techniques have a greater impact on the environment. This isn’t necessarily true. Although more water is used per well, there are far fewer wells per unit area. In the Bradford oil fields in the 1950s, a 640-acre parcel of land might have held more than 100 wells, requiring some 18 kilometres of roads, and with a lattice of surface pipelines. During the Marcellus development today, that same parcel of land is served by a single pad of five acres, with a 0.8-kilometre right-of-way for roads and pipelines.

Although ‘fracking’ has emerged as a scare term in the press,

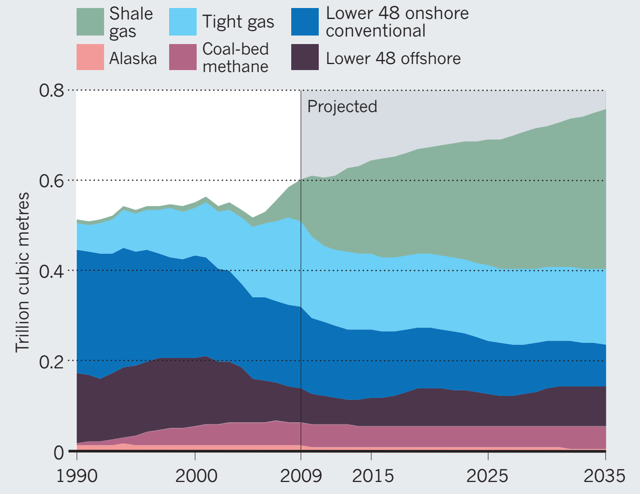


GLOBAL GAS RESERVES

Using fracking to access shale gas would vastly increase gas resources in many countries. Russia and the Middle East are not included because their large reserves of easily accessible gas will render shale gas less important there.

*Estimates vary greatly

SOURCE: EIA



US NATURAL-GAS PRODUCTION SET TO EXPLODE

Shale-gas output already matches production from offshore wells in the lower 48 states (mainland US states excluding Alaska). Gas (shale and tight) extracted by fracking is set to overtake all other sources.

SOURCE: US EIA *ANNUAL ENERGY OUTLOOK 2011*

hydraulic fracturing is not so strange or frightening. The process happens naturally: high-pressure magma, water, petroleum and gases deep inside Earth can crack rock, helping to drive plate tectonics, rock metamorphism and the recycling of carbon dioxide between the mantle and the atmosphere.

Oil and gas have their origins in muds rich with organic matter in low-oxygen water bodies. Over millions of years, some of these deposits were buried and 'cooked' in the deep Earth, turning the organic matter into fossil fuel and the mud to shale rocks. In many areas, natural hydraulic fracturing allowed a large portion of oil and gas to escape from the dense, impermeable shale and migrate into neighbouring, more porous rocks. Some of this fossil fuel was trapped by cap rock, creating the conventional reserves that mankind has long tapped. The groundwater above areas that host such conventional deposits naturally contains methane, thanks to natural hydraulic fracturing of the rock and the upward seeping of gas into the water table over long time periods.

More than 96% of all oil and gas has been released from its original source rocks; industrial hydraulic fracturing aims to mimic nature to access the rest. As in nature, industrial fracking can be done with a wide variety of gases and liquids. Nitrogen can be used to open cracks in the shale, for example. But this is inefficient, because of the energy lost by natural decompression of the nitrogen gas. Water is more efficient, because very little energy is wasted in decompression. Sand is added to prop open the cracks, and compounds such as surface-tension reducers are added to improve gas recovery.

UNDER CONTROL

Two main environmental concerns are water use and water contamination. Millions of gallons of water are required to stimulate a well. In Pennsylvania, high rainfall means that water is abundant, and regulations ensure that operators stockpile rainwater during the wet season to use during drier months (thus the injection of massive volumes of water in the Bradford oil fields for secondary recovery of oil, once the well pressure has fallen, flew under the radar of environmentalists for half a century). Obtaining adequate water for industrial fracking in dry regions such as the Middle East and western China is a local concern, but is no reason for a global moratorium.

Press reports often repeat strident concerns about the chemicals added to fracking fluids. But many of these compounds are relatively benign. One commonly used additive is similar to simethicone, which is also used in antacids to reduce surface tension and turn small bubbles in the stomach into larger ones that can move along more easily. Many of the industrial additives are common in household products. Material safety data sheets for these additives are required by US regulation. Industry discloses additives on a website called FracFocus.org, run by state regulators.

Some people have expressed worries that fracking fluids might migrate more than 2 kilometres upwards from the cracked shale into groundwater. The Ground Water Protection Council, a non-profit national association of state groundwater and underground-injection control agencies headquartered in Oklahoma City, has found no instance in which injected fluid contaminated groundwater from below[2]. This makes sense: water cannot flow this distance uphill in timescales that matter. This is the premise by which deep disposal wells, used to hold toxic waste worldwide, are considered safe. During gas production, the pressure of methane is reduced: this promotes downward, not upward flow of these fluids.

Gas shale contains a number of materials that are carried back up the pipe to the surface in flowback water, including salts of barium and radioactive isotopes, that might be harmful in concentrated form. According to a recent *New York Times* analysis, these elements can be above the US Environmental Protection Agency's sanctioned background concentrations in some flowback tanks. Industry is moving towards complete recycling of these fluids so this should be of less concern to the public. However, production water will continue to flow to the surface in modest volumes throughout the life of a well; this water needs to be, and currently is, treated to ensure safe disposal.

The real risk of water contamination comes from these flowback fluids leaking into streams or seeping down into groundwater after reaching the surface. This can be caused by leaky wellheads, holding tanks or blowouts. Wellheads are made sufficiently safe to prevent this eventuality; holding tanks can be made secure; and blowouts, while problematic, are like all accidents caused by human error — an unpredictable risk with which society lives.

Although methane coming up to the surface within the steel well pipe cannot escape into the surrounding rocks or groundwater, it is possible that the cement seal between the well and the bedrock might allow methane from shallow sandstone layers (rather than the reservoir deep below) to seep up into groundwater. Methane is a tasteless and odourless component of groundwater that can be consumed without ill effect when dissolved. It is not a poison. Long before gas-shale drilling, regulators warned that enclosed spaces, such as houses, should be properly ventilated in areas with naturally occurring methane in groundwater.

> ***"With hydraulic fracturing, as in many cases, fear levels exceed the evidence."***

An alarm has been sounded too about the effect of escaped methane on global warming. The good news is that methane has a very short half-life in the atmosphere: carbon dioxide emitted during the building of the first Sumerian cities is still affecting our climate, whereas escaped methane from the fracturing of the Barnett shale in 1997 is more than half gone. Industry can and should take steps to reduce air emissions, by capturing or flaring methane and converting motors and compressors from diesel to natural gas.

Risk perception is ultimately subjective: facts are all too easily combined with emotional responses. With hydraulic fracturing, as in many cases, fear levels exceed the evidence. ■

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Examining the feasibility of converting New York State's all-purpose energy infrastructure to one using wind, water, and sunlight

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HIGHLIGHTS

- ▶ New York State's all-purpose energy can be derived from wind, water, and sunlight.
- ▶ The conversion reduces NYS end-use power demand by ~37%.
- ▶ The plan creates more jobs than lost since most energy will be from in state.
- ▶ The plan creates long-term energy price stability since fuel costs will be zero.
- ▶ The plan decreases air pollution deaths 4000/yr ($33 billion/yr or 3% of NYS GDP).

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ABSTRACT

This study analyzes a plan to convert New York State's (NYS's) all-purpose (for electricity, transportation, heating/cooling, and industry) energy infrastructure to one derived entirely from wind, water, and sunlight (WWS) generating electricity and electrolytic hydrogen. Under the plan, NYS's 2030 all-purpose end-use power would be provided by 10% onshore wind (4020 5-MW turbines), 40% offshore wind (12,700 5-MW turbines), 10% concentrated solar (387 100-MW plants), 10% solar-PV plants (828 50-MW plants), 6% residential rooftop PV (~5 million 5-kW systems), 12% commercial/government rooftop PV (~500,000 100-kW systems), 5% geothermal (36 100-MW plants), 0.5% wave (1910 0.75-MW devices), 1% tidal (2600 1-MW turbines), and 5.5% hydroelectric (6.6 1300-MW plants, of which 89% exist). The conversion would reduce NYS's end-use power demand ~37% and stabilize energy prices since fuel costs would be zero. It would create more jobs than lost because nearly all NYS energy would now be produced in-state. NYS air pollution mortality and its costs would decline by ~4000 (1200–7600) deaths/yr, and $33 (10–76) billion/yr (3% of 2010 NYS GDP), respectively, alone repaying the 271 GW installed power needed within ~17 years, before accounting for electricity sales. NYS's own emission decreases would reduce 2050 U.S. climate costs by ~$3.2 billion/yr.

1. Introduction

This is a study to examine the technical and economic feasibility of and propose policies for converting New York State's (NYS's) energy infrastructure in all sectors to one powered by wind, water, and sunlight (WWS). The plan is a localized microcosm of that developed for the world and U.S. by Jacobson and Delucchi (2009, 2011) and Delucchi and Jacobson (2011). Recently, other plans involving different levels of energy conversion for some or multiple energy sectors have been developed at national or continental scales (e.g., Alliance for Climate Protection, 2009; Parsons-Brinckerhoff, 2009; Kemp and Wexler, 2010; Price-Waterhouse-Coopers, 2010; Beyond Zero Emissions, 2010; European Climate Foundation (ECF), 2010; European Renewable Energy Council (EREC), 2010; World Wildlife Fund, 2011).

Limited plans are currently in place in New York City (PlaNYC, 2011) and NYS (Power, 2011) to help the city and state, respectively, provide predictable and sustainable energy, improve the

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quality of life, and reduce climate-relevant emissions. NYS also has a renewable portfolio standard requiring 30% of its electric power to come from renewable sources by 2015 (NYSERDA (New York State Energy Research and Development Authority), 2012). Although current plans for NYS and other states, countries, and continents are visionary and important, the plan here goes further by proposing a long-term sustainable energy infrastructure that supplies *all* energy from wind, water, and solar power, and provides the largest possible reductions in air pollution, water pollution, and global warming impacts. This study represents the first effort to develop a plan for an individual state to provide 100% of its all-purpose energy from WWS and to calculate the number of WWS energy devices, land and ocean areas, jobs, and policies needed for such an infrastructure. It also provides new calculations of air pollution mortality and morbidity impacts and costs in NYS based on multiple years of high-resolution air quality data.

In brief, the plan requires or results in the following changes:

(1) Replace fossil-fuel electric power generators with wind turbines, solar photovoltaic (PV) plants and rooftop systems, concentrated solar power (CSP) plants, solar hot water heater systems, geothermal power plants, a few additional hydroelectric power plants, and a small number of wave and tidal devices.

(2) Replace all fossil-fuel combustion for transportation, heating and cooling, and industrial processes with electricity, hydrogen fuel cells, and a limited amount of hydrogen combustion. Battery-electric vehicles (BEVs), hydrogen fuel cell vehicles (HFCVs), and BEV–HFCV hybrids sold in NYS will replace all combustion-based passenger vehicles, trucks, buses, non-road machines, and locomotives sold in the state. Long-distance trucks will be primarily BEV-HFCV hybrids and HFCVs. Ships built in NYS will similarly run on hydrogen fuel cells and electricity. Today, hydrogen-fuel-cell ships, tractors, forklifts, buses, passenger vehicles, and trucks already exist, and electric vehicles, ferries, and non-road machinery also exist. Electricity-powered air- and ground-source heat pumps, heat exchangers, and backup electric resistance heaters will replace natural gas and oil for home heating and air conditioning. Air- and ground-source heat pump water heaters powered by electricity and solar hot water preheaters will provide hot water for homes. High-temperatures for industrial processes will be obtained with electricity and hydrogen combustion. Petroleum products may still be used for lubrication and plastics as necessary, but such products will be produced using WWS power for process energy.

(3) Reduce energy demand beyond the reductions described under (2) through energy efficiency measures. Such measures include retrofitting residential, commercial, institutional, and government buildings with better insulation, improving the energy-out/energy-in efficiency of end uses with more efficient lighting and the use of heat-exchange and filtration systems; increasing public transit and telecommuting, designing future city infrastructure to facilitate greater use of clean-energy transport; and designing new buildings to use solar energy with more daylighting, solar hot water heating, seasonal energy storage, and improved passive solar heating in winter and cooling in summer.

(4) Boost economic activity by implementing the measures above. Increase jobs in the manufacturing and installation industries and in the development of new and more efficient technologies. Reduce social costs by reducing health-related mortality and morbidity and reducing environmental damage to lakes, streams, rivers, forests, buildings, and statues resulting from air and water pollution. Reduce social costs by slowing the increase in global warming and its impacts on coastlines, agriculture, fishing, heat stress, severe weather, and air pollution (which otherwise increases with increasing temperatures). Reduce long-term macroeconomic costs by eliminating exposure to future rises in fossil fuel prices.

(5) The plan anticipates that the fraction of new electric power generators as WWS will increase starting today such that, by 2020, all new generators will be WWS generators. Existing conventional generators will be phased out over time, but by no later than 2050. Similarly, BEVs and HFCVs should be nearly the only new vehicles types sold in NYS by 2020. The growth of electric vehicles will be accompanied by a growth of electric charging stations in residences, commercial parking spaces, service stations, and highway rest stops.

(6) All new heating and cooling technologies installed by 2020 should be WWS technologies and existing technologies should be replaced over time, but by no later than 2050.

(7) To ensure reliability of the electric power grids, several methods should be used to match renewable energy supply with demand and to smooth out the variability of WWS resources. These include (A) combining geographically-dispersed WWS resources as a bundled set of resources rather than as separate resources and using hydroelectric power to fill remaining gaps; (B) using demand-response grid management to shift times of demand to match better with the timing of WWS power supply; (C) over-sizing WWS peak generation capacity to minimize the times when available WWS power is less than demand and to provide power to produce heat for air and water and hydrogen for transportation and heating when WWS power exceeds demand; (D) integrating weather forecasts into system operation to reduce reserve requirements; (E) storing energy in thermal storage media, batteries or other storage media at the site of generation or use; and (F) storing energy in electric-vehicle batteries for later extraction (vehicle-to-grid).

2. How the technologies were chosen

The WWS energy technologies chosen for the NYS plan exist and were ranked the highest among several proposed energy options for addressing pollution and public health, global warming, and energy security (Jacobson, 2009). That analysis used a combination of 11 criteria (carbon-dioxide equivalent emissions, air-pollution mortality and morbidity, resource abundance, footprint on the ground, spacing required, water consumption, effects on wildlife, thermal pollution, water chemical pollution/radioactive waste, energy supply disruption, and normal operating reliability) to evaluate each technology.

Mined natural gas and liquid biofuels are excluded from the NYS plan for the reasons given below. Jacobson and Delucchi (2011) explain why nuclear power and coal with carbon capture are also excluded.

2.1. Why not natural gas?

Natural gas is excluded for several reasons. The mining, transport, and use of conventional natural gas for electric power results in at least 60–80 times more carbon-equivalent emissions and air pollution mortality per unit electric power generated than does wind energy over a 100-year time frame. Over the 10–30 year time frame, natural gas is a greater warming agent relative to all WWS technologies and a danger to the Arctic sea ice due to its leaked methane and black carbon-flaring emissions (discussed more below). Natural gas mining, transport, and use also produce carbon monoxide, ammonia, nitrogen oxides, and organic gases.

Natural gas mining degrades land, roads, and highways and produces water pollution.

The main argument for increasing the use of natural gas has been that it is a ‘‘bridge fuel’’ between coal and renewable energy because of the belief that natural gas causes less global warming per unit electric power generated than coal. Although natural gas emits less carbon dioxide per unit electric power than coal, two factors cause natural gas to increase global warming relative to coal: higher methane emissions and less sulfur dioxide emissions per unit energy than coal.

Although significant uncertainty still exists, several studies have shown that, without considering sulfur dioxide emissions from coal, natural gas results in either similar or greater global warming-relevant-emissions than coal, particularly on the 20-year time scale (Howarth et al., 2011, 2012a, 2012b; Howarth and Ingraffea, 2011; Wigley, 2011; Myhrvold and Caldeira, 2012). The most efficient use of natural gas is for electricity, since the efficiency of electricity generation with natural gas is greater than with coal. Yet even with optimistic assumptions, Myhrvold and Caldeira (2012) demonstrated that the rapid conversion of coal to natural gas electricity plants would “do little to diminish the climate impacts” of fossil fuels over the first half of the 21st Century. Recent estimates of methane radiative forcing (Shindell et al., 2009) and leakage (Howarth et al., 2012b; Pétron et al., 2012) suggest a higher greenhouse-gas footprint of the natural gas systems than that estimated by Myhrvold and Caldeira (2012). Moreover, conventional natural gas resources are becoming increasingly depleted and replaced by unconventional gas such as from shale formations, which have larger methane emissions and therefore a larger greenhouse gas footprint than do conventional sources (Howarth et al., 2011, 2012b; Hughes, 2011).

Currently, most natural gas in the U.S. and NYS is not used to generate electricity but rather for domestic and commercial heating and for industrial process energy. For these uses, natural gas offers no efficiency advantage over oil or coal, and has a larger greenhouse gas footprint than these other fossil fuels, particularly over the next several decades, even while neglecting the climate impact of sulfur dioxide emissions (Howarth et al., 2011, 2012a, 2012b). The reason is that natural gas systems emit far more methane per unit energy produced than do other fossil fuels (Howarth et al., 2011), and methane has a global warming potential that is 72–105 times greater than carbon dioxide over an integrated 20-year period after emission and 25–33 times greater over a century period (Intergovernmental Panel on Climate Change (IPCC), 2007; Shindell et al., 2009). As discussed below, the 20-year time frame is critical.

When used as a transportation fuel, the methane plus carbon dioxide footprint of natural gas is greater than for oil, since the efficiency of natural gas is less than that of oil as a transportation fuel (Alvarez et al., 2012). When methane emissions due to venting of fuel tanks and losses during refueling are accounted for, the warming potential of natural gas over oil rises further.

When sulfur dioxide emissions from coal are considered, the greater air-pollution health effects of coal become apparent, but so do the lower global warming impacts of coal versus natural gas, indicating that both fuels are problematic. Coal combustion emits significant sulfur dioxide and nitrogen oxides, most of which convert to sulfate and nitrate aerosol particles, respectively. Natural gas also emits nitrogen oxides, but not much sulfur dioxide. Sulfate and nitrate aerosol particles cause direct air pollution health damage, but they are ‘‘cooling particles’’ with respect to climate because they reflect sunlight and increase cloud reflectivity. Thus, although the increase in sulfate aerosol from coal increases coal’s air-pollution mortality relative to natural gas, it also decreases coal’s warming relative to natural gas because sulfate offsets a significant portion of coal’s CO_2-based global warming over a 100-year time frame (Streets et al., 2001; Carmichael et al., 2002). Coal also emits ‘‘warming particles’’ called soot, but pulverized coal in the U.S. results in little soot. Using conservative assumptions about sulfate cooling, Wigley (2011) found that electricity production from natural gas causes more warming than coal over 50–150 years when coal sulfur dioxide is accounted for. The low estimate of 50 years was derived from an unrealistic assumption of zero leaked methane emissions.

Thus, natural gas is not a near-term ‘‘low’’ greenhouse-gas alternative, in absolute terms or relative to coal. Moreover, it does not provide a unique or special path to renewable energy, and as a result, it is not bridge fuel and is not a useful component of a sustainable energy plan.

Rather than use natural gas in the short term, we propose to move to a WWS-power system immediately, on a worldwide scale, because the Arctic sea ice may disappear in 20–30 years unless global warming is abated (e.g., Pappas, 2012). Reducing sea ice uncovers the low-albedo Arctic Ocean surface, accelerating global warming in a positive feedback. Above a certain temperature, a tipping point is expected to occur, accelerating the loss to complete elimination (Winton, 2006). Once the ice is gone, regenerating it may be difficult because the Arctic Ocean will reach a new stable equilibrium (Winton, 2006).

The only potential method of saving the Arctic sea ice is to eliminate emissions of short-lived global warming agents, including methane (from natural gas leakage and anaerobic respiration) and particulate black carbon (from natural gas flaring and diesel, jet fuel, kerosene burning, and biofuel burning). The 21-country Climate and Clean Air Coalition to Reduce Short-Lived Climate Pollutants recognized the importance of reducing methane and black carbon emissions for this purpose (UNEP (United Nations Environmental Program), 2012). Black carbon controls for this reason have also been recognized by the European Parliament (Resolution B7–0474/2011, September 14, 2011). Jacobson (2010a) and Shindell et al. (2012) quantified the potential benefit of reducing black carbon and methane, respectively, on Arctic ice.

Instead of reducing these problems, natural gas mining, flaring, transport, and production increase methane and black carbon, posing a danger to the Arctic sea ice on the time scale of 10–30 years. Methane emissions from the natural-gas system and nitrogen-oxide emissions from natural-gas combustion also contribute to the global buildup of tropospheric ozone resulting in additional respiratory illness and mortality.

2.2. Why not liquid biofuels?

This study also excludes the future use of liquid biofuels for transportation and heating. In addition to their creating more air pollution than gasoline for transportation, their tank-to-wheel efficiency of combustion is 1/4th to 1/5th the plug-to-wheel efficiency of electricity for transportation. This tends to make the energy cost-per-distance much higher for biofuel vehicles than electric vehicles. In addition, the land required to power a fleet of flex-fuel vehicles on corn or cellulosic ethanol is about 30 times the spacing area and a million times the footprint area on the ground required for wind turbines to power an equivalent fleet of electric vehicles (Jacobson, 2009).

Liquid biofuels are partially renewable with respect to carbon since they remove carbon dioxide from the air during photosynthetic growth. However, liquid biofuels require energy to grow and, in some cases (e.g., corn for ethanol) fertilize crops, irrigate crops (although not in NYS), distill the fuel (in the case of ethanol), transport crops to energy production plants, and transport the liquid fuel to its end use locations. For transportation, the resulting environmental costs of liquid biofuels are high, particularly for air and water quality (Delucchi, 2010), and greenhouse gas emissions are at best only slightly less than from using fossil fuels, and may be far worse when indirect land-use changes due to using land for fuel instead of food are fully considered (Searchinger et al., 2008). Moreover, carbon emissions from an advanced biofuel, cellulosic ethanol for flex-fuel vehicles, are about 125 times those from wind energy powering electric vehicles without considering indirect land use changes (Jacobson, 2009) and higher if indirect land use changes are accounted for (Searchinger et al., 2008). For these reasons alone, reviews by international agencies have recommended against the use of liquid biofuels for transportation (Bringezu et al., 2009; Howarth and Bringezu, 2009).

Ethanol combustion, regardless of the source, increases average air pollution mortality relative to gasoline due to the aldehyde and unburned ethanol emissions from ethanol fuel combustion (Jacobson, 2009; Anderson, 2009), and the effect increases at low temperature (Ginnebaugh et al., 2010, 2012). Ethanol and biodiesel fuel also increase air pollution from their upstream production more than do gasoline or diesel fuel, respectively (Delucchi, 2006). By contrast, electric and hydrogen fuel cell vehicles eliminate nearly all such pollution (Jacobson et al., 2005).

Much less analysis of the impacts of liquid biofuels for heating has been done than for transportation, but the fundamental issues remain the same. Namely, liquid biofuels for heating produce air pollution because they are combusted; require energy to grow, produce, and transport thus result in more emissions, and require much more land than solar power for the same energy output.

2.3. *Temporary role of solid biofuels*

The NYS plan allows for the temporary heating use of certain solid biofuels, such as wood pellets, energy crops grown on unused farmland, and agricultural waste and of biogas extracted from landfills and derived from anaerobic digestion of organic wastes. The use of such solid biofuels and biogas will be phased out by 2030–2050.

Solid biofuels combusted for cogeneration of electric power and heat are more efficient than liquid biofuels for transportation and are widely used in this way across northern Europe (Campbell et al., 2009; Howarth and Bringezu, 2009; Bringezu et al., 2009). Much of NYS is rural, with large expanses of old abandoned agricultural land, much of it now second-growth forest. Such land can produce large quantities of biomass. For example, the 8-county (Broome, Chemung, Chenango, Delaware, Schulyer, Steuben, Tioga, and Tompkins) Southern Tier economic development region of NYS is estimated to be able to produce 1.9 million dry tons annually of biomass for energy, with half of this coming from wood-chip harvest and the rest from dedicated energy crops such as switchgrass or willow (Woodbury et al., 2010). This is equivalent to 3 tons per year for every resident of this area, more than enough to alone supply all domestic heating needs.

Using biomass for heat allows farmers and forest owners to produce an energy crop on land that would not otherwise be used and to make use of low-value wood, increasing economic productivity and producing agricultural and forestry jobs. However, solid biomass should be used carefully so as not to over-harvest forestlands or use high-quality agricultural land. The scale of use is important as well, as moving and processing solid biomass takes substantial energy and carbon; the biomass should be used near the point of harvest to reduce this energy cost and the resulting environmental pollution. Using landfill biogas allows methane that would otherwise escape to the air to be used for energy. Similarly, converting organic waste to biogas allows the use of material for energy that would be processed biologically and released to the air in any case.

For two reasons, the use of solid biofuels and biogas in our plan is only temporary. First, biomass or biogas for energy requires much more land than solar power producing the same electricity and heat. For example, the growth of switchgrass for electric power requires about 115 times more land area than the use of solar PV to provide the same electric power based on biomass data from Kansas Energy Report (2011). If biomass combustion is used for both electricity and heat, switchgrass still requires 70 times more land area than does solar PV. Thus, one acre of land growing switchgrass for electricity produces 1/70th to 1/115th the usable energy of the same land with PV on it. Since electricity can run (a) air-source heat pumps very efficiently, (b) electric-resistance backup heating to produce heat, and (c) electrolyzers to produce hydrogen that can be used safely for home and building heat (KeelyNet, 2009), the use of solar PV for electricity and electricity-derived heat is more efficient than is the use of biomass for the same purpose in terms of land use and reducing air pollution,

Second, the use of solid biofuels or biogas for electricity and heat is still a combustion process, resulting in similar air pollution health and mortality impacts as fossil fuel combustion. Because solid biofuels for energy would be grown and processed in NYS, NYS "upstream" air pollution emissions from such processing will likely increase compared with current fossil fuel upstream emissions, most of which occur out of state (Woodbury et al., 2010). Because feedstock will be transported primarily by truck, road congestion, erosion, and pollution emissions will also likely increase (Woodbury et al., 2010). For these reasons, solid biofuels and biogas are to be phased out during 2030–2050 in the NYS plan.

3. Change in NYS power demand upon conversion to WWS

Table 1 summarizes the changes in global, U.S., and NYS end-use power demand between 2010 and 2030 upon a conversion to a 100% WWS infrastructure (zero fossil fuels, biofuels, and nuclear

Table 1
Contemporary (2010) and projected (2030) end-use power demand (TW) for all purposes by sector, for the world, U.S., and NYS if conventional fossil-fuel and wood use continue as projected and if all conventional fuels are replaced with WWS technologies.
Source: Jacobson and Delucchi (2011) for the world and U.S., NYS values are calculated with the same methodology but using EIA (Energy Information Administration, U.S.), 2012a end-use demand data. The U.S. and NYS populations in 2010 were 307,910,000 and 19,378,000, respectively. Those in 2030 are estimated to be 358,410,000 (USCB (United States Census Bureau), 2011) and 19,795,000 (Cornell Program on Applied Demographics, 2011), respectively, giving the U.S. and NYS population growths as 16.4% and 2.15%, respectively.

Energy sector	Conventional fossil fuels and wood 2010			Conventional fossil fuels and wood 2030			Replacing fossil fuels and wood with WWS 2030		
	World	U.S.	NYS	World	U.S.	NYS	World	U.S.	NYS
Residential	1.77	0.38	0.026	2.26	0.43	0.025	1.83	0.35	0.020
Commercial	0.94	0.28	0.023	1.32	0.38	0.025	1.22	0.35	0.022
Industrial	6.40	0.86	0.009	8.80	0.92	0.009	7.05	0.74	0.007
Transportation	3.36	0.97	0.036	4.53	1.10	0.037	1.37	0.33	0.011
Total	12.47	2.50	0.094	16.92	2.83	0.096	11.47	1.78	0.060
Percent change							(−32%)	(−37%)	(−37%)

energy). The table was derived on a spreadsheet from annually-averaged end-use power demand data as in Jacobson and Delucchi (2011). All end uses that feasibly can be electrified will use WWS power directly, and remaining end uses (some heating, high-temperature industrial processes, and some transportation) will use WWS power indirectly in the form of electrolytic hydrogen (hydrogen produced by splitting water with WWS power). As such, electricity requirements will increase, but the use of oil and gas for transportation and heating/cooling will decrease to zero. The increase in electricity use will be much smaller than the decrease in energy embodied in gas, liquid, and solid fuels because of the high efficiency of electricity for heating and electric motors.

The power required in 2010 to satisfy all end use power demand worldwide for all purposes was about 12.5 trillion watts (terawatts, TW). (End-use power excludes losses incurred during production and transmission of the power.) About 35% of primary energy worldwide in 2010 was from oil, 27% was from coal, 23% was from natural gas, 6% was from nuclear power, and the rest was from biofuel, sunlight, wind, and geothermal power. Delivered electricity was about 2.2 TW of all-purpose end-use power.

If the world follows the current trajectory of fossil-fuel growth, all-purpose end-use power demand will increase to ~17 TW by 2030, U.S. demand will increase to ~3 TW, and NYS power demand will increase to ~96 GW (Table 1). Conventional power demand in NYS will increase much less in 2030 than in the U.S. as a whole because the NYS population is expected to grow by only 2.15% between 2010 and 2030, whereas the U.S. population is expected to grow by 16.4% (Table 1, footnote).

Table 1 indicates that a conversion to WWS will reduce world, U.S., and NYS end-use power demand and power required to meet that demand by ~32%, ~37%, and ~37%, respectively. The reductions in NYS by sector are 21.0% in the residential, 12.3% in the commercial, 20.0% in the industrial, and 69.5% in the transportation sectors. Only 5–10 percentage points of each reduction are due to modest energy-conservation measures. Some of the remainder is due to the fact that conversion to WWS reduces the need for upstream coal, oil, and gas mining and processing of fuels, such as petroleum or uranium refining. The remaining reason is that the use of electricity for heating and electric motors is more efficient than is fuel combustion for the same applications (Jacobson and Delucchi, 2011). Also, the use of WWS electricity to produce hydrogen for fuel cell vehicles, while less efficient than the use of WWS electricity to run BEVs, is more efficient and cleaner than is combusting liquid fossil fuels for vehicles (Jacobson et al., 2005). Combusting electrolytic hydrogen is slightly less efficient but cleaner than is combusting fossil fuels for direct heating, and this is accounted for in the table.

4. Numbers of electric power Generators needed

How many WWS power plants or devices are needed to power NYS for all purposes assuming end use power requirements in Table 1 and accounting for electrical transmission and distribution losses?

Table 2 provides one of several possible future scenarios for 2030. In this scenario, onshore wind comprises 10% of New York's

Table 2
Number of WWS power plants or devices needed to provide New York's total annually-averaged end-use power demand for all purposes in 2030 (0.061 TW from Table 1) assuming the given fractionation of demand among plants or devices and accounting for transmission, distribution, and array losses. Also shown are the footprint and spacing areas required to power NYS as a percentage of New York's land area, 122,300 km^2.

Energy technology	Rated power of one plant or device (MW)	Percent of 2030 power demand met by plant/device	Number of plants or devices needed for NYS	Nameplate capacity of all devices (MW)	Footprint area (percent of NYS land area)	Spacing area (percent of NYS land area)
Onshore wind	5	10	4020	20,100	0.000041	1.46
Offshore wind	5	40	12,700	63,550	0.00013	4.62
Wave device	0.75	0.5	1910	1435	0.00082	0.039
Geothermal plant	100	5	36	3600	0.010	0
Hydroelectric plant	1300	5.5	6.6[a]	8520	3.50[a]	0
Tidal turbine	1	1	2600	2600	0.00061	0.0095
Res. roof PV system	0.005	6	4.97 million[b]	24,900	0.15[c]	0
Com/gov roof PV system	0.10	12	0.497 million	49,700	0.30[c]	0
Solar PV plant	50	10	828[b]	41,400	0.25	0[c]
CSP plant	100	10	387	38,700	0.60	0[c]
Total		100		254,000	4.82	6.13
Total new land required					0.96[d]	1.46[e]

Rated powers assume existing technologies. Percent power of each device assumes wind and solar are the only two resources that can power NYS independently (Section 5) and should be in approximate balance to enable load matching (Section 6) but that wind is less expensive (Section 7) so will dominate more. The number of devices is calculated by multiplying the NYS end use power demand in 2030 from Table 1 by the fraction of power from the source and dividing by the annual power output from each device, which equals the rated power multiplied by the annual capacity factor of the device. The capacity factor is determined for each device as in the Supplementary Information spreadsheet of Jacobson (2009), except that onshore wind turbines are assumed here to be located in mean annual wind speeds at hub height of 7.75 m/s and offshore turbines, 8.5 m/s (Dvorak et al., 2012a). From that study, 9200 km^2 of NYS land area has mean wind speeds >7.75 m/s at 90 m, and the average wind speed in those areas is 8.09 m/s. From the present table, only 1786 km^2 of onshore wind is needed. Land and spacing areas are similarly calculated as in the Supplementary Information of Jacobson (2009).

[a] NYS already produces about 89% of the hydroelectric power needed for the plan (Section 5). See Jacobson (2009) for a discussion of apportioning the hydroelectric footprint area by use of the reservoir.

[b] The solar PV panels used for this calculation were Sun Power E20 panels. The average capacity factor for solar assumed was 18%.

[c] For central solar PV and CSP plants, nominal "spacing" between panels is included in the plant footprint area.

[d] The total footprint area requiring new land is equal to the footprint area for onshore wind and geothermal, plus 2.75% of the footprint area for hydroelectric, plus the footprint area for solar PV and CSP plants. Offshore wind, wave and tidal are in water, and so do not require new land. The footprint area for rooftop solar PV does not entail new land because the rooftops already exist and are not used for other purposes (that might be displaced by rooftop PV). Only 2.75% of the hydropower requires new land because 89% of hydroelectric capacity is already in place and, of the remaining 11%, three-quarters will come from existing reservoirs or run-of-the-river.

[e] Only onshore wind entails new land for spacing area. The other energy sources are either in water or on rooftops, or do not use additional land for spacing. The spacing area for onshore wind can be used for multiple purposes, such as open space, agriculture, grazing, etc.

supply; offshore wind, 40%; residential solar rooftop PV, 6%; commercial/government solar rooftop PV, 12%; PV power plants, 10%; CSP plants, 10%; hydroelectric power, 5.5% (of which 89% is already in place), geothermal power, 5%; tidal power, 1%; and wave power, 0.5%.

Rooftop PV in this scenario is divided into residential (5-kW systems on average) and commercial/government (100-kW systems on average). Rooftop PV can be placed on existing rooftops or on elevated canopies above parking lots and structures without taking up additional undeveloped land. PV power plants are sized, on average, relatively small (50 MW) to allow them to be placed optimally in available locations.

Wind (50%) and solar (38%) are the largest generators of electric power under this plan because they are the only resources sufficiently available to power NYS on their own, and both are needed in combination to ensure the reliability of the grid. Wind is currently less expensive than solar, particularly at latitudes as high as in NYS, so wind is proposed to play a slightly larger role.

Since most wind and all wave and tidal power will be offshore under the plan, most transmission will be under water and out of sight. Transmission for new onshore wind, solar power plants, and geothermal power plants will be along existing pathways but with enhanced lines to the greatest extent possible, minimizing zoning issues. Four methods of increasing transmission capacity without requiring additional rights of way or increasing the footprint of transmission lines include the use of dynamic line rating equipment; high-temperature, low-sag conductors; voltage up-rating; and flexible AC transmission systems (e.g., Holman, 2011). To the extent existing pathways need to be expanded or new transmission pathways are required, they will be applied for using regulatory guidelines already in place.

Footprint is the physical space on the ground needed for each energy device, whereas spacing is the space between some devices, such as wind, tidal, and wave power. Spacing area can be used for open space, agriculture, grazing, etc. Table 2 provides footprint and spacing areas required for each energy technology. The table indicates that the total new land footprint required for this plan is about 0.96% of New York's land area, mostly for solar PV and CSP power plants (as mentioned, rooftop solar does not take up new land). Some additional footprint is proposed for hydroelectric as well, but that portion may not be needed if run-of-the-river hydro, imported hydro, or hydro from existing reservoirs that do not currently produce electric power is used. Additional space is also needed between onshore wind turbines. This space can be used for multiple purposes and can be reduced if more offshore wind resources are used than proposed here. The total additional land footprint needed (0.96% of the state) is minimal compared with the footprint of agriculture in the state (23.8%) and the footprint of house lots, ponds, roads, and wasteland used for agriculture (1.9%) (USDA (United States Department of Agriculture), 2011). Fig. 1 shows the relative footprint and spacing areas required in NYS.

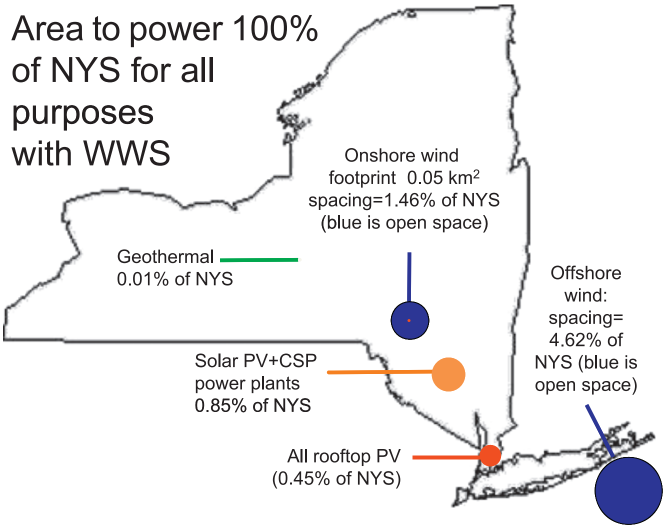

Fig. 1. Spacing and footprint areas required to implement the plan proposed here for NYS, as derived in Table 2. Actual locations would differ. The dots are only representative areas. For wind, the small red dot in the middle is footprint on the ground and the blue is spacing. For the others, the footprint and spacing are similar to each other. In the case of rooftop PV, the dot represents the rooftop area to be used. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

The number of devices takes into account the availability of clean resources as well as of land and ocean areas. NYS has more wind, solar, geothermal, and hydroelectric resources than is needed to supply the state's energy for all purposes in 2030. These resources are discussed next.

5. WWS resources available

This section discusses raw WWS resources available in NYS. Fig. 2 shows NYS's onshore and offshore annual wind resources from Dvorak et al. (2012a) in terms of a wind turbine's capacity factor, which is the annual average power produced divided by the rated power of a turbine. If only half the high-wind-speed land (capacity factor $>30\%$) in NYS were used for wind development, 327 TWh of wind energy would be harnessed, enough to provide more than 60% of NYS's 2030 WWS end-use power demand for all purposes. However, this plan proposes that only 10% of NYS's 2030 power demand come from onshore wind.

Dvorak et al. (2012a) mapped the East Coast offshore wind resources and Dvorak et al. (2012b) proposed locations for an efficiently interconnected set of offshore East Coast wind farms, one of which would be off of Long Island's coast. Offshore resources significantly exceed those onshore. The U.S. has not yet built an offshore wind farm, and some have expressed a concern over their potential environmental impacts. However, a study of over a decade of experience of offshore wind in Denmark by the International Advisory Panel of Experts on Marine Ecology found little damage to wildlife (Dong Energy, Vattenfall Danish Energy Authority, and Danish Forest and Nature Agency, 2006).

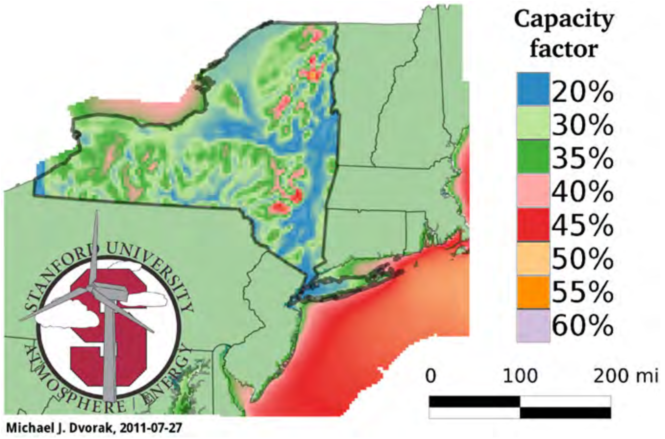

Fig. 2. Capacity factors at 90-m hub height in NYS and offshore in Lake Ontario, Lake Erie, and the Eastern seaboard, as calculated with a 3-D computer model evaluated against data assuming 5-MW RE-Power wind turbines with rotor diameter $D=126$ m from simulations run in Dvorak et al. (2012a, 2012b). Capacity factors of 30% or higher are the most cost-effective for wind energy development.

Despite NYS's high latitude, solar resources in the state are significant. NREL (National Renewable Energy Laboratory) (2008) estimates NYS's solar resources as 4–4.5 kWh/m^2/day. Based on these numbers, only 0.85% of additional land (beyond existing rooftops) is needed to provide 38% of the state's energy for all purposes in 2030 in the forms of CSP plants, PV power plants, and rooftop PV. This assumes that 18% of the state's new energy comes from rooftop PV on existing urban structures (Table 2).

Geothermal resources in NYS (NREL (National Renewable Energy Laboratory), 2009) are also abundant. Geothermal energy production requires little land area (Table 2) and is proposed to provide only 5% of NYS's total energy in 2030.

NYS has a hydroelectric potential of 38.6 kW/km^2 (5 GW, or 43.8 TWh/yr) of delivered power (DOE (Department of Energy), 2004). It can currently produce about 60% of this. For example, in 2009, hydroelectric supplied about 26.1 TWh/yr (3 GW delivered power), or 21% of NYS's electric power consumption of 131 TWh/yr. Under the plan, hydro will produce about 3.3 GW, or 5.5% of the total delivered power for all purposes in NYS in 2030. Hydro currently produces 89% of this amount. Sufficient in-state and, if necessary, imported hydroelectric power is available to provide the difference. Most additional in-state hydro may be obtainable from existing dams that do not have turbines associated with them.

Tidal (or ocean current) and wave power are proposed to comprise a combined 1.5% of NYS's overall power in 2030 (Table 2). Tidal and wave resources off the East Coast are both modest. However, tidal power has already been used to generate electricity in the East River through the Verdant Power Roosevelt Island Tidal Energy Project.

6. Matching electric power supply with demand

An important concern to address in a clean-energy economy is whether electric power demand can be met with WWS supply on a minutely, daily, and seasonal basis. Previous work has described multiple methods to match renewable energy supply with demand and to smooth out the variability of WWS resources (Delucchi and Jacobson, 2011). Such methods include (A) combining geographically-dispersed WWS resources as a bundled set of resources rather than separate resources and using hydroelectric or stored concentrated solar power to balance the remaining load; (B) using demand-response management to shift times of demand to better match the availability of WWS power; (C) over-sizing WWS peak generation capacity to minimize the times when available WWS power is less than demand and provide power to produce heat for air and water and hydrogen for transportation and heating when WWS power exceeds demand; (D) integrating weather forecasts into system operation; (E) storing energy in batteries or other storage media at the site of generation or use; and (F) storing energy in electric-vehicle batteries for later extraction (vehicle-to-grid). Here, we discuss updated information on only a couple of these methods since Delucchi and Jacobson (2011) discuss the other methods.

Several studies have examined whether up to 100% penetrations of WWS resources could be used reliably to match power demand (e.g., Jacobson and Delucchi, 2009; Mason et al., 2010; Hart and Jacobson, 2011, 2012; Connolly et al., 2011; Elliston et al., 2012; NREL (National Renewable Energy Laboratory), 2012; Rasmussen et al., 2012; Budischak et al., 2013). Using hourly load and resource data and accounting for the intermittency of wind and solar, both Hart and Jacobson (2011) and Budischak et al. (2013) found that up to $>99.8\%$ of delivered electricity could be produced carbon-free with WWS resources over multiple years. The former study obtained this conclusion for the California grid over 2 years; the latter, over the PJM Interconnection in the eastern U.S., adjacent to NYS, over 4 years. Both studies accounted for the variability in the weather, including extreme events. Although WWS resources differ in NYS compared with these other regions, the differences are not expected to change the conclusion that a WWS power system in NYS can be reliable. NYS has WWS resources not so different from those in PJM (more offshore wind and hydroelectric than PJM but less solar).

Eliminating remaining carbon emission is challenging but can be accomplished in several ways. These include using demand response and demand management, which will be facilitated by the growth of electric vehicles; oversizing the power grid and using the excess power generated to produce district heat through heat pumps and thermal stores and hydrogen for other sectors of the energy economy (e.g. heat for buildings, high-temperature processes, and fuel-cell vehicles); using concentrated solar power storage to provide solar power at night; and storing excess energy at the site of generation with pumped hydroelectric power, compressed air (e.g., in underground caverns or turbine nacelles), flywheels, battery storage packs, or batteries in electric vehicles (Kempton and Tomic, 2005).

Oversizing the peak capacity of wind and solar installations to exceed peak inflexible power demand can reduce the time that available WWS power supply is below demand, thereby reducing the need for other measures to meet demand. The additional energy available when WWS generation exceeds demand can be used to produce hydrogen (a storage fuel) by electrolysis for heating processes and transportation and to provide district heating. Hydrogen must be produced in any case as part of the WWS solution. Oversizing and using excess energy for hydrogen and district heating would also eliminate the current practice of shutting down (curtailing) wind and solar resources when they produce more energy than the grid can accommodate. Denmark currently uses excess wind energy for district heating using heat pumps and thermal stores (e.g., Elsman, 2009).

7. Costs

An important criterion in the evaluation of WWS systems is to ensure that the full costs per unit energy delivered, including capital, land, operating, maintenance, storage, and transmission costs, are comparable with or better than costs of conventional fuels.

Table 3 presents estimates of 2005–2012 and 2020–2030 costs of electric power generation for WWS technologies, assuming standard (but not extra-long-distance) transmission and excluding distribution. The table also shows the average U.S. delivered electricity cost for conventional fuels (mostly fossil) under the same assumptions. For fossil-fuel generation, the externality cost, which includes the hidden costs of air pollution morbidity and mortality and global warming damage (e.g., coastline loss, agricultural and fish losses, human heat stress mortality, increases in severe weather and air pollution), is also shown. Table 4 breaks down the externality costs.

Table 3 indicates that the 2005–2012 costs of onshore wind, hydroelectric, and geothermal plants are the same or less than those of typical new conventional technologies (such as new coal-fired or natural gas power plants) when externality costs of the conventional technologies are ignored. Solar costs are higher. When externality costs are included, WWS technologies cost less than conventional technologies.

The costs of onshore wind, geothermal, and hydroelectric power are expected to remain low (4–8.8 cents/kWh) in 2020–2030. Costs of other WWS technologies are expected to decline to 5–11 cents/kWh (Table 3). These estimates include the costs of local AC transmission. However, many wind and solar farms may be sufficiently far from population centers to require long-distance transmission.

For long-distance transmission, high-voltage direct-current (HVDC) lines are common because they result in lower transmission

Table 3
Approximate fully annualized generation and short-distance transmission costs for WWS power (2007 U.S. cents/kWh-delivered), including externality costs. Also shown are generation costs and externality costs (from Table 4) of new conventional fuels. Actual costs in NYS will depend on how the overall system design is optimized as well as how energy technology costs change over time.

Energy technology	2005–2012[*]	2020–2030[*]
Wind onshore	4[a]–10.5[b]	≤ 4[a]
Wind offshore	11.3[c]–16.5[b]	7[b]–10.9[c]
Wave	> 11.0[a]	4–11[a]
Geothermal	9.9–15.2[b]	5.5–8.8[g]
Hydroelectric	4.0–6.0[d]	4[a]
CSP	14.1–22.6[b]	7–8[a]
Solar PV (utility)	11.1–15.9[b]	5.5[g]
Solar PV (commercial rooftop)	14.9–20.4[b]	7.1–7.4[h]
Solar PV (residential rooftop)	16.5–22.7[e]	7.9–8.2[h]
Tidal	> 11.0[a]	5–7[a]
New conventional (plus externalities)[f]	**9.6–9.8 (+5.3)=14.9–15.1**	**12.1–15.0 (+5.7)=17.8–20.7**

[*] $0.01/kWh for transmission was added to all technologies as in Delucchi and Jacobson (2011) except for distributed generation projects (i.e. commercial and residential solar PV).

[a] Delucchi and Jacobson (2011).

[b] Lazard (2012).

[c] Levitt et al. (2011).

[d] REN21 (Renewable Energy Policy Network for the 21st Century) (2010).

[e] SEIA (Solar Energy Industries Association) (2012). Residential LCOE: Calculated by multiplying the Lazard (2012) Commercial LCOE by the ratio of the Residential PV $/Watt to the Commercial PV $/Watt=$0.149 ($5.73/$5.16)–$0.204($5.73/$5.16).

[f] The current levelized cost of conventional fuels in NYS is calculated by multiplying the electric power generation by conventional source in NYS (EIA (Energy Information Administration, U.S.), 2012b) by the levelized cost of energy for each source (Lazard, 2012 for low estimate; EIA (Energy Information Administration, U.S. (2012c) for high estimate) and dividing by the total generation. The future estimate assumes a 26.5% increase in electricity costs by 2020 (the mean increase in electricity prices in NYS from 2003 to 2011, EIA (Energy Information Administration, U.S.), 2012d), and twice this mean increase by 2030. Externality costs are from Table 4.

[g] Google (2011), 2020 projection.

[h] The ratio of present-day utility PV to present-day commercial and residential PV multiplied by the projected LCOE of utility PV.

Table 4
Mean (and range) of environmental externality costs of electricity generation from coal and natural gas (Business as Usual—BAU) and renewables in the U.S. in 2007 (U.S. cents/kWh). Water pollution costs from natural gas mining and current energy generation are not included. Climate costs are based on a 100-year time frame. For a 20-year time frame, the NG climate costs are about 1.6 times those of coal for the given shale:conventional gas mixes.
Source: Delucchi and Jacobson (2011) but modified for mean shale and conventional natural gas carbon equivalent emissions from Howarth et al. (2011) assuming a current shale:conventional NG mix today of 30:70 and 50:50 in 2030 and a coal/NG mix of 73%/27% in 2005 and 60%/40% in 2030. The costs do not include costs to worker health and the environment due to the extraction of fossil fuels from the ground. (These estimates apply to the U. S. Section 8 estimates external costs specifically for NYS.)

	2005			2030		
	Air pollution	Climate	Total	Air pollution	Climate	Total
Coal	3.2	3.0	6.2 (1.2–22)	1.7	4.8	6.5 (3.3–18)
Natural gas (NG)	0.16	2.7	2.9 (0.5–8.6) [a]	0.13	4.5	4.6 (0.9–8.9) [a]
Coal/NG mix	2.4	2.9	5.3 (1.0–18)	1.1	4.6	5.7 (2.7–15)
Wind, water, and solar	<0.01	<0.01	<0.02	<0.01	<0.01	<0.02

[a] McCubbin and Sovacool (2013) estimate slightly higher air pollution-plus-climate-change costs for natural-gas fired power plants in California: 1.4–9.5 cents/kWh for 1987–2006, and 1.8–11.8 cents/kWh projected for 2012–2031 (2010 dollars).

losses per unit distance than alternating-current (AC) lines. The cost of extra-long-distance HVDC transmission on land (1200–2000 km) ranges from 0.3 to 3 U.S. cents/kWh, with a median estimate of ~1 U.S. cent/kWh (Delucchi and Jacobson, 2011). A system with up to 25% undersea transmission would increase the additional long-distance transmission cost by less than 20%. Transmission costs can be reduced by considering that decreasing transmission capacity by 20% reduces aggregate power among interconnected wind farms by only 1.6% (Archer and Jacobson, 2007). The main barrier to long distance transmission is not cost, but local opposition to the siting of lines and decisions about who will pay the costs. These issues must be addressed during the planning process.

In sum, even with extra-long-distance HVDC transmission, the total social costs of all WWS resources in 2020–2030, including solar PV, are expected to be less than the 17.8–20.7 cents/kWh average direct plus externality cost of conventional electricity.

WWS will provide a stable, renewable source of electric power not subject to the same fuel supply limitations as fossil fuels and nuclear power. Due to the eventual depletion of coal, oil, natural gas, and uranium resources, their prices should ultimately rise although technology improvements may delay this rise. Table 5 projects fuel costs from 2009 to 2030 of selected conventional fossil fuels used for transportation, heating, and electricity production in NYS. The table indicates a 19–37% anticipated increase in the cost of natural gas and a 109% increase in the cost of gasoline during this period. A benefit of WWS is that it hedges NYS against volatility and rises in long-term fossil fuel prices by providing energy price stability due to zero cost of WWS fuel.

Table 5
Projected unit costs of selected conventional fossil fuels over the period 2009–2030 in NYS.
Source: NYSEPB (New York State Energy Planning Board) (2009), Energy Price and Demand Long-Term Forecast (2009–2028). Annual growth rate factors provided in reference document have been extrapolated for the period 2029–2030.

Fuel type	Projected changes in fuel cost, 2009–2030 (2009 dollars/ MMBTU)		Percent change (%)
	2009	2030	
Gasoline—all grades	\$19.30	\$40.39	109
Natural gas—electric	\$6.30	\$10.14	27
Natural gas—residential	\$13.58	\$16.19	19
Natural gas—commercial	\$10.27	\$13.06	27
Natural gas—industrial	\$8.73	\$11.98	37

8. Air pollution and global warming cost Reductions in NYS due to WWS

Conversion to a WWS energy infrastructure will reduce air pollution mortality and morbidity, health costs associated with mortality and morbidity, and global warming costs in NYS. These impacts are quantified here.

Air pollution mortality in New York is estimated in two ways, a top-down approach and a bottom-up approach. The top-down approach is described first. The premature mortality rate in the U.S. due to cardiovascular disease, respiratory disease, and complications from asthma due to air pollution has been calculated conservatively to be at least 50,000–100,000 per year by several sources. From Braga et al. (2000), the U.S. air pollution mortality rate was estimated at about 3% of all deaths. The all-cause death rate in the U.S. is about 804 deaths per 100,000 population and the U.S. population in 2011 was 308.7 million. This suggests an air pollution mortality rate in the U.S. of ~75,000 per year. Similarly, from Jacobson (2010b), the U.S. death rate due to ozone and particulate matter was calculated with a three-dimensional air pollution-weather model to be 50,000–100,000 per year. These results are consistent with those of McCubbin and Delucchi (1999), who estimated 80,000–137,000 due to all anthropogenic air pollution in the U. S. in 1990, when air pollution levels were higher than today.

The population of NYS in 2011 was 19.5 million, or 6.3% of the U.S. population. A simple scaling of population to the U.S. premature mortality rate from Jacobson (2010b) yields at least 3000–6000 annual premature deaths in NYS. Since a large segment of New York's population lives in cities, this estimate is likely conservative since the intake fraction of air pollution is much greater in cities than in rural areas.

Mortalities from airborne inhalation of particulate matter ($PM_{2.5}$) and ozone (O_3) are next calculated with a bottom-up approach. This involves combining measured countywide or regional concentrations of each pollutant with a relative risk as a function of concentration and U.S. Census Bureau population by county or region. From these three pieces of information, low, medium, and high mortality estimates of $PM_{2.5}$ and O_3 are calculated with a health-effects equation (Jacobson, 2010b).

Tables 6 and 7 show the resulting low, medium, and high 2006 premature mortalities estimates in NYS due to $PM_{2.5}$ and ozone respectively. The medium values for the state as a whole were about 3300 $PM_{2.5}$ mortalities/yr, with a range of 800–6500/yr and ~710 O_3 mortalities/yr, with a range of 360–1100/yr. Thus, overall, the bottom-up approach gave ~4000 (1200–7600) premature mortalities per year for $PM_{2.5}$ plus O_3. The top-down estimate falls within this range.

Table 6
NYS annually-averaged 2006 $PM_{2.5}$ concentrations and resulting estimated annual premature mortalities. Appendix Table A1 contains details and data by county.

New York State	2006 $PM_{2.5}$ ($\mu g/m^3$)	Population (thousands)	Total 2006 Mortalities from $PM_{2.5}$		
			Low estimate	Medium estimate	High estimate
Total	**9.3**	**19,380**	**820**	**3260**	**6480**

Concentration data were from NYSDH (New York State Department of Health) (2011). The methodology is described in the text.

Table 7
Average Annual 2009–2011 premature mortalities due to ground-level ozone by New York region.

	Annual premature mortalities due to ground-level ozone		
	Low estimate	Medium estimate	High estimate
Region 1	55.1	110	164
Region 2	103	205	306
Region 3	37.7	75.1	112
Region 4	10.7	21.4	32.0
Region 5	26.5	52.8	78.9
Region 6	8.4	16.8	25.1
Region 7	18.9	37.7	56.4
Region 8	15.8	31.5	46.8
Region 9	80.8	164	244
Total	**356**	**713**	**1070**

Hourly ozone data at individual monitoring stations were obtained for January 2009–October 2011 from NYDEC (New York State Department of Environmental Conservation (2011). The 1-h maximum ozone for each day was determined from all hourly values during the day. Monitoring stations were then grouped by regions defined by the NYS Department of Environmental Conservation. Region 1=Western New York, Great Lakes Plain; Region 2=Catskill Mountains and West Hudson River Valley; Region 3=Southern Tier; Region 4=New York City and Long Island; Region 5=East Hudson and Mohawk River Valleys; Region 6=Tug Hill Plateau; Region 7=Adirondack Mountains. Mortalities were calculated each day for each region based on ozone relative risks and a health-risk equation, as in Jacobson (2010b). The low-threshold for ozone premature mortality referenced in this study was 35 ppbv.

USEPA (United States Environmental Protection Agency) (2006) and Levy et al. (2010) provided a central estimate to the value of a statistical life at \$7.7 million in 2007 dollars (based on 2000 GDP). The value of life is determined by economists based on what people are willing to pay to avoid health risks as determined by how much employers pay their workers to take additional risks (Roman et al., 2012). With this value of life, 4000 (1200–7600) premature mortalities (both adult and infant) due to air pollution cost NYS roughly \$31 (\$9–\$59) billion/yr.

Additional costs due to air pollution result from increased illness (morbidity from chronic bronchitis, heart disease, and asthma), hospitalizations, emergency-room visits, lost school days, lost work days, visibility degradation, agricultural and forest damage, materials damage, and ecological damage. USEPA (United States Environmental Protection Agency), 2011 estimates that these non-mortality-related costs comprise an additional ~7% of the mortality-related costs. These are broken down into morbidity (3.8%), recreational plus residential visibility loss (2.8%), agricultural plus forest productivity loss (0.45%), and materials plus ecological loss (residual) costs. These estimates are conservative, as other studies in the economics literature indicate considerably higher non-mortality costs. McCubbin and Delucchi's (1999) detailed, comprehensive analysis of air-pollution damages at every air quality monitor in the U.S found that the morbidity cost of air pollution (mainly chronic illness from exposure to particulate matter) is 25–30% of the mortality costs. Delucchi and McCubbin (2011) summarize studies that indicate that the cost of visibility and agriculture damages from motor-vehicle air pollution in the U.S. is at least 15% of the cost of health damages (including morbidity damages) from motor-vehicle air pollution. Thus, the total cost of air pollution, including morbidity and non-health damages, is at the very least ~$8.2 million/death, and probably over $10 million/death.

Given this information, the total social cost due to air pollution mortality, morbidity, lost productivity, and visibility degradation in NYS is conservatively estimated to be $33 (10–76 [using $10 million/death for the upper end]) billion per year. Reducing these costs represents a savings equivalent to ~3% of NYS's gross 2010 domestic product of $1.1 trillion.

One set of cost estimates for global warming (in 2006 U.S. dollars) to the U.S. alone is $271 billion/yr by 2025, $506 billion/yr by 2050, $961 billion/yr by 2075, and $1.9 trillion/yr by 2100 (Ackerman et al, 2008). That analysis accounted for severe-storm and hurricane damage, real estate loss, energy-sector costs, and water costs. The largest of these costs was water costs. It did not account for increases in mortality and illness due to increased heat stress, influenza, malaria, and air pollution or increases in forest-fire incidence; thus, it may be conservative.

Averaged between 2004 and 2009, NYS contributed to 3.39% of U.S. and 0.636% of world fossil-fuel CO_2 emissions (EIA (Energy Information Administration, U.S.), 2011). Since the global warming cost to the U.S. is caused by emissions from all states and countries worldwide, it is necessary to multiply the cost of global warming to the U.S. by NYS's fraction of global CO_2 emissions to give the cost of global warming to the U.S. due to NYS's greenhouse gas emissions. The result is $1.7 billion/yr by 2025, $3.2 billion/yr by 2050; $6.1 billion/yr by 2075; and $12 billion/yr by 2100. NYS's emissions are also increasing the health and climate costs to other countries of the world.

In sum, the current fossil-fuel energy infrastructure in NYS causes ~4000 (1200–7600) annual premature mortalities, which together with other air-pollution damages cost the state ~$33 billion/yr (~3% of its annual GDP). Fossil fuels emitted in the state will also result in ~$1.7 billion/yr in global warming costs to the U.S. alone by 2025. Converting to WWS in the state will eliminate these externalities and their costs.

Since every 1 MW of installed WWS capacity costs ~$2.1 million averaged over all generation technologies needed, the $33 billion annual air-pollution cost is equivalent to ~16 GW of installed WWS power every year. Since the state needs ~271 GW of installed WWS power to deliver the 60 GW needed (Table 1) to power the state for all purposes in 2030, the payback time to convert the state as a whole to WWS, is ~16 years from the mean air-pollution-cost savings alone. The payback time accounting for air-pollution plus global-warming-cost savings is ~15 years; that accounting for air-pollution plus warming-cost benefits plus electricity sales at no profit is 10 years; that accounting for these plus 7% profit is ~9.8 years.

9. Jobs and earnings due to new electric power plants and devices

This section discusses job creation and earnings resulting from implementing the WWS electric power infrastructure described in Table 2. The analysis is limited to the electric power generation sector to provide an example. Additional jobs are expected in the electricity transmission industry, electric vehicle and hydrogen fuel cell vehicle industries, in the heating and cooling industries, and with respect to energy use for high-temperature industrial processes, but estimates for these sectors are not provided here due to the large undertaking such a calculation requires.

9.1. Onshore and offshore wind

The job creation and revenue stream resulting from generating half of NYS's all-purpose power in 2030 from onshore plus offshore wind (Table 2) were estimated with the Jobs and Economic Development Impact (JEDI) wind model (DOE (Department of Energy), 2012).

Scenarios were run assuming the development by 2025 of 200 onshore wind farms containing 4020 5-MW turbines with a total nameplate capacity of 20,100 MW and 400 offshore wind farms containing 12,700 turbines with a total nameplate capacity of 63,550 MW.

The development of the onshore wind farms is calculated to create ~61,300 full-time jobs and >$4 billion in earnings in the form of wages, services, and supply-chain impacts during the construction period. It is also estimated to create ~2260 annual full-time jobs and >$162 million in annual earnings in the form of wages, local revenue, and local supply-chain impacts post-construction.

The development of the offshore wind farms is estimated to create 320,000 full-time jobs and >$21.4 billion in earnings during construction and 7140 annual full-time jobs and >$514 million in annual earnings post-construction. (Section 9.5 discusses the extent to which WWS jobs merely displace jobs in the current energy sector.)

9.2. Concentrated solar power plants, solar PV power plants, and rooftop solar PV

The job creation and revenue stream resulting from generating 38% of NYS's all-purpose energy in 2030 with concentrated solar power (CSP, 10%) and solar PV plants and residential rooftop devices (PV, 28%), were estimated with the JEDI Concentrated Solar Power Trough and PV models (DOE (Department of Energy), 2012).

Scenarios were run assuming the development by 2025 of 38,700 MW in nameplate capacity of CSP projects, 41,400 MW of solar PV plant projects, and 75,000 MW of residential, commercial, and government rooftop PV projects.

The CSP projects are estimated to create ~401,000 full-time jobs and >$41 billion in earnings during construction and ~15,700 full-time jobs and >$2 billion in annual earnings post-construction.

Solar PV plants are estimated to create ~1,160,000 full-time jobs (>$83 billion in earnings) during construction and ~5690 full-time jobs (>$390 million in annual earnings) post-construction.

Rooftop PV systems are estimated to create ~2,420,000 full-time jobs (~$159 billion in earnings) during construction and ~9620 full-time jobs (>$676 million in annual earnings) post-construction.

9.3. Hydroelectric, tidal, and wave

In line with the guidelines of PlaNYC, nearly 7% of NYS's total energy in 2030 will be generated from hydroelectric, tidal, and wave power (Table 2). At most, about 944 MW of additional installed hydroelectric will be needed for the present plan, since 89% of hydroelectric is in place (Table 2). This translates into 2360 additional post-construction full time jobs assuming 2–3 full time jobs are created per MW of hydropower generated in 2025 (Navigant Consulting, 2009). Temporary construction and other supply chain jobs are not included in this projection. Temporary construction jobs for hydroelectric are estimated as 6.5 full-time equivalent (FTE) jobs/MW. FTEs are jobs during the life of the construction phase (Navigant Consulting, 2009). This gives 6200 construction jobs for hydroelectric. With the approximate ratio of

$70,000 per job (based on the ratios determined here for wind and solar), the earnings during construction of hydroelectric plants are estimated as ~$430 million during construction and $165 million/yr after construction.

For wave power (1430 MW needed) and tidal power (2600 MW needed) the same number of construction and permanent jobs per installed MW as offshore wind power are assumed, giving 7200 construction jobs and 161 annual permanent jobs for wave power and 13,100 construction jobs and 292 annual permanent jobs for tidal power. Earnings during the construction period of wave farms are estimated as ~$504 million, and those during operation, ~$11 million/yr. Earnings during construction of tidal farms are estimated as ~$920 million, and those during operation, ~$20.5 million/yr.

9.4. *Geothermal*

The construction of 5635 MW of geothermal capacity in the western United States has been estimated previously to create 90,160 construction and manufacturing jobs plus 23,949 full time jobs after construction (Western Governor's Association, 2010). Assuming the same relationship holds for NYS in 2025, the 3600 MW of geothermal energy (5% of total) needed for NYS will amount to the creation of ~57,600 construction and manufacturing jobs and ~15,300 post-construction jobs. With the approximate ratio of $70,000 per job, the earnings during construction of geothermal plants will be ~$4 billion during the construction period and $1 billion/yr thereafter.

9.5. *Summary of jobs and earnings*

Summing the job production from each sector above gives ~4.5 million jobs created during construction and ~58,000 permanent annual jobs thereafter for the energy facilities alone developed as part of this plan. Total earnings during the construction period for these facilities (in the form of wages, local revenue, and local supply-chain impacts) are estimated as ~$314 billion and permanent annual earnings during operation of the facilities, ~$5.1 billion/yr

Additional jobs and earnings are associated with the enhancement of the transmission system and with the conversion to electric and hydrogen fuel cell vehicles, electricity-based appliances for home heating and cooling, and electricity and hydrogen use for some heating and high-temperature industrial processes.

The number of permanent jobs created by the electric power sector alone is expected to exceed significantly the number of lost jobs in current fossil-fuel industries. The reason is that nearly all energy for NYS with the proposed plan will be produced within the state, whereas currently, most oil, natural gas, and coal used in the state is mined out of the state or country, so jobs in those industries are not in NYS. In fact, the total number of mining jobs (for all natural resources combined) in NYS in 2011 was approximately 5700 (NYSDL (New York State Department of Labor), 2011). The total number of workers in the NYS utility industry in 2011 was about 37,100 (NYSDL (New York State Department of Labor), 2011). Even if the current electric utility industry plus mining jobs were lost due to a conversion with the present plan, they would be more than made up by with the 58,000 permanent jobs resulting from the present plan. The present plan would also result in the replacement of gas stations with electric charging and hydrogen fueling stations, likely exchanging the jobs between the industries. Similarly, the plan will require the growth of some appliance industries at the expense of others, resulting in job exchange between industries.

The increase in the number of jobs due to WWS versus the current fossil fuel infrastructure is supported independently by Pollin et al. (2009), who determined from economic modeling that, for each million dollars spent on energy production in the United States, oil and gas create 3.7 direct and indirect jobs, whereas wind and solar create 9.5 and 9.8 jobs, respectively. The difference in relative numbers of jobs created in NYS is likely to be larger than this due to the fact that many oil and gas workers and suppliers come from out of state. Since WWS resources are generated in state, their capture will provide more jobs to NYS residents. In addition, even though some of the jobs in NYS might come at the expense of jobs in other states, Pollin et al. (2009) indicate that for the U.S. as a whole, the wind and solar power industry will employ many more people than will an energy-equivalent fossil-fuel industry.

In addition, the development of the large-scale energy infrastructure proposed here should motivate research and development of new technologies and methods of improving efficiency. Much of this research will come from higher education and research institutes in NYS, creating jobs in these sectors. Demands created by infrastructure development should similarly motivate inner-city job training programs in the energy-efficient building and renewable energy industries.

10. State and local tax revenue and other cost considerations

The implementation of this plan will likely affect NYS's tax revenue and may require tax policy changes to ensure that state revenue remains at the level needed. Some revenues will increase and others will decline.

The increase in the number of jobs due to the plan over the current energy infrastructure is expected to increase personal income tax receipts. In addition, as more of NYS's infrastructure is electrified under the plan, revenues from the Utility Tax, which currently accounts for slightly less than 1.5% of state tax revenue, will increase.

NYS may experience higher property tax revenues than under an alternative, natural gas, infrastructure. Property values may decrease with shale gas drilling due to the increases in noise, conflicts with neighbors, lawsuits with gas companies, health complaints, and increases in crime in previously sparsely populated rural areas. In addition, banks may be unwilling to issue residential-rate mortgages on residential properties in gas drilling areas since industrial activity and the storing of hazardous material on the property violate residential mortgage requirements. Similarly, some insurance companies may not issue policies on such properties. Property tax revenues are expected to increase with some WWS technologies, such as rooftop PV and solar thermal due to the higher home values that result from installation of these local energy technologies. A study of the effects of 24 existing wind farms within 10 miles of residential properties in 9 states found no effect on property values (Hoen et al., 2009). Thus, a conversion to WWS should result in higher property values and tax revenues than should a fossil fuel-based infrastructure.

Finally Delucchi and Murphy (2008) show that in 1991 and 2000, the effective U.S. federal corporate income tax rate (tax paid divided by taxable income) in the oil industry was half that of all other industries, resulting in a tax "subsidy" in the year 2000 of $9.4 billion. Replacing fossil fuels with WWS energy in NYS alone could result in higher corporate income-tax revenues to the nation and may set an example for other states.

Revenues directly associated with the sale of petroleum fuels, such as the Motor Fuel Tax and the Petroleum Business Tax, will diminish as the vehicle fleet is made more efficient and ultimately transitions away from petroleum altogether. These tax revenues currently account for less than 2.5% of state tax revenue; however, they are sources of funds for the Highway and Bridge Trust Fund, the Dedicated Mass Transportation Trust Fund, and the

Mass Transportation Operating Assistance Fund. Another potential loss in tax revenue will be from the ad valorem tax on shale gas development.

As diesel fuel is phased out, goods will increasingly be transported by means other than commercial freight, and revenue from the Highway Use Tax will diminish. This tax accounts for less than 0.2% of state tax revenue at present, but is also a large contributor to transportation infrastructure and operation funds (NYSA (New York State Assembly), 2011).

Other tax revenues associated with passenger vehicle use are not expected to decrease significantly. These include Motor Vehicle Fees, Taxi Surcharge fees, and Auto Rental Tax. These collectively account for approximately 2% of State tax revenue and contribute to the state's dedicated mass transportation and highway and bridge funds.

Some lost revenues can be regained by applying a mileage-based road use tax on noncommercial vehicles similar to the Highway Use Tax levied on commercial vehicles in NYS. This has been considered at the Federal level (NSFIFC (National Surface Transportation Infrastructure Financing Commission), 2009) and piloted in Oregon (ODT (Oregon Department of Transportation), 2007).

There are other cost considerations. For example, the conversion from fossil fuels to WWS will likely reduce environmental externality costs, thereby possibly preserving some jobs that would otherwise be lost under future fossil fuel development in NYS. Some industries that are vital to upstate NY economies and require clean water and air include agriculture, tourism, organic farming, wine making, hunting and fishing, and other outdoor recreation industries. WWS development is unlikely to adversely impact these industries, whereas future shale gas development may negatively impact these industries.

It is expected that costs to communities in NYS will increase with shale gas development, and these costs will likely be much lower or not exist with WWS development. Such costs include increased demand on police, fire departments, first responders, social services, and local hospitals. Damage to roads and resulting repair and maintenance costs have been substantial where shale gas development has taken place, especially in Texas and Arkansas. WWS development is unlikely to cause such extensive long-term damage to roads and infrastructure.

Thousands of miles of natural gas pipelines represent an opportunity cost to NYS, as future building and economic development will not be possible on or adjacent to the pipelines. The tradeoff for these pipelines with WWS is an increase in transmission lines. However, transmission lines, while resulting in some similar issues, do not carry the risk of gas leakage or explosive fires, such as the $5 billion fire that destroyed a residential neighborhood in San Bruno, California, on September 10, 2010.

Finally, extractive industries, including fossil fuels, are known for their boom and bust cycles. Renewable energy industries, and in particular WWS, are long-term sustainable industries, unlikely to be subject to boom and bust cycles.

11. Reducing energy use in Buildings, Neighborhoods, and commercial complexes

The proposed plan will continue existing efforts to improve energy efficiency in residential, commercial, institutional, and government buildings to reduce the demand for electric power in NYS. It will also encourage the conversion of buildings, neighborhoods, and commercial complexes to sustainable ones that use and store their energy more efficiently.

First, energy efficiency measures in buildings, appliances, and processes have the potential to reduce end-use power demand in the U.S. by up to 23% by 2020 (McKinsey and Company, 2009). Such a demand reduction exceeds the modest reduction of 5–10% proposed in Table 1 of the present study. The NYS demand reduction is conservative to ensure that it does not underestimate the number of energy devices and plants needed for NYS. If demand reduction is larger than 5–10%, then the NYS plan will be easier to implement. Efficiency measures include improving wall, floor, ceiling, and pipe insulation, sealing leaks in windows, doors, and fireplaces, converting to double-paned windows, using more passive solar heating, monitoring building energy use to determine wasteful processes, performing an energy audit to discover energy waste, converting to LED light bulbs, changing appliances to those using less electricity, and using hot water circulation pumps on a timer, among others.

Historically, efficiency programs targeting multifamily households have resulted in overall energy savings of approximately 20% (Falk and Robbins, 2010). For such households, the NYSERDA Home Performance with Energy Star program reportedly achieved annual savings of approximately 15% of average household electricity usage and over 50% of heating fuel savings for natural gas-heated homes (NYSERDA (New York State Energy Research and Development Authority), 2011).

Second, designing new buildings, neighborhoods and commercial complexes or retrofitting existing ones to use and store energy more efficiently has the potential to reduce significantly building energy required from the grid, transmission needs, and costs. Four methods of improving energy use and storage in buildings include: (1) extracting heat in the summer and cold in the winter from the air and solar devices and storing it in the ground for use in the opposite season, (2) recovering heat from air conditioning systems and using it to heat water or air in the same or other buildings, (3) extracting heat (or cold) from the ground, air, or water with heat pumps and using it immediately to heat (or cool) air or water, and (4) using solar energy to generate electricity through PV panels, to recover heat from water used to cool the panels, and to heat water directly for domestic use (e.g., Tolmie et al., 2012). The Drake Landing solar community is a prototype community designed primarily around the first method, that of seasonal energy storage (Drake Landing, 2012).

12. Timing of plan

This plan anticipates that the fraction of new electric power generators as WWS will increase starting today such that, by 2020, all new generators will be WWS generators. Existing conventional generators will be phased out gradually, but no later than 2050. Similarly, all new heating and cooling technologies will be WWS technologies by 2020 and existing technologies will be replaced over time, but by no later than 2050.

For transportation, the transition to BEVs and HFCVs has potential to occur rapidly due to the rapid turnover time of the vehicle fleet (~15 years) and the efficiency of BEVs and HFCVs over fossil-fuel combustion vehicles. However, the actual rate of transition will depend on policies put in place and the resulting vehicle and energy costs. BEVs and HFCVs exist today, but due to their efficiency over combustion, they are proposed to be the only new vehicles sold in NYS by 2020. Several electric vehicles are currently available (e.g., Tesla Model S, 499 km (310 mile) range; Tesla Roadster, 391 km (243 mile); Renault Fluence Z.E., 185 km (115 mile); Citroen C-Zero, 177 km (110 mile); Mitsubishi I MiEV, 177 km (110 mile); Tazzari Zero, 140 km (87 mile); Ford Focus, 129 km (80 mile); Nissan Leaf, 117 km (73 mile)). The growth of electric vehicles will be accompanied by an increase in electric charging stations in residences, commercial parking spaces, and service stations. Most charging will be done with 220 V chargers over several hours, but 440 V chargers are now available for faster charging. For example, the Tesla Model S includes 440 V, 160 A charging capability that will allow sufficient power for a 310 mile range in about 1 h.

13. Recommended first Steps

Below are recommended short-term policy steps to start the conversion to WWS in NYS.

13.1. Large energy projects: offshore/onshore wind; solar PV/CSP, geothermal, hydro

- Direct the New York State Energy Research and Development Authority (NYSERDA) to issue a new main tier solicitation to meet its existing renewable portfolio standard (RPS) commitments through 2015, selecting and contracting with sufficient wind and solar projects to do so.
- Extend the RPS in NYS. The 30% RPS currently sunsets in 2015. Propose to ramp up the RPS each year to get to 50% by 2025 (2% per year).
- Set a goal of at least 5000 MW offshore wind by 2020. Direct the New York Power Authority (NYPA) and the Long Island Power Authority (LIPA) to issue requests for proposals (RFPs) for new power generation from offshore wind as part of their generation and procurement budgets.
- Set up a Green Bank, which is a vehicle for public–private financing in conjunction with long-term contracts for large wind and solar development projects in NYS. An example Green Bank exists in Connecticut. The Green Bank would include a statewide version of the Department of Energy Loan Guarantee Program that focuses specifically on WWS energy generation projects. Such a program will reinvigorate private lending activity.
- Lock in upstate coal-fired power plants to retire under enforceable commitments. At the same time, streamline the permit approval process for WWS power generators and the associated high-capacity transmission lines and eliminate bureaucratic hurdles involved in the application process. Promote expanding transmission of power between upstate and downstate and between onshore and offshore, in particular.
- Work with regions and localities, and the federal government (in the case of offshore wind) to reduce the costs and uncertainty of projects by expediting their physical build-out by managing zoning and permitting issues or pre-approving sites.
- Encourage regulators to require utilities to obtain permission for a certain capacity of electric power to be installed before auctioning off projects to lowest-bidding developers. Currently, a pre-approved Power Purchase Agreement between a utility and particular project developer is required before permission from the regulators can be obtained. This change will ensure end-users obtain electricity at the lowest price.

13.2. Small energy projects: residential commercial, and government rooftop solar PV

- Extend the New York Sun (NY Sun) program to a multi-year program to finance rooftop and on-site solar projects in the state.
- Implement virtual net metering (VNM) for small-scale energy systems. The following recommendations will render utility-scale wind and solar power net metering conducive to corporate clients, and pave the way for a more widespread subscription to off-site generating project for the public at large.

(1) Remove the necessity for subscribers to have proprietorship in the energy-generating site.
(2) Expand or eliminate the capacity limit of renewable power under remote net-metering for each utility.
(3) Remove the barrier to inter-load zone transmission of net-metered renewable power.
(4) Expand Public Service Law 66.j to reduce red tape and enable off-site virtual net-metering from upstate to downstate, and from the outer boroughs to Manhattan.

- Streamline the small-scale solar and wind installation permitting process. Currently, each municipality has its own permitting process and fee structure. Creating common codes, fee structures, and filing procedures across a state would reduce a barrier to the greater implementation of small-scale solar and wind.
- Develop community renewable energy facilities, whereby a community buys power from a centralized generation facility. The facility feeds power into the grid, and the utility credits the kilowatt-hours to the accounts of individuals, businesses, and any other electricity customer that sign up. The facility may be located anywhere in the utility's service territory, since all that is required is a bill crediting arrangement by the utility. This brings many advantages: economies of scale of the facility, siting in an ideal location, and broader inclusiveness. Many electricity users cannot install a renewable energy system, because they are renters or because their property is not suitable for a system. Community renewable energy is inclusive because it enables anyone, whether living in rural New York or an apartment building in Manhattan, to buy the power without having to host the system. New York already has a community renewable energy program, but it is restrictive. A simple legislative fix would enable this approach to be used widely.
- Encourage clean-energy backup emergency power systems rather than diesel/gasoline generators. For example, work with industry to implement home energy storage (through battery systems) accompanying rooftop solar to mitigate problems associated with grid power losses.
- Implement feed-in tariffs (FITs) for small-scale energy systems. FITs are financial incentives to promote investment in renewable power generation infrastructure, typically by providing payments to owners of small-scale solar PV systems to cover the difference between renewable energy generation cost (including grid connection costs) and wholesale electricity prices.

13.3. Energy efficiency in buildings and the grid

- The current target for energy efficiency is 15% less energy use below forecasted levels by 2015. Expand the target significantly beyond 2015 and increase investment fivefold from both public and private sources. This requires the New York State Public Service Commission (NYSPSC) to increase NYSERDA and utility requirements and budgets for efficiency.
- Promote, through municipal financing, incentives, and rebates, energy efficiency measures in buildings, appliances, and processes. Efficiency measures include improving wall, floor, ceiling, and pipe insulation, sealing leaks in windows, doors, and fireplaces, converting to double-paned windows, using more passive solar heating, monitoring building energy use to

determine wasteful processes, performing an energy audit to discover energy waste, converting to LED light bulbs, changing appliances to those using less electricity, and using hot water circulation pumps on a timer, among others.

- Encourage conversion from natural gas water and air heaters to heat pumps (air and ground-source) and rooftop solar thermal hot water pre-heaters. Incentivize the use of efficient lighting in buildings and on city streets.
- Encourage utilities to use demand-response grid management to reduce the need for short-term energy backup on the grid. This is a method of giving financial incentives to electricity users to shift times of certain electricity uses to times when more energy is available.
- Institute, through Empire State Development Corporation, a revolving loan fund to pay for feasibility analyses for commercial Energy Services Agreements. The revenues from these retrofits are amortized as a majority percentage of the Energy-Cost Savings realized as direct result of these retrofits. ROI's can be realized in 5–10 years with 10–20 year Energy Services Contracts. Allocating some of these revenues back to the fund will render it sustainable.
- Extract heat in the summer and cold in the winter from the air and solar devices and store it in the ground for use in the opposite season. The Drake Landing solar community is a prototype community designed primarily around seasonal energy storage (Drake Landing, 2012).
- Recover heat from air conditioning systems and use it to heat water or air in the same or other buildings at the same time.
- Extract heat (or cold) from the ground, air, or water with heat pumps and use it immediately to heat (or cool) air or water.
- Recover heat from water used to cool solar PV panels to heat water directly for domestic use.

13.4. Vehicle electrification

- Coordinate items below so that vehicle programs and public charging stations are developed in sync. Create a governor-appointed EV Advisory Council, as has been done in states such as Illinois and Connecticut, to recommend strategies for EV infrastructure and policies. Council members should include representatives from state agencies, environmental groups, utilities, auto companies, and EV charging infrastructure companies.
- Leverage and augment the technical and financial assistance of the U. S. Department of Energy's "Clean Cities Program" activities, focusing on the deployment of EVs.
- Adopt legislation mandating the transition to plug-in electric vehicles for short- and medium distance government transportation and encouraging the transition for commercial and personal vehicles through purchase incentives and rebates.
- Encourage fleets of electric and/or hydrogen fuel cell/electric hybrid buses starting with a few and gradually growing the fleets. Electric or hydrogen fuel cell ferries, riverboats, and other local shipping should be encouraged as well.
- Encourage and ease the permitting process for the installation of electric charging stations in public parking lots, hotels, suburban metro stations, on streets, and in residential and commercial garages.
- Ensure that new charging infrastructure is vehicle-to-grid (V2G)-capable, and integrated into a statewide "smart grid" system.
- Set up time-of-use electricity rates to encourage charging at night.
- Provide electric vehicle drivers access to high-occupancy vehicle (HOV) lanes.
- Use excess wind and solar produced by WWS electric power generators to produce hydrogen (by electrolysis) for transportation and industry and to provide district heating (as done in Denmark) instead of curtailing the wind and solar.

13.5. Industrial processes

- Provide incentives for industry to convert to electricity and electrolytic hydrogen for high temperature and manufacturing processes where they are not currently used.
- Encourage industries to use WWS electric power generation for on-site electric power (private) generation.

14. Conclusions

This study examined the technical and economic feasibility of and proposed policies for converting New York State's energy infrastructure for all purposes into a clean and sustainable one powered by wind, water, and sunlight producing electricity and hydrogen. Such a conversion is estimated to improve the health and welfare of NYS residents, thereby lowering their medical, insurance, and related costs, and is expected to create jobs to manufacture, install, and manage the infrastructure.

The study found that complete conversion to WWS in NYS will reduce end-use power demand by ~37%, due mostly to the efficiency of electricity versus combustion, but also due partly to energy efficiency measures.

If complete conversion to WWS occurs, the 2030 NYS power demand for all purposes (not only electricity) could be met by 4020 onshore 5-MW wind turbines (providing 10% of NYS's energy for all purposes), 12,770 off-shore 5-MW wind turbines (40%), 387 100-MW concentrated solar plants (10%), 828 50-MW solar-PV power plants (10%), 5 million 5-kW residential rooftop PV systems (6%), 500,000 100-kW commercial/government rooftop systems (12%), 36 100-MW geothermal plants (5%), 1910 0.75-MW wave devices (0.5%), 2600 1-MW tidal turbines (1%), and 7 1300-MW hydroelectric power plants (5.5%), of which 89% are already in place. The onshore wind capacity installed under this plan (~20.1 GW) would be less than twice the 2012 installed capacity of Texas.

Several methods exist to match renewable energy supply with demand and to smooth out the variability of WWS resources. These include (A) combining geographically-dispersed WWS resources as a bundled set of resources rather than as separate resources and using hydroelectric power to fill in remaining gaps; (B) using demand-response grid management to shift times of demand to match better with the timing of WWS power supply; (C) over-sizing WWS peak generation capacity to minimize the times when available WWS power is less than demand and to provide power to produce heat for air and water and hydrogen for transportation and heating when WWS power exceeds demand; (D) integrating weather forecasts into system operation to reduce reserve requirements; (E) storing energy in thermal storage media, batteries or other storage media at the site of generation or use; and (F) storing energy in electric-vehicle batteries for later extraction (vehicle-to-grid).

The additional footprint on land for WWS devices is equivalent to about 0.96% of New York's land area, mostly for CSP and PV. An additional on-land spacing area of about 1.46% is required for on-shore wind, but this area can be used for multiple purposes, such as open space, agricultural land, or grazing land, for example.

The land footprint and spacing areas (open space between devices) in the proposed scenario can be reduced by shifting more land based WWS generators to the ocean, lakes, and rooftops.

2020–2030 electricity costs are estimated to be 4–8.8 cents/kWh for most WWS technologies and 5–11 cents/kWh for others (including local transmission and distribution), which compares with about 17.8–20.7 cents/kWh for fossil-fuel generators in 2030, of which 5.7 cents/kWh are externality costs. Long-distance transmission costs on land are estimated to be 1 (0.3–3) cent/kWh for 1200–2000 km high-voltage direct current transmission lines.

Although the cost of WWS electricity is expected to be lower than that of fossil fuels and all energy in a WWS world will be transformed to electricity, infrastructure conversion will result in other cost tradeoffs not quantified here. For example, conversion from combustion vehicles to electric and hydrogen fuel cell vehicles and from current combustion-based heating technologies to electricity based technologies may result in large initial cost increases to consumers, when relatively low levels of vehicles are being manufactured. However, as production of new vehicles increases and technology matures, manufacturing costs will decline, and this, combined with the lower energy and operating costs of electric vehicles, may result eventually in electric vehicles having a total lifetime cost comparable with that of conventional gasoline vehicles (Delucchi and Lipman, 2010),

The plan is estimated to create ∼4.5 million jobs during construction and ∼58,000 permanent annual jobs thereafter for the proposed energy facilities alone. Total earnings during the construction period for these facilities (in the form of wages, local revenue, and local supply-chain impacts) will be ∼$314 billion and permanent annual earnings during operation of the facilities will be ∼$5.1 billion/yr

The implementation of this plan will likely increase personal income, property, and utility tax revenues in NYS relative to the current infrastructure. At the same time, it will reduce fuel-tax revenues. These can be made up from either the utility taxes or mileage-base road fees.

The plan effectively pays for the 100% WWS energy generation infrastructure to power NYS for all purposes over 15 years solely by the reduction in air-pollution costs to the state and global warming costs to the U.S. from state emissions. Annual electricity sales equal to the cost of the plant divided by its expected life (∼30 years) reduce the payback time to ∼10 years. The current fossil-fuel infrastructure does not provide the air-quality benefits to NYS, so it's payback time with annual electricity sales equal to the cost of the plant and fuel divided by the expected plant life is ∼30 years; assuming a 7% profit, it is ∼28 years.

This plan may serve as a template for plans in other states and countries. Results here suggest that the implementation of plans such as this in countries worldwide should reduce global warming, air, soil, and water pollution, and energy insecurity.

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Appendix A1

See Appendix Table A1.

Table A1
NYS annually-averaged 2006 $PM_{2.5}$ concentrations and resulting estimated annual premature mortalities by county.

County	2006 $PM_{2.5}$ ($\mu g/m^3$)	Population (thousands)	Total 2006 Mortalities from $PM_{2.5}$		
			Low estimate	Medium estimate	High estimate
Albany	9.4	304	8.4	33.4	66.5
Alleghany*	8.2	49	0.9	3.5	6.9
Bronx	13.9	1385	88.4	351	695
Broome**	10.3	201	7.0	27.8	55.4
Cattaraugus*	9.6	80	2.3	9.3	18.6
Cayuga*	8.3	80	1.5	5.9	11.8
Chautauqua	8.3	135	2.5	10.0	20.0
Chemung*	8.2	89	1.6	6.3	12.6
Chenango*	10.3	50	1.8	7.0	13.9
Clinton*	5.5	82	0.9	3.6	7.3
Columbia*	9.4	63	1.7	6.9	13.8
Cortland*	8.3	49	0.9	3.7	7.3
Delaware*	10.3	48	1.7	6.7	13.2
Dutchess**	10.7	297	11.3	45.1	89.7
Erie	10.9	919	36.4	145	289
Essex	5.5	39	0.4	1.7	3.5
Franklin*	6.0	52	0.6	2.5	4.9
Fulton*	11.5	56	2.5	9.8	19.6
Genesee*	10.3	60	2.1	8.3	16.5
Greene*	9.4	49	1.4	5.4	10.8
Hamilton*	6.0	5	0.1	0.2	0.5
Herkimer*	6.4	65	0.8	3.3	6.6
Jefferson*	6.4	116	1.5	6.0	12.0
Kings	12.8	2505	138	547	1090
Lewis*	6.4	27	0.4	1.4	2.8
Livingston*	8.9	65	1.5	6.0	12.0
Madison*	8.3	73	1.4	5.5	10.9
Monroe	9.5	744	21.1	84.1	168
Montgomery*	11.5	50	2.2	8.9	17.7
Nassau	10.8	1340	52.0	207	412
New York	14.4	1586	108	427	845
Niagara	10.4	216	7.7	30.7	61.2
Oneida**	10.5	235	8.5	34.1	67.8
Onondaga	8.3	467	8.7	34.7	69.1
Ontario*	8.9	108	2.5	9.9	19.8
Orange	9.7	373	11.2	44.5	88.7
Orleans*	10.0	43	1.4	5.5	10.9
Oswego*	8.3	122	2.3	9.1	18.1
Otsego*	10.5	62	2.3	9.0	18.0
Putnam*	10.4	100	3.5	14.0	27.9
Queens	11.6	2231	101	402	800
Rensselaer*	9.4	159	4.4	17.5	34.9
Richmond	12.2	469	23.5	93.5	186
Rockland*	10.4	312	11.0	43.7	87.1
St. Lawrence	6.4	112	1.4	5.8	11.5
Saratoga*	11.5	220	9.8	38.9	77.3
Schenectady**	11.5	155	6.9	27.4	54.5
Schoharie*	9.4	33	0.9	3.6	7.2
Schuyler*	8.2	18	0.3	1.3	2.6
Seneca*	8.2	35	0.6	2.5	5.0
Steuben**	8.2	99	1.8	7.0	14.0
Suffolk	10.4	1493	53.1	212	422
Sullivan*	9.7	78	2.3	9.3	18.4
Tioga*	10.3	51	1.8	7.1	14.1
Tompkins*	9.4	102	2.8	11.0	21.9
Ulster*	9.7	182	5.5	21.8	43.4
Warren*	5.5	66	0.7	2.9	5.8
Washington*	5.5	63	0.7	2.8	5.6
Wayne*	9.5	94	2.7	10.6	21.1
Westchester	11.0	949	38.4	153	304
Wyoming*	10.9	42	1.7	6.7	13.2
Yates*	8.7	25	0.5	2.2	4.3
Total	**9.3**	**19,380**	**820**	**3260**	**6480**

Concentration data were from NYSDH (New York State Department of Health) (2011). The methodology is described in the text.

* 2006 data for these counties were not available, so an average of data from adjacent or nearby counties was used.

** 2006 data for these counties were not available, so 2003 values were used.

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Quantifying sources of methane using light alkanes in the Los Angeles basin, California

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Abstract

Methane (CH_4), carbon dioxide (CO_2), carbon monoxide (CO), and C_2–C_5 alkanes were measured throughout the Los Angeles (L.A.) basin in May and June 2010. We use these data to show that the emission ratios of CH_4/CO and CH_4/CO_2 in the L.A. basin are larger than expected from population-apportioned bottom-up state inventories, consistent with previously published work. We use experimentally determined CH_4/CO and CH_4/CO_2 emission ratios in combination with annual State of California CO and CO_2 inventories to derive a yearly emission rate of CH_4 to the L.A. basin. We further use the airborne measurements to directly derive CH_4 emission rates from dairy operations in Chino, and from the two largest landfills in the L.A. basin, and show these sources are accurately represented in the California Air Resources Board greenhouse gas inventory for CH_4. We then use measurements of C_2–C_5 alkanes to quantify the relative contribution of other CH_4 sources in the L.A. basin, with results differing from those of previous studies. The atmospheric data are consistent with the majority of CH_4 emissions in the region coming from fugitive losses from natural gas in pipelines and urban distribution systems and/or geologic seeps, as well as landfills and dairies. The local oil and gas industry also provides a significant source of CH_4 in the area. The addition of CH_4 emissions from natural gas pipelines and urban distribution systems and/or geologic seeps and from the local oil and gas industry is sufficient to account for the differences between the top-down and bottom-up CH_4 inventories identified in previously published work.

1. Introduction

In California, methane (CH_4) emissions are regulated by Assembly Bill 32, enacted into law as the California Global Warming Solutions Act of 2006, requiring the state's greenhouse gas (GHG) emissions in the year 2020 not to exceed 1990 emission levels. To this end, the California Air Resources Board (CARB) was tasked with compiling and verifying an inventory of GHG emissions for the state. Two published works [*Wunch et al.*, 2009; *Hsu et al.*, 2010] have concluded that atmospheric emissions of CH_4 in the Los Angeles (L.A.) area were greater than expected from a per capita apportionment of the statewide 2006 CARB GHG inventory and from a bottom-up accounting of CH_4 sources, respectively.

Several recent works have estimated CH_4 emissions to the South Coast Air Basin (SoCAB; Fig. 1a), which are summarized in Table 1. *Wunch et al.* [2009] used a Fourier transform infrared spectrometer at the Jet Propulsion Laboratory (JPL) in Pasadena, California to measure vertically-integrated total column enhancement ratios of CH_4 relative to CO and to CO_2. The observed column enhancement ratios, multiplied by CARB inventory values of CO for 2008 and an average of 2006 CARB GHG inventory and 2005 Emission Database for Global Atmospheric Research (EDGAR) for CO_2, were used to derive a lower limit to CH_4 emissions of 400 ± 100 Gg CH_4/yr (based on CO) or 600 ± 100 Gg CH_4/yr (based on CO_2) for the SoCAB. One reason for the discrepancy in their top-down analysis was that their observed CO/CO_2 enhancement ratio of 11 ± 2 ppb CO/ppm CO_2 was greater than the 8.6 ppb CO/ppm CO_2 calculated from the inventories. *Wunch et al.* [2009] contrasted these top-down assessments to a bottom-up estimate of 260 Gg CH_4/yr using the statewide 2006 CARB GHG inventory apportioned by population after removal of agricultural and forestry emissions, and concluded that 140 – 340 Gg CH_4/yr were not accounted for in the CARB CH_4 inventory for the SoCAB.

Hsu et al. [2010] took a similar top-down approach and used observed atmospheric enhancement ratios of CH_4 to CO from *in situ* whole air samples taken at Mt. Wilson (34.22° N, 118.06° W, 1770 m above sea level), scaled by the projected CARB CO inventory for 2008, to derive CH_4 emissions of 200 ± 10 Gg CH_4/yr for just the Los Angeles (L.A.) County (Figure 1b) portion of the SoCAB (L.A. County $\cap$ SoCAB). They used methods prescribed by the Intergovernmental Panel for Climate Change (IPCC) to create the CARB GHG inventory and reached a bottom-up estimate of 140 Gg CH_4/yr, or 60 Gg less than their top-down calculation for the L.A. County portion of the SoCAB. *Hsu et al.* [2010] used higher spatial resolution emissions data from CARB to construct their bottom-up inventory, and therefore did not have to rely on population apportionment methods used by *Wunch et al.* [2009].

The difference between the top-down CH_4 emissions reported by *Wunch et al.* [2009] and by *Hsu et al.* [2010] (400 Gg and 200 Gg, respectively, both based on the CARB CO inventory) are in part due to the different geographic areas for which they calculate CH_4 emissions, and in part due to differences in observed CH_4/CO enhancements between these two studies: 0.66 ± 0.12 mol/mol for *Wunch et al.* [2009] [*Wennberg et al.*, 2012] and 0.52 ± 0.02 mol/mol for *Hsu et al*. [2010]. Both works suggested that fugitive losses of natural gas (NG) could be the source of the CH_4 missing from the bottom-up inventories.

More recently, *Townsend-Small et al.* [2012] analyzed stable CH_4 isotope ratios in atmospheric samples taken at Mt. Wilson and elsewhere in the western L.A. basin and showed they were consistent with isotope ratios in natural gas sources.

Wennberg et al. [2012] used the different atmospheric ethane/CH_4 enhancement ratios observed from research aircraft during the Arctic Research of the Composition of the Troposphere from Aircraft and Satellites (ARCTAS) field project in 2008 and the California Research at the Nexus of Air Quality and Climate Change (CalNex) field project [*Ryerson et al.*, in review] in 2010 to estimate an upper limit of 390 Gg CH_4/yr from natural gas leakage in the SoCAB. Further, their top-down analysis resulted in a calculated total emission of 440 Gg CH_4/yr in the SoCAB. *Wennberg et al.* [2012] also recalculated the data used by *Hsu et al.* [2010] to derive CH_4 emissions for the entire SoCAB, and calculated a SoCAB CH_4 emission from 2008 using data from ARCTAS. The results are summarized in Table 1.

Here we use ambient measurements in the SoCAB taken in May and June 2010 aboard the National Oceanic and Atmospheric Administration (NOAA) P-3 research aircraft during the CalNex field study to derive CH_4 emissions from the SoCAB using methods different from *Wennberg et al.* [2012]. We further examine CH_4 emissions from landfills and dairy farms in the SoCAB identified in the bottom-up CH_4 inventories reported by *Hsu et al.* [2010] and *Wennberg et al.* [2012]. We then expand on these previous studies by examining light alkane emissions from Los Angeles area data sets. In addition to CH_4 and ethane, we examine propane, *n*- and *i*-butane, and *n*- and *i*-pentane measurements to derive emissions of each of these light alkanes in the SoCAB, and use them in a system of linear equations to further quantify the source apportionment of CH_4 in the L.A. basin.

2. Measurements

We use trace gas measurements from a subset of platforms and sites from the CalNex field study. The NOAA P-3 research aircraft flew all or parts of 16 daytime flights in and around the L.A. basin. Two independent measurements of CH_4 and CO_2 were made aboard the aircraft by wavelength-scanned cavity ring-down spectroscopy (WS-CRDS; Picarro 1301-m) [*Peischl et al.*, 2012], and by quantum cascade laser direct absorption spectroscopy (QCLS) [*Kort et al.*, 2011]. Imprecision of the 1-Hz Picarro CH_4 measurement is ± 1.4 ppbv (all uncertainties herein are 1-σ) and inaccuracy is estimated at ± 1.2 ppbv. Imprecision of the 1-Hz QCLS CH_4 measurement is ±1 ppbv and inaccuracy is estimated at ± 15 ppbv. Imprecision of the 1-Hz Picarro CO_2 measurement is ± 0.14 ppmv and inaccuracy is estimated at ± 0.12 ppmv. Imprecision of the 1-Hz QCLS CO_2 measurement is ± 0.05 ppmv and inaccuracy is estimated at ± 0.10 ppmv. All CH_4 and CO_2 measurements are reported as dry air mole fractions. For this work, CH_4 and CO_2 data from the Picarro instrument are used, and QCLS CH_4 data from May 8 are used when the Picarro instrument was not operating. The 1-Hz CO data used in this analysis were measured by vacuum ultraviolet fluorescence spectroscopy [*Holloway et al.*, 2000]. Imprecision of the 1-Hz CO data is ± 1 ppbv; inaccuracy is estimated at ± 5%. C_2 to C_5 alkanes, and their structural isomers, were measured in whole air samples [*Colman et al.*, 2001], periodically filled during flight. Imprecision of these alkane measurements is ±5%; inaccuracies are estimated at ±10%. Wind measurements were derived from various sensors aboard the NOAA P-3; the uncertainty of the 1-Hz wind speed is estimated to be ± 1 m/s. Sensors aboard the NOAA P-3 also measured relative humidity, ambient temperature, and potential temperature with an estimated 1-Hz uncertainty of ± 0.5° C, ± 0.5° C, and ± 0.5 K, respectively.

At the CalNex Pasadena ground site, located on the California Institute of Technology (Caltech) campus, measurements of C_2–C_5 alkanes were made by a gas chromatograph-mass spectrometer (GC-MS) on 5 minute integrated samples taken every half hour [*Gilman et al.*, 2010]. Imprecision of these measurements are ±8% for ethane and ±6% for propane; inaccuracy is estimated at ± 15% for each. Data from the ground site were taken between 15 May and 15 June, 2010. CH_4 was not measured at the Pasadena ground site.

Additionally, whole-air flask samples were taken twice daily at the Mount Wilson Observatory (MWO) for most days during May and June 2010, and analyzed for a variety of trace gas species, including CH_4, CO_2, CO, and hydrocarbons [*Dlugokencky et al.*, 2011; *Conway, et al.*, 2011; *Novelli et al.*, 2010]. Imprecision of the CH_4 measurement is ± 1 ppb; imprecision of the CO_2 measurement is ± 0.1 ppm; imprecision of the CO measurement is ± 1 ppbv, and inaccuracy of CO measurement is estimated to be ± 5%.

We also analyze alkane data from whole air samples taken in the L.A. basin prior to 2010. Ethane and propane were measured in whole air samples taken on four flights in L.A. aboard an instrumented National Aeronautics and Space Administration (NASA) DC-8 research aircraft during ARCTAS in June 2008 [*Simpson et al.*, 2010]. Ethane and propane were also measured on one flight in L.A. aboard the NOAA P-3 during the Intercontinental Transport and Chemical Transformation (ITCT) study in May 2002 [*Schauffler et al.*, 1999].

3. Methods

To ensure sampling from the L.A. basin, we consider aircraft data collected between 33.6 and 34.3° N latitude and 118.5 and 116.8° W longitude (Figure 1d, dashed box) in the following analysis. Aircraft data were further limited to samples taken between 1000 and 1700 PST, between 200 and 800 m above ground, and below 1400 m above sea level, to ensure daytime sampling was within the well-mixed boundary layer, which averaged 1000 ± 300 m above ground level for the daytime L.A. flights [*Neuman et al.*, 2012]. Ground-based measurements at Pasadena were retained between 1000 and 1700 PST to ensure sampling of a well-mixed daytime boundary layer. For MWO measurements, afternoon samples, which typically occurred between 1400 and 1500 PST, were retained to capture upslope transportation from the L.A. basin [*Hsu et al.*, 2010]. Linear fits to the data presented below are orthogonal distance regressions [*Boggs et al.*, 1989] weighted by instrument imprecision [*Bevington*, 1969] (weighted ODR). The total uncertainty in the fitted slope is calculated by quadrature addition of the fit uncertainty and the measurement uncertainties.

For flux determinations, crosswind transects were flown downwind of known point sources. Enhancements of CH_4 above background levels were integrated along the flight track, and a flux was calculated using the following equation:

$$flux = v\ cos(\alpha) \int_{Z_0}^{Z_1} n(z)\,dz \int_{-y}^{y} X_m(y)\,dy \tag{1}$$

where v cos(α) is the component of the average wind velocity normal to the flight track, n is the number density of the atmosphere, z_0 is the ground level, z_1 is the estimated boundary layer height, and X_m is the measured mixing ratio enhancement above the local background along the flight track [*White et al.*, 1976; *Trainer et al.*, 1995; *Ryerson et al.*, 1998; *Nowak et al.*, 2012]. Boundary layer heights are estimated from vertical profiles of relative humidity, ambient temperature, and potential temperature made prior to and after the crosswind transects.

We assume the plume is vertically homogeneous within the mixed layer at the point of measurement and the wind velocity is constant between emission and measurement. We estimate the uncertainty in these assumptions, combined with the uncertainties of the wind speed, wind direction, temperature, and integrated atmospheric enhancements, to be ± 50% for the plumes studied here [*Nowak et al.*, 2012]. Weighted averages of the fluxes are calculated following *Taylor* [1997]. When calculating the CH_4 flux from dairies, CH_4 variability immediately upwind of the dairies is sufficiently large to complicate interpolation from the downwind local background. To account for this, we take the weighted ODR slope of CH_4/CO immediately upwind, multiply this ratio by the measured CO downwind of the dairies, and integrate the plume CH_4 enhancement calculated from CO ($CO \times [CH_4/CO]_{upwind}$), similar to the integrations performed by *Nowak et al.* [2012]. This assumes the dairies emit a negligible amount of CO.

As with previously published works [*Wunch et al.*, 2009; *Hsu et al.*, 2010; *Wennberg et al.*, 2012], we estimate total CH_4 emissions in the SoCAB by multiplying enhancement ratios of CH_4 to CO and CO_2 by inventory estimates of CO and CO_2 for that region:

$$E_{CH_4} = \left(\frac{CH_4}{X}\right)_{ODR\ slope} \times \left(\frac{MW_{CH_4}}{MW_X}\right) \times E_X \qquad (2)$$

where E_{CH4} is the emission of CH_4, X is either CO or CO_2, MW is the molecular weight, and E_X is the inventory emission value of either CO or CO_2. Although not necessarily emitted from the same sources, we assume emissions of CH_4, CO, and CO_2 are well-mixed by the time they are sampled from the NOAA P-3.

We use the following latest-available inventories for our analysis below: the 2010 CARB emissions inventory for CO projected from the base-year 2008 inventory (http://www.arb.ca.gov/app/emsinv/fcemssumcat2009.php), and the 2009 CARB GHG inventory (http://www.arb.ca.gov/cc/inventory/data/data.htm). Both inventories were accessed in November 2012.

CARB projects the total 2010 annually averaged CO emissions in the SoCAB at 979 Gg CO/yr (Table 2). We use the annually averaged CARB inventory that excludes biomass burning CO emissions because no known biomass burning events were observed in the L.A. basin during CalNex. This estimate is 4% less than the summertime CO inventory without biomass burning emissions, and approximately 6% less than the annually averaged CO inventory including biomass burning emissions used by *Wennberg et al.* [2012]. To estimate 2010 CH_4 emissions in the SoCAB using the 2009 CARB GHG inventory, we follow the method used by *Wunch et al.* [2009], and take the total statewide emission of 1525 Gg CH_4/yr, less agricultural and forestry CH_4 emissions of 898 Gg CH_4/yr, then apportion the remainder by population. In 2010, the SoCAB comprised 43% of California's population (http://www.arb.ca.gov/app/emsinv/trends/ems_trends.php). However, unlike *Wunch et al.* [2009], we include SoCAB dairy emissions of 31.6 Gg CH_4/yr, calculated in section 4.3 below. Therefore, we attribute a total of 301 Gg CH_4/yr to the SoCAB based on the 2009 CARB GHG inventory (Table 2).

According to CARB's mobile source emission inventory (EMFAC 2011) for the Los Angeles County portion of the SoCAB (http://www.arb.ca.gov/jpub/webapp//EMFAC2011WebApp/emsSelectionPage_1.jsp), mobile source CO_2 emissions remained essentially unchanged between 2009 and 2010 (39.94 versus 39.95 Tg CO_2/yr). Additionally, the statewide CARB GHG inventory for CO_2, with out-of-state electricity generation emissions removed, decreased by less than 2% between 2008 and 2009. Therefore, we assume errors due to sampling year are negligible in examining the CO_2 emission inventories in the SoCAB from 2009–2010. To estimate 2010 CO_2 emissions in the SoCAB using the 2009 CARB GHG inventory, we take the total statewide emission of 465.7 Tg CO_2/yr, subtract out-of-state electricity generation of 47.9 Tg CO_2/yr, then apportion the remainder by population. We therefore attribute 180 Tg CO_2/yr to the SoCAB using the 2009 CARB GHG inventory (Table 2). We do not compare to the Vulcan CO_2 inventory [*Gurney et al.*, 2009] because at present it is only available for the 2002 reporting year.

4. Results and Discussion

4.1. Total derived emission of CH_4 in L.A. and comparison to inventories

In this section, we use P-3 measurements of CH_4, CO, and CO_2 to calculate enhancement ratios representative of the integrated emissions from the L.A. basin. We then use tabulated CO and CO_2 emissions taken from the CARB inventories to derive total CH_4 emissions based on enhancement ratios observed in CalNex, and compare to earlier estimates of total CH_4 emissions in L.A.

Figure 1c shows known stationary sources of CH_4 in the L.A. area, which include landfills, dairies, wastewater treatment facilities, and oil fields, as well as the location of measurement sites used in this study. Dairy sources are sized by estimated CH_4 emissions from enteric fermentation, as explained in section 4.3. Landfills are sized by CH_4 emissions from the 2008 CARB GHG inventory (L. Hunsaker, personal communication, 2011).

Point sources are sized by 2009 CARB individual facility CH_4 emissions (https://ghgreport.arb.ca.gov/eats/carb/index.cfm), but do not stand out in the map due to their low CH_4 emissions relative to the landfills and dairies. Figure 1d shows the locations of daytime boundary-layer CH_4 data from the P-3, colored by observed mixing ratio, that were retained for the analysis as described previously. The largest concentrations of CH_4 were typically encountered along the mountains at the north edge of the L.A. basin, likely driven by transport of air within the basin, as typical daytime winds in the L.A. basin were from the west and southwest during May and June 2010 [*Washenfelder et al.*, 2011]. CalNex CH_4 data are plotted against observed CO in Figure 2a. Weighted ODR fits to these data resulted in derived enhancement ratios of 0.74 ± 0.04 and 0.68 ± 0.03 ppbv CH_4/ppbv CO from the NOAA P-3 and MWO, respectively. We note that the same CH_4/CO enhancement ratio of 0.74 ± 0.03 was reported by *Wennberg et al.* [2012] using the CalNex P-3 data with different selection criteria. We include box and whisker plots in Figure 2a to show that the weighted ODR fit to the data is insensitive to the relatively few data points of higher CH_4. The ratio calculated from the CARB inventory (Table 2) is 0.54 ppb CH_4/ppb CO, and is displayed for comparison.

CalNex CH_4 data are plotted against observed CO_2 in Figure 2b. The slope from a weighted ODR of P-3 data is 6.70 ± 0.01 ppb CH_4/ppm CO_2 and of MWO data is 6.60 ± 0.04 ppb CH_4/ppm CO_2. The ratio of the CARB inventories from Table 2 is 4.64 ppb CH_4/ppm CO_2, and is displayed for comparison. In this case, because CH_4 and CO_2 are measured with high precision and accuracy, the largest uncertainties in interpreting the slope as an emissions ratio are likely determined by the extent of mixing of emissions from different sources within the Los Angeles air shed. Similarly, Figure 2c shows a correlation plot of CO against CO_2.

The slope from a weighted ODR of P-3 data is 9.4 ± 0.5 ppb CO/ppm CO_2 and of MWO data is 10.4 ± 0.5 ppb CO/ppm CO_2. The ratio of the CARB inventories from Table 2 is 8.5 ppb CO/ppm CO_2, and is plotted for comparison. We estimate a ± 7.5% uncertainty in each of the CARB CO and CO_2 inventories, which is sufficient to explain the difference between the CO/CO_2 enhancement ratio measured from the NOAA P-3 and the ratio calculated from the CARB inventories. Quantitative agreement between emission ratios derived from P-3 and MWO data (Figures 2a–c) is likely due to the fact that the transport within the basin was driven by the land-sea breeze, meaning typical daytime winds in the Pasadena area near Mt. Wilson were from the southwest [*Washenfelder et al.*, 2011]. This transport, and the highest values of CH_4 and CO_2 in the P-3 data that are not seen at MWO (Figures 2a and b), also suggests that MWO preferentially samples the western part of the L.A. basin [*Hsu et al.*, 2009]. We therefore use enhancement ratios determined from the NOAA P-3 data to derive CH_4 emissions from the entire basin.

We note that the ratio of the latest CARB CO and CO_2 inventories (Table 2) are in better agreement with ambient enhancement ratios in the CalNex data than was the case for *Wunch et al.* [2009]. This is likely due to either improved CARB inventories, the present use of a basin-wide data set to determine basin-wide emission ratios, or both.

With the slopes and inventory values quantified, we next derive a CH_4 emission using equation (2). Using the CH_4/CO slope derived from the weighted ODR fit to the 2010 NOAA P-3 data and the projected 2010 CARB annually-averaged CO emission inventory in equation (2) yields an estimated SoCAB emission of 410 ± 40 Gg CH_4/yr. The stated uncertainty is the quadrature propagation of the measurement uncertainty, errors on the slope of the ODR fit to P-3 data, and an estimated uncertainty in the CARB CO inventory. We note our derived emission of 410 ± 40 Gg CH_4/yr is similar to that derived from the P-3 data by *Wennberg et al.* [2012], which was 440 ± 100 Gg CH_4/yr using different selection criteria. It is further consistent with the emission derived by *Wunch et al.* [2009] of 400 ± 100 Gg CH_4/yr, which assumed a CARB CO inventory uncertainty of 15%. We also determine CH_4 emissions using estimates of CO_2 emissions in the SoCAB. P-3 measurements of the CH_4/CO_2 enhancement ratio observed during CalNex and SoCAB CO_2 emissions inferred from the 2009 CARB GHG inventory result in a derived CH_4 emission rate of 440 ± 30 Gg CH_4/yr, with the stated uncertainties determined by quadrature propagation of the measurement uncertainty, errors on the slope of the ODR fit to P-3 data, and an estimated uncertainty in the CARB CO_2 inventory. This value, based on the CO_2 inventory, is consistent with that derived using P-3 measurements and the CO inventory, further supporting both our assessment of uncertainties in the CARB CO and CO_2 inventories, and our assumption of sampling well-mixed emissions in the SoCAB, since any outlying CH_4 data do not affect the overall emission estimates significantly.

The derived 2010 top-down SoCAB CH_4 emission of 410 and 440 Gg CH_4/yr reported here using the CARB CO or CO_2 inventories, respectively, are in quantitative agreement, in contrast to that reported for 2008 [*Wunch et al.*, 2009]. The 2010 estimates are a factor of 1.35 to 1.45 greater than the modified population-apportioned 2009 CARB GHG inventory value of 301 Gg CH_4/yr (Table 2). A concurrent inverse modeling study by *Brioude et al.* [2012] has found no statistical difference between the total SoCAB CO emissions reported by CARB for 2010 and a top-down approach that estimated CO emissions in the SoCAB region using the same CO measurements used in this paper. For this reason, and for consistency with published works [*Wunch et al.*, 2009; *Hsu et al.*, 2010; *Wennberg et al.*, 2012], we use 410 ± 40 Gg CH_4/yr from the top-down CH_4 assessment based on 2010 P-3 measured CH_4/CO enhancement ratios and the CARB CO inventory for the remainder of our analysis.

4.2. Methane emissions from L.A. basin landfills

Landfills are the largest non-fossil fuel CH_4 emission source in the bottom-up inventories compiled by *Hsu et al.* [2010] and by *Wennberg et al.* [2012], but these two studies disagree on the magnitude of this source. *Hsu et al.* [2010] estimated annual emissions from landfills totaled 90 Gg CH_4/yr from the Los Angeles County portion of the South Coast Air Basin. *Wennberg et al.* [2012] reported landfill emissions of just 86 Gg CH_4/yr for the entire South Coast Air Basin. However, that number is too low due to an error in their gridded landfill emissions inventory [*P. Wennberg,* personal communication, 2012] and is discarded in the following analysis.

In the CARB GHG inventory, CH_4 emissions are calculated for individual landfills using methods prescribed by the IPCC and summed over all landfills to estimate a statewide total. Annual CH_4 emission values for individual landfills were obtained directly from CARB [*L. Hunsaker,* personal communication, 2011] to facilitate direct comparison to the P-3 data from CalNex. We use the P-3 data to calculate emissions from two of the largest CH_4-emitting landfills in the statewide GHG inventory, both of which are located in the SoCAB.

The first landfill results we examine are from the Olinda Alpha landfill (33.934° N, 117.841° W) in Brea, Orange County, California. The NOAA P-3 flew five daytime boundary-layer transects on five different days downwind of this landfill (Figure 3), and a CH_4 emission flux was determined for each transect using equation (1). The results are summarized in Table 3.

For the three transects when both the WS-CRDS and QCLS CH_4 instruments were sampling ambient air, flux determinations using these independent CH_4 measurements agreed within 3%. In these cases, the flux was averaged and reported in Table 3. Three nearby CH_4 point sources are identified in the 2009 CARB GHG inventory: an oil and gas field power plant, which burns natural gas for fuel; the landfill power plant at Olinda Alpha, which burns landfill gas for fuel; and general stationary combustion from the landfill operations. Inventory data suggest that these three sources together emit between 0.0004 and 0.0015 Gg CH_4/yr, negligible amounts relative to CH_4 emitted directly from the landfill. On 19 May, the NOAA P-3 sampled plumes from the nearby oil and gas power plant and the landfill's power plant, both of which burn natural gas as fuel (Figure 3c). A large spike in CO_2, some CH_4, and perhaps a small amount of CO were encountered in the landfill power plant plume. However, downwind of the landfill in the large plume of CH_4, the CO_2 enhancement does not stand out significantly above the background variability. Therefore, our analysis of P-3 data supports the conclusion from the inventory that landfill CH_4 emissions dominate the observed plume enhancements downwind of Olinda Alpha landfill. Using NOAA P-3 CH_4 data from all five transects, we directly calculate a weighted average CH_4 emission flux via equation (1) of $(1.49 \pm 0.35) \times 10^{25}$ molecules/s, equal to 12.5 ± 2.9 Gg CH_4/yr assuming a constant emission, where the weights are the 50% uncertainty of each determination. For comparison, the CARB GHG inventory emission estimate from the Olinda Alpha landfill is 11.0 Gg/yr for 2008, showing agreement within the errors of the direct estimate using P-3 airborne data.

The second landfill results we examine in-depth are from the Puente Hills landfill (34.020° N, 118.006° W) in City of Industry, Los Angeles County, California. Of all California landfills, Puente Hills is the largest emitter of CH_4 in the 2008 CARB GHG inventory.

Nearby sources of CH_4 in the 2008 CARB GHG inventory include the Puente Hills power plant (0.00045 Gg CH_4/yr) and the Savage Hills Canyon landfill (1.1 Gg CH_4/yr), both of which are small relative to the CARB GHG inventory of 39 Gg CH_4/yr emission rate for Puente Hills. The NOAA P-3 conducted three daytime boundary layer plume transects from which we determine an average emission flux of $(4.06 \pm 1.18) \times 10^{25}$ molecules/s, which extrapolates to 34.0 ± 9.9 Gg CH_4/yr assuming a constant emission (Table 3). Similar to the findings for Olinda Alpha, the CARB GHG inventory of 39 Gg CH_4/yr for the Puente Hills landfill is in agreement within the errors of the direct estimate using P-3 airborne data.

Quantitative agreement between CH_4 flux estimates from the NOAA P-3 and the 2008 CARB GHG inventory for these two examples supports the use of that inventory to quantify total CH_4 emissions from landfills in the South Coast Air Basin. According to the 2008 CARB GHG inventory, CH_4 emissions from landfills totaled 117 Gg CH_4/yr in the L.A. County portion of the SoCAB, 30% higher than the 90 Gg CH_4/yr for the same geographic area using the CARB GHG inventory in 2008 reported by *Hsu et al.* [2010], which we attribute to different versions of the CARB GHG inventory.

The 2008 CARB GHG inventory further predicts an emission from landfills of 164 Gg CH_4/yr for the entire SoCAB. On the basis of the agreement with the CARB inventory described above for the emission rates from the two landfills quantified directly by the CalNex P-3 data (50 Gg CH_4/yr, or 30% of the inventory total for the SoCAB), we assume the remaining CARB landfill CH_4 emission estimates are accurate.

4.3. Methane emissions from L.A. basin dairies

Salas et al. [2008] published dairy locations in California for the year 2005, with an estimate of dairy cow population for each. The locations are plotted as filled yellow circles in Figure 1c, and sized by the expected CH_4 emission from enteric fermentation according to the 2009 CARB GHG inventory (144 kg CH_4 per cow per year). According to *Salas et al.* [2008], all dairies in San Bernardino and Riverside counties were also located in the SoCAB, and 87% of the dairy cows in the SoCAB in 2005 were located in the Chino area (the large grouping of dairies in Figure 1c). The Chino-area dairy operations, which at one time were distributed across the Riverside-San Bernardino county line in satellite images, now appear to be located mainly in San Bernardino County as the Riverside dairies have been converted to residential neighborhoods (*e.g.*, see Google Earth historical imagery since 2000). This declining number of dairies is confirmed by the United States Department of Agriculture (USDA) (http://www.nass.usda.gov/Statistics_by_State/California/Publications/County_Estimates/201005lvscef.pdf), which reports a decrease in dairy cows in San Bernardino and Riverside Counties from 200,000 head in 2005 to 137,500 head in 2010. In addition to dairy cows, dairies also stock immature heifers. Further, there are beef operations in the SoCAB, but these are negligible compared to the San Bernardino and Riverside dairy populations. According to the USDA, there were a total of 431,000 cattle in San Bernardino and Riverside counties in 2005, and 295,000 cattle in 2010. For both years, dairy cows represented approximately 46.5% of the cattle population in the SoCAB. From these dairy and cattle populations, we construct a bottom-up emissions inventory for the SoCAB using the same emission factors as the CARB GHG inventory.

We begin with CH_4 emissions from enteric fermentation. We assign to each of the 137,500 dairy cows in the SoCAB an emission factor of 144 kg CH_4/yr. We assume the remaining 157,500 head are dairy replacements, and assign each an emission factor of 57.7 kg CH_4/yr, or the average emission factor for 0–1 and 1–2 year old dairy replacements in the CARB GHG inventory. We calculate a total of 28.9 Gg CH_4/yr emitted solely from enteric fermentation in the SoCAB.

In addition to enteric fermentation, manure management practices have a substantial effect on CH_4 emissions from livestock operations. In the L.A. basin, dairies typically practice solid storage (http://www.aqmd.gov/rules/doc/r1127/pr1127_task1rpt_20020101.pdf and http://www.arb.ca.gov/planning/sip/sjv_report/addtl_resources.pdf), which emits relatively low levels of CH_4 (17 kg/yr per cow) according to the 2009 CARB GHG inventory. The tradeoff for this practice is that it emits larger amounts of NH_3 than other types of manure management (http://www.epa.gov/ttn/chief/ap42/ch09/draft/draftanimalfeed.pdf). Therefore, if we attribute dry manure management emissions to the SoCAB dairy cow population, and the dry lot emission rate of 2.1 kg CH_4/yr for the remaining heifers, we get an additional 2.7 Gg CH_4/yr from dairy operation manure management in the SoCAB. This results in a total of 31.6 Gg CH_4/yr from enteric fermentation and manure management for the SoCAB dairy operations. This is the emission from agriculture and forestry that we add back into the population-apportioned CARB CH_4 inventory above (Table 2).

Our estimate of 31.6 Gg CH_4/yr, based on inventory data, is less than half of the 76 Gg CH_4/yr estimated by *Wennberg et al.* [2012]. We attribute this difference in bottom-up inventories to the different assumptions of manure management practices.

Wennberg et al. [2012] scaled total California CH_4 emissions by livestock population, which also assumes the manure management practices from the San Joaquin Valley apply to the L.A. basin. For example, the anaerobic lagoons more commonly used in the San Joaquin Valley emit 325 kg CH_4 per cow per year according to the 2009 CARB GHG inventory, significantly higher than 17 kg CH_4 per cow per year from dry manure management practices typical of the L.A. basin.

Nowak et al. [2012] used P-3 data from CalNex to derive emissions of ammonia (NH_3) from dairy farms in the Chino area. From NOAA P-3 measurements, we determine a CH_4 flux from the Chino-area dairies for the same three downwind transects analyzed by *Nowak et al.* [2012]. Using the Chino to SoCAB population apportionment by *Salas et al.* [2008], we expect these same Chino-area dairies to emit approximately 28 Gg CH_4/yr. CH_4 fluxes determined from equation (1) range from 24 ± 12 to 88 ± 44 Gg CH_4/yr, and the average of the three transects is 49 ± 25 Gg CH_4/yr. This value derived from airborne flux determination lies between the 28 Gg CH_4/yr calculated from the inventory assuming dry manure management practices described above, and the estimate by *Wennberg et al.* [2012] of 76 Gg CH_4/yr (less livestock emissions from the SoCAB that are not in the Chino area) assuming mainly wet management practices. We attribute the differences to actual practices in the region, which are likely a mixture of the two manure management approaches. Satellite images of the area show what appear to be several anaerobic lagoons near Chino, California. Our flux determination is therefore consistent with our bottom-up CH_4 emission inventory, with room for a mixture of manure management practices, including some anaerobic lagoons, in the L.A. basin.

4.4. Spatial distribution of methane sources

Townsend-Small et al. [2012] concluded that the CH_4 emissions in the L.A. region had a stable isotope ratio similar to that of fossil-fuel CH_4. This conclusion was based on measurements made at the Mt. Wilson Observatory. A back-trajectory [*White et al.*, 2006; http://www.esrl.noaa.gov/psd/programs/2010/calnex/traj/] from MWO for 5 August 2009, the specific day that *Townsend-Small et al.* [2012] used to determine the excess CH_4 stable isotopic ratio, shows the prevailing winds to MWO were from the southwest, or from downtown L.A. and the coast west of downtown L.A. The trajectory tool also shows winds from the eastern basin on the previous day, which was excluded by *Townsend-Small et al.* [2012] due to lower correlation between the excess CH_4 and $\delta^{13}C$. We conclude that the MWO data interpreted by *Townsend-Small et al.* [2012] were dominated by emissions from the western basin only, and were not influenced by emissions from either the largest landfills (Puente Hills and Olinda Alpha), or from the dairies in the eastern part of the L.A. basin. This spatially-biased sampling is consistent with their conclusion that landfills do not contribute significantly to the total atmospheric CH_4 burden in L.A.

Evidence for the heterogeneous spatial distribution of CH_4 sources in the SoCAB can be seen in the NOAA P-3 data. Figure 4 shows that the correlation of ethane with CH_4 is dependent on the sample location in the L.A. basin. Also shown in Figure 4 is the slope used by *Wennberg et al.* [2012] to represent the ethane/CH_4 ratio (16.5 ± 2.5 ppt ethane/ppb CH_4) in pipeline-quality dry natural gas from the Southern California Gas Company (SoCalGas), the major provider of natural gas to the SoCAB, for 2010.

The chemical data in Figure 4 reflect the known source types shown on the map in Figure 1c: the large CH_4 sources in the eastern L.A. basin, primarily landfills and dairies, are not significant sources of ethane relative to CH_4.

We can reconcile the conclusions of *Townsend-Small et al.* [2012] and *Wennberg et al.* [2012] with the CARB GHG inventory by noting that fossil fuel CH_4 emissions predominate in the western basin, and that landfill and livestock CH_4 emissions predominate in the eastern basin. However, in contrast to the findings of *Wennberg et al.* [2012], we find that natural gas leaks from the SoCalGas and in-home pipelines are not the only possible source of fossil fuel CH_4 to the western basin, as described below.

4.5. Light alkane emissions from local natural gas production

Los Angeles was one of only three out of 28 cities characterized by propane and ethane levels within 10% of one another in the atmosphere [*Baker et al.*, 2008], consistent with an enhanced propane source term in L.A. Figure 5 shows correlations of propane vs. ethane in whole-air samples from various aircraft projects in the Los Angeles region (ITCT 2002, ARCTAS 2008, and CalNex 2010), as well as measurements from the CalNex Pasadena ground site in 2010. Also plotted are lines representing the composition ratios of other possible sources of ethane and propane in Los Angeles.

The L.A. basin is home to oil and gas operations (see Figure 1c); the composition ratios depicting possible emissions from local natural gas (gray lines) and local geologic seeps (salmon lines) in Figure 5 are those reported by *Jeffrey et al.* [1991]. The lower propane content relative to ethane seen in the seeps (*e.g.*, the La Brea tar pits) compared to the local natural gas is attributed to near-surface microorganisms forming shorter-chain alkanes from longer-chain alkanes during the time the natural gas migrates toward the surface [*Jeffrey et al.*, 1991]. The average propane/ethane ratio for processed gas in SoCalGas pipelines [*Wennberg et al.*, 2012] is plotted as a dashed black line. Pipeline-quality dry natural gas has a low propane/ethane ratio because the natural gas has been processed (*i.e.*, the higher alkanes have been removed from the natural gas) before distribution. The SoCalGas ratio is representative of natural gas piped in from out of state (*e.g.*, from Texas, Wyoming, and Canada); approximately 90% of natural gas used in California is imported (http://www.socalgas.com/regulatory/documents/cgr/2010_CGR.pdf). The on-road emissions are taken from a San Francisco Bay-area tunnel study by *Kirchstetter et al.* [1996], who reported a vehicular emission ratio of 0.13 mol propane/mol ethane roughly similar to those by *Fraser et al.* [1998] (0.27 mol propane/mol ethane) and by *Lough et al.* [2005] (0.06 – 0.18 mol propane/mol ethane). Vehicle engine exhaust typically contains small, decreasing amounts of CH_4, ethane, and propane due to incomplete combustion, as gasoline and diesel fuel do not contain significant amounts of these light alkanes. The on-road emissions, local geologic seeps, and the pipeline-quality dry natural gas from SoCalGas contain 3–5 times more ethane than propane, and therefore cannot alone explain the ambient ratios measured in the L.A. basin. The propane and ethane composition of unprocessed natural gas from local wells, on the other hand, closely matches the SoCAB ambient measurements from three aircraft campaigns, the CalNex ground site measurements, and the *Baker et al.* study [2008]. Propane and ethane were also typically enhanced at the same time, with the exception of one sample with elevated propane near the Long Beach area (Figure 1e).

The data in Figure 5 suggest that local oil and gas wells contribute significantly to the atmospheric propane burden in the SoCAB. However, *Wennberg et al.* [2012] invoked a large source of propane from fugitive losses from the liquefied petroleum gas (LPG) industry (*i.e.*, propane tanks), in addition to leaks from the pipeline-quality dry natural gas distribution system in the L.A. basin. This would be consistent with past works that have found significant fugitive losses of propane in other cities, such as Mexico City [*Blake and Rowland*, 1995]. We therefore extend our analysis to incorporate ethane, propane, and C_4 (*n*- and *i*-butane) and C_5 (*n*- and *i*-pentane) isomers to better attribute and quantify the sources of light alkanes and CH_4 to the SoCAB atmosphere. Light alkanes are plotted in Figure 6, with lines depicting the composition of natural gas in SoCalGas pipelines [*Wennberg et al.*, 2012] and of on-road emissions [*Kirchstetter et al.*, 1996]. We neglect chemical processing of these long-lived alkanes ($\tau \geq 3$ days at OH = 1×10^6 molecules/cm^3) as we find no detectable difference between daytime and nighttime enhancement ratios relative to CO, similar to the findings of *Borbon et al.* [2013] for *n*-butane and CO at the CalNex Pasadena ground site. Atmospheric enhancement ratios of propane, *n*-butane, and *i*-butane (Figures 6b–d) relative to ethane are consistent with emissions having the composition of local natural gas [*Jeffrey et al.*, 1991]. On-road emissions do not appear to contribute significantly to the CH_4, ethane, and propane in the L.A. atmosphere, and pipeline-quality dry natural gas and/or local geologic seeps do not appear to contribute significantly to the propane and *n*-butane relative to ethane in the L.A. atmosphere. Based on these observations, we conclude that the local natural gas industry contributes a significant fraction to the total atmospheric C_2-C_4 alkane abundances, including propane, in the L.A. basin. We infer CH_4 emissions from the local natural gas industry are non-negligible as well, as discussed below.

4.6. Source Attribution

Here we quantify total emissions of C_2–C_5 alkanes in the L.A. basin by multiplying their observed enhancement ratios to CO by the CARB SoCAB emission inventory for CO. Figure 7 shows C_2–C_5 alkanes plotted versus CO with their respective ODR fits. The slopes from these fits are used in equation (2) along with the projected 2010 CARB CO inventory to calculate annual alkane emissions in the SoCAB. We assume the slopes represent a direct emission with no chemical aging. These emissions are listed in the right-most column of Table 4. Also listed in Table 4 are the estimated contributions from mobile sources in the SoCAB, using C_1–C_5 to CO emission ratios from *Kirchstetter et al.* [1996] (modified as discussed below) and CO emissions from the mobile sources category in the projected 2010 CARB CO inventory, equal to 920 Gg CO/yr, in equation (2).

Wennberg et al. [2012] attributed the inventory CH_4 shortfall [*Wunch et al.*, 2009; *Hsu et al.*, 2010] by ascribing much of the CH_4 and ethane enhancements to fugitive losses of processed pipeline-quality dry natural gas. They further suggest the majority of atmospheric propane is due to LPG industry/propane tank fugitive losses. Here, we consider other possible explanations of the sources of CH_4 and light alkanes in the L.A. basin for the following two reasons. First, the source attribution by *Wennberg et al.* [2012] leaves little room for CH_4 emissions from landfills, wastewater treatment plants, and dairies in the L.A. basin. This solution seems unlikely based on direct emissions flux estimates using the P-3 data downwind of landfills and dairies in the SoCAB, as described above. Second, the attribution by *Wennberg et al.* [2012] would leave a shortfall in both *n*- and *i*-butane emissions that cannot be explained by gasoline evaporation or emissions from mobile sources.

We use a multivariate approach based on a linear combination of the CH_4 and light alkane compositions from known sources in order to attribute and quantify total CH_4 and C_2–C_5 alkane emissions in the South Coast Air Basin.

We include 7 different source types (sectors) with distinct and known CH_4 and C_2–C_5 alkane compositions (Figure 8) in the following analysis: 1) Leaks of processed dry natural gas from pipelines, and/or emissions from local geologic seeps (this approach cannot distinguish between pipeline-quality dry natural gas and local seeps); 2) CH_4-dominated emissions, such as from landfills, wastewater treatment plants, and dairies; 3) Leaks of unprocessed, local natural gas; 4) Leaks of liquefied petroleum gas from propane tanks; 5) On-road combustion emissions from mobile sources; 6) Emissions of CH_4 and C_2–C_5 alkanes in the SoCAB from other source sectors; and 7) Evaporative emissions from gasoline. These are described briefly below.

1. The South Coast Air Basin contains 14.8 million people, and SoCalGas delivers approximately 11 Tg/yr of natural gas to the Los Angeles area. Additionally, the Earth's natural degassing is a known source of CH_4, ethane, and propane to the atmosphere [*Etiope et al.*, 2008; *Etiope and Ciccioli*, 2009], and the L.A. basin contains abundant geologic hydrocarbon reserves [*Jeffrey et al.*, 1991]. We group fugitive losses from processed pipeline-quality dry natural gas with the emissions from local geologic seeps because the C_1–C_4 emissions from these sources are not sufficiently different to be treated separately in our linear combination analysis (illustrated by the similarity in slopes of the dashed black and salmon-colored lines in Figure 6).

Both pipeline-quality dry natural gas and local seep emissions contain similar amounts of CH_4 and ethane relative to one another, and have less C_3–C_5 alkanes relative to ethane than local, unprocessed natural gas. For pipeline-quality dry natural gas, most C_{3+} alkanes are removed during the processing stage, which is typically done close to the source, which for ~90% of the natural gas used in California is in Canada, Wyoming, and/or Texas. For local seeps, most C_{3+} alkanes are either preferentially adsorbed in shallow sediments compared to CH_4, or biodegraded by microbes in the earth's crust during the seepage of local natural gas to the surface [*Jeffrey et al.*, 1991]. We use SoCalGas samples of pipeline-quality natural gas from 2010 [*Wennberg et al.*, 2012] to represent this source, and estimate the uncertainty of the composition at 15%.

2. CH_4-dominant emission sources, which for this analysis include landfills, wastewater treatment plants, and livestock, emit CH_4 but no significant amounts of C_2–C_5 alkanes. This is represented in our analysis as a unit vector containing only CH_4.
3. From 2007–2009, the oil and gas industry in the L.A. basin produced roughly 12–13 billion cubic feet of natural gas per year, mostly associated gas from oil wells (http://www.conservation.ca.gov/dog/pubs_stats/annual_reports/Pages/annual_reports.aspx). We use an average of the samples reported by *Jeffrey et al.* [1991] weighted by 2009 gross natural gas production per field, and estimate the uncertainty of this composition at 25%.
4. Two types of LPG are sold in the Los Angeles area: one is almost completely composed of propane, the other has traces of *n*- and *i*-butane (http://www.arb.ca.gov/research/apr/past/98-338_1.pdf).

We use the ratios reported by *Blake and Rowland* [1995] from direct analysis of LPG in Los Angeles, which is consistent with an average of the two types of LPG sold in L.A, and estimate the uncertainty of the composition at 10%.

5. On-road combustion emissions are modified from the work of *Kirchstetter et al.* [1996] by multiplying emission ratios of alkanes to CO by the 925 Gg CO/yr from on-road sources in the projected 2010 CARB CO inventory. The C_4–C_5 emissions represent unburned fuel and are typically proportional to the fuel composition; the C_1–C_3 emissions typically represent incomplete combustion products. To account for differing fuel compositions since the time of the *Kirchstetter et al.* [1996] study, the *i*- and *n*-butane emissions calculated for mobile sources in the SoCAB (Table 4) have been scaled to the *i*-pentane emissions based on their relative abundance in gasoline [*Gentner et al.*, 2012].
6. There are additional sources of light alkanes in the SoCAB. We use the 2010 CARB speciated inventory for total organic gases (http://arb.ca.gov/ei/speciate/interopt10.htm) and projected 2010 total organic gas emissions (http://www.arb.ca.gov/app/emsinv/fcemssumcat2009.php) for the SoCAB to estimate emissions of light alkanes not specified in other source sectors. These include emissions from aerosol spray cans and other consumer products, coatings and solvents, adhesives and sealants, and fiberglass and plastics manufacturing. For example, propane, *n*- and *i*-butane are commonly used as propellants in aerosol spray cans, having replaced CFCs in the United States in the 1970s (*e.g.*, CARB estimates 0.6 Gg of aerosol antiperspirant vapors were emitted to the SoCAB in 2010, of which 0.14 Gg, 0.03 Gg, and 0.15 Gg were propane, *n*-, and *i*-butane, respectively).

These emissions are summed and listed in the "CARB other" column in Table 4. Emissions from natural gas leaks, petroleum refining, petroleum marketing (gas stations), landfills and composting, and mobile sources are not included in these totals, because they are accounted for elsewhere in other source sectors. We estimate a 25% uncertainty in the "CARB other" inventory.

7. Emissions ratios from evaporated gasoline were calculated from ten gasoline samples from five Pasadena gas stations in the summer of 2010, weighted by estimated sales of 80% regular and 20% premium [*Gentner et al.*, 2012]. Uncertainties are those reported by *Gentner et al.* [2012].

First, we start with estimated annual C_1–C_5 emissions in the SoCAB (right-most column of Table 4), then subtract modified on-road emissions [*Kirchstetter et al.*, 1996] and projected emissions of C_1–C_5 alkanes from other sources (source sector 6, above). Next, we place the remaining source sector characteristics into a matrix and solve for the fraction each source contributes to the remaining alkane observations for the L.A. basin based on each source's relative abundances of various light alkanes. The matrix has five columns representing the five remaining source sectors, and seven rows containing C_1–C_5 alkanes. We solve the equation [*e.g.*, see §4.2 *Kim et al.*, 2011]

$$\mathbf{A}_{i,j}\,\mathbf{x}_j = \mathbf{b}_i \qquad (3)$$

where $\mathbf{A}_{i,j}$ is a matrix of the C_1–C_5 alkane composition, *i*, for the source sectors, *j*, defined above; x_j is the fraction each source contributes to the total observed emissions, and b_i is the total observed emission of alkane *i* minus the contributions from the mobile and "other" source sectors (Table 4). The columns of the matrix **A** are proportional to the first five columns of

Table 4. We use LAPACK (http://www.netlib.org/lapack/) to solve for the linear least squares solution that minimizes ($\mathbf{Ax} - \mathbf{b}$). Uncertainties in the derived $\mathbf{x}_j$ are estimated by a sensitivity study, where we run the solution 1,000,000 times by randomly varying $\mathbf{A}_{i,j}$ and $\mathbf{b}_i$ according to their estimated uncertainties, then use the standard deviation of the 1,000,000 $\mathbf{x}_j$ determinations to estimate the uncertainty in the source attribution fraction. The source attribution fractions and their uncertainties are multiplied by the total estimated SoCAB emission for each alkane, then are summed with the uncertainties added in quadrature. CH_4 and C_2–C_5 alkane emissions totals, their uncertainties, and the contributions from each source type are given in Table 4. The source attribution solution solves the observed SoCAB alkane emission to within each alkane's emission uncertainty.

Our modeled source attribution differs from the alkane source distribution in the L.A. basin as set forth by *Wennberg et al.* [2012]. From a total calculated source of 410 ± 40 Gg CH_4/yr in the SoCAB, we determine that 47% comes from leaks of processed pipeline-quality dry natural gas and/or from local geologic seeps; 44% of the CH_4 comes from the sum of landfill, wastewater treatment, and dairy emissions; 8% from the leaks of unprocessed natural gas from production in the western L.A. basin; and 1% from mobile sources. The attribution is presented graphically in Figure 8. Figure 8a displays the total SoCAB emissions as a black horizontal line in each panel, with contributions from the different source sectors given below the line by the filled bars. Figure 8b shows the proportion that each source sector contributes to the derived total emissions of each alkane.

Our analysis attributes CH_4 emissions of 192 ± 54 Gg CH_4/yr to leaks of pipeline-quality dry processed natural gas and/or leaks from local geologic seeps, but does not distinguish further between these two different sources. This value is nearly a factor of 5 greater than the population-apportioned 2009 CARB GHG emissions inventory estimate of 40 Gg CH_4/yr lost from natural gas pipelines in the SoCAB. Our estimate of 192 Gg CH_4/yr is less than the maximum emission of 400 ± 150 Gg CH_4/yr estimated by *Wennberg et al.* [2012]. Our estimate would represent approximately 2% of the natural gas delivered to customers in the SoCAB and, including storage and deliveries to customers outside the SoCAB, 1% of the gas flowing into the basin [*Wennberg et al.*, 2012]. These percentages would decrease linearly with any CH_4 emissions attributed to local geologic seeps. *Farrell et al.* [*in press*, 2012] estimate up to 55 Gg CH_4/yr are emitted from the La Brea Tar Pits in western L.A. County alone; if accurate, this would imply pipeline leaks of only 0.7% of the gas flowing into the basin, or a factor of at least two lower than the 2% proposed by *Wennberg et al.* [2012].

Our analysis attributes 182 ± 54 Gg CH_4/yr in the SoCAB to emissions from landfills, wastewater treatment, and dairies. SoCAB landfills account for 164 Gg CH_4/yr in the 2008 CARB GHG inventory, a value supported by our analysis in section 4.2. In section 4.3, we estimated in a bottom-up inventory that SoCAB dairies emitted 31.6 Gg CH_4/yr. *Wennberg et al.* [2012] estimated an emission of 20 Gg CH_4/yr from wastewater treatment. These independent estimates sum to 216 Gg CH_4/yr and are consistent with our source apportionment using NOAA P-3 data.

CH_4 emissions of 31.9 ± 6.5 Gg CH_4/yr are ascribed to leaks of local, unprocessed natural gas, and would represent 17% of the local production in 2009, the latest year for which data are available (http://www.conservation.ca.gov/dog/pubs_stats/annual_reports/Pages/annual_reports.aspx). This number assumes a CH_4 composition of 72.5% by volume for natural gas produced in the South Coast Air Basin, which is calculated as an average from the samples reported by *Jeffrey et al.* [1991] weighted by 2009 production. Our derived value of 17%, although a surprisingly high amount of local production, is consistent with a nascent bottom-up estimate under way at CARB. A new bottom-up inventory survey, conducted by CARB for the calendar year 2007 but not yet incorporated into the official GHG inventory, indicates that 109 Gg CH_4/yr, since revised to 95.5 Gg CH_4/yr [*S. Detwiler,* personal communication, October 2012], were emitted throughout California by the oil and gas industry via combustion, venting, and fugitive losses (table 3-1, http://www.arb.ca.gov/cc/oil-gas/finalreport.pdf). This updated value is a factor of 2.5 larger than the current CARB GHG inventory tabulation of 38 Gg CH_4/yr from oil and gas extraction for 2007 in California. CH_4-specific emissions for the South Coast Air Management District in the new CARB survey report show 24.6 Gg CH_4/yr were emitted in the SoCAB [S. Detwiler, personal communication, October 2012]. According to the survey, emissions in the SoCAB accounted for 26% of the revised statewide total oil and gas operations CH_4 emission in 2007, despite accounting for only 4.4% of statewide natural gas production in the basin that year (http://www.conservation.ca.gov/dog/pubs_stats/annual_reports/Pages/annual_reports.aspx). Thus, the survey responses suggest a CH_4 leak rate of 12% of local production in the L.A. basin.

Thus, our estimate of CH_4 emissions from local natural gas for 2010 based on P-3 data from CalNex is within a factor of 1.5 of the CARB bottom-up inventory currently in development based on the 2007 survey. According to the survey, other oil and gas producing regions in California show smaller CH_4 loss rates than that from the SoCAB. For instance, statewide losses of CH_4 represent approximately 2.1% of statewide production, and CH_4 losses from the San Joaquin Air Quality District represent approximately 1.4% of production (from Oil and Gas Districts 4 and 5). This indicates that losses from natural gas production are proportionally larger in the L.A. basin than elsewhere in the State of California.

A propane emission of 6.6 ± 2.9 Gg/yr from LPG/propane tanks would represent approximately 1% of sales (http://www.aqmd.gov/ceqa/documents/2012/aqmd/finalEA/PAR1177/1177_FEA.pdf), which is less than the ~4% calculated by *Wennberg et al.* [2012], and closer to the 0.6% estimated from the document cited.

Finally, our analysis suggests a resolution to the discrepancies noted above between previous top-down assessments and the bottom-up inventory calculations for CH_4 in the SoCAB [*e.g.*, *Wunch et al.*, 2009; *Hsu et al.*, 2010; *Townsend-Small et al.*, 2012; *Wennberg et al.*, 2012]. We conclude the most probable source for the excess atmospheric CH_4 is likely due to a combination of primarily leaks, not accurately represented in the current CARB GHG inventory, from natural gas pipelines and urban distribution systems and/or from local geologic seeps, and secondarily leaks of unprocessed natural gas from local oil and gas production centered in the western L.A. basin.

This finding is based on the characteristic enhancement ratios of CH_4 and the various C_2–C_5 alkanes consistently observed in the L.A. atmosphere, and is further supported by the spatial information provided by P-3 samples during CalNex. Finally, the updated values for local oil and gas industry emissions in the recent GHG survey commissioned by CARB, when incorporated fully into the official CARB GHG record, will likely help to reduce this long-standing discrepancy between top-down assessments and bottom-up inventories.

5. Conclusions

We use aircraft measurements of CH_4, CO, and CO_2 during the CalNex field campaign to show that emissions of CH_4 to the L.A. basin are greater than can be explained by official state bottom-up inventories apportioned by population, consistent with published work. The ratio of the CARB CO and CO_2 inventories is in better agreement with our measurements of CO/CO_2 in the Los Angeles atmosphere than was the case for the analysis by *Wunch et al.* [2009], which we attribute either to improved CARB inventories, the present use of a basin-wide data set to determine basin-wide emission ratios, or both.

From crosswind plume transects downwind of the two largest landfills in the basin, we determine CH_4 fluxes that are consistent with the 2008 CARB GHG inventory values, which total 164 Gg CH_4/yr emitted from all landfills in the South Coast Air Basin. CH_4 emission fluxes were also determined for Chino-area dairies in the eastern L.A. basin. Flux estimates from these dairies ranged from 24 ± 12 to 87 ± 44 Gg CH_4/yr, and the average flux is consistent with a revised bottom-up inventory originally compiled by *Salas et al.* [2008] and with previous inventory estimates [*Wennberg et al.*, 2012].

Finally, we present a top-down assessment of C_2–C_5 alkane sources in the L.A. basin, then apportion CH_4 and the C_2–C_5 alkanes to specific source sectors in the region. Using this source apportionment approach, we estimate that 32 ± 7 Gg of CH_4/yr, or 8% of the total CH_4 enhancement observed in the SoCAB during CalNex, came from the local oil and gas industry. This number represents approximately 17% of the natural gas produced in the region, within a factor of 1.5 of that calculated from a recent survey that will be used to update the CARB bottom-up inventory. We estimate 182 ± 54 Gg CH_4/yr are emitted by landfills, dairies, and wastewater treatment, which is consistent with bottom-up inventories, and 192 ± 54 Gg CH_4/yr are emitted of processed pipeline-quality dry natural gas and/or from geologic seeps in the region. We further conclude that leaks of processed pipeline-quality dry natural gas and/or local geologic seeps, and unprocessed natural gas from local oil and gas production are the most likely major contributors to the previously noted discrepancy between CH_4 observations and State of California inventory values for the South Coast Air Basin. Our findings suggest that basin-wide mobile studies targeting CH_4 and C_2–C_5 alkane emissions from natural gas pipelines and urban distribution systems, geologic seeps, and local oil and gas industry production sites would be useful to further distinguish the sources of CH_4 in the L.A. basin.

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Table 1. Summary of past studies investigating CH_4 emissions in the L.A. basin.

Study	Time of study	Geographic area	Percentage of California population in geographic area	CH_4 Emission (Gg/yr)	Inventory referenced	Bottom-up CH_4 emission inventory (Gg/yr)
Wunch et al. [2009]	August 2007 – June 2008	SoCAB	43%	400 ± 100	CARB CO 2007	260[b]
				600 ± 100	(CARB CO_2 2006 + EDGAR CO_2 2005)/2	
Hsu et al. [2010]	April 2007 – May 2008	L.A. County ∩ SoCAB	27%	200 ± 10	CARB CO 2007	140
Wennberg et al. [2012]	April 2007 – May 2008	SoCAB	43%	380[a] ± 100	CARB CO 2007	---
	June 2008	SoCAB	43%	470 ± 100	CARB CO 2008	---
	May 2010 – June 2010	SoCAB	43%	440 ± 100	CARB CO 2010	---

[a] *Wennberg et al.* [2012] recalculated the data reported by *Hsu et al.* [2010] to estimate a CH_4 emission from the entire SoCAB.

[b] *Wunch et al.* [2009] apportioned the statewide CARB GHG inventory for CH_4, less agriculture and forestry emissions, by population

Table 2. Inventories used in current analysis

Emission	Inventory	Year	Geographic Area
180 Tg CO_2/yr	CARB GHG[a]	2009	SoCAB[c]
979 Gg CO/yr	CARB[b]	2010	SoCAB[c]
301 Gg CH_4/yr	CARB GHG[a]	2009	SoCAB[c]

[a] 2009 CARB CO_2 and CH_4 emissions (http://www.arb.ca.gov/cc/inventory/data/data.htm)

[b] projected 2010 CARB CO emissions (http://www.arb.ca.gov/app/emsinv/fcemssumcat2009.php)

[c] statewide inventory apportioned by SoCAB population

Table 3. Landfill emission fluxes determined aboard the NOAA P-3 in 2010 from downwind plume transects.

Landfill	Transect Date	Flux, 10^{25} molecules/s	Flux, Gg/yr	2008 CARB GHG inventory,[a] Gg/yr
Olinda Alpha	8 May	1.13	9.5	
	14 May	1.45	12.2	
	16 May	1.74	14.6	
	19 May	1.61	13.5	
	20 June	2.90	24.3	
	average[b]	**1.49 ± 0.35**	**12.5 ± 2.9**	**11.0**
Puente Hills	8 May	4.29	36.0	
	19 May	3.62	30.4	
	20 June	4.48	37.6	
	average[b]	**4.06 ± 1.18**	**34.0 ± 9.9**	**38.8**

[a] data from CARB [L. Hunsaker, personal communication, June 2011]

[b] weighted average, assuming a 50% uncertainty in the individual flux determinations [*Taylor*, 1997]

Table 4. Derived emissions in the South Coast Air Basin (in Gg/yr) for 2010 from each source sector used in linear analysis.

	Pipeline-quality dry NG/ local seeps	CH_4-dominant (landfills, dairies, etc.)	Local NG	LPG/propane	Evaporated gasoline	Mobile sources	CARB other	Summed source totals	Estimated SoCAB total[a]
CH_4	192 ± 54	182 ± 54	32 ± 7	---	---	4.9 ± 1.3	1.2 ± 0.3	411 ± 77	411[b] ± 37
ethane	5.9 ± 1.7	---	4.5 ± 1.0	0.05 ± 0.02	0.0 ± 0.0	0.6 ± 0.1	0.3 ± 0.1	11.4 ± 1.9	11.4[b] ± 1.6
propane	1.5 ± 0.4	---	9.9 ± 2.0	6.6 ± 2.9	0.006 ± 0.001	0.1 ± 0.0	1.6 ± 0.4	19.8 ± 3.6	19.8 ± 2.7
n-butane	0.3 ± 0.1	---	5.9 ± 1.2	0.02 ± 0.01	0.5 ± 0.1	0.3 ± 0.1	1.4 ± 0.4	8.5 ± 1.3	8.3 ± 1.2
i-butane	0.3 ± 0.1	---	2.2 ± 0.5	0.13 ± 0.06	0.08 ± 0.02	0.04 ± 0.01	1.8 ± 0.5	4.6 ± 0.6	5.1 ± 0.7
n-pentane	0.07 ± 0.02	---	2.2 ± 0.5	---	2.6 ± 0.4	1.0 ± 0.1	0.3 ± 0.1	6.6 ± 0.6	6.5 ± 0.9
i-pentane	0.11 ± 0.03	---	2.4 ± 0.5	0.003 ± 0.001	7.6 ± 1.0	3.9 ± 0.5	0.03 ± 0.01	14.1 ± 1.2	14.1 ± 1.8

[a] includes measurement, ODR fit, and inventory uncertainty

[b] *Wennberg et al.* [2012] estimate emissions to the SoCAB of 440 ± 100 Gg CH_4/yr and 12.9 ± 0.9 Gg ethane/yr

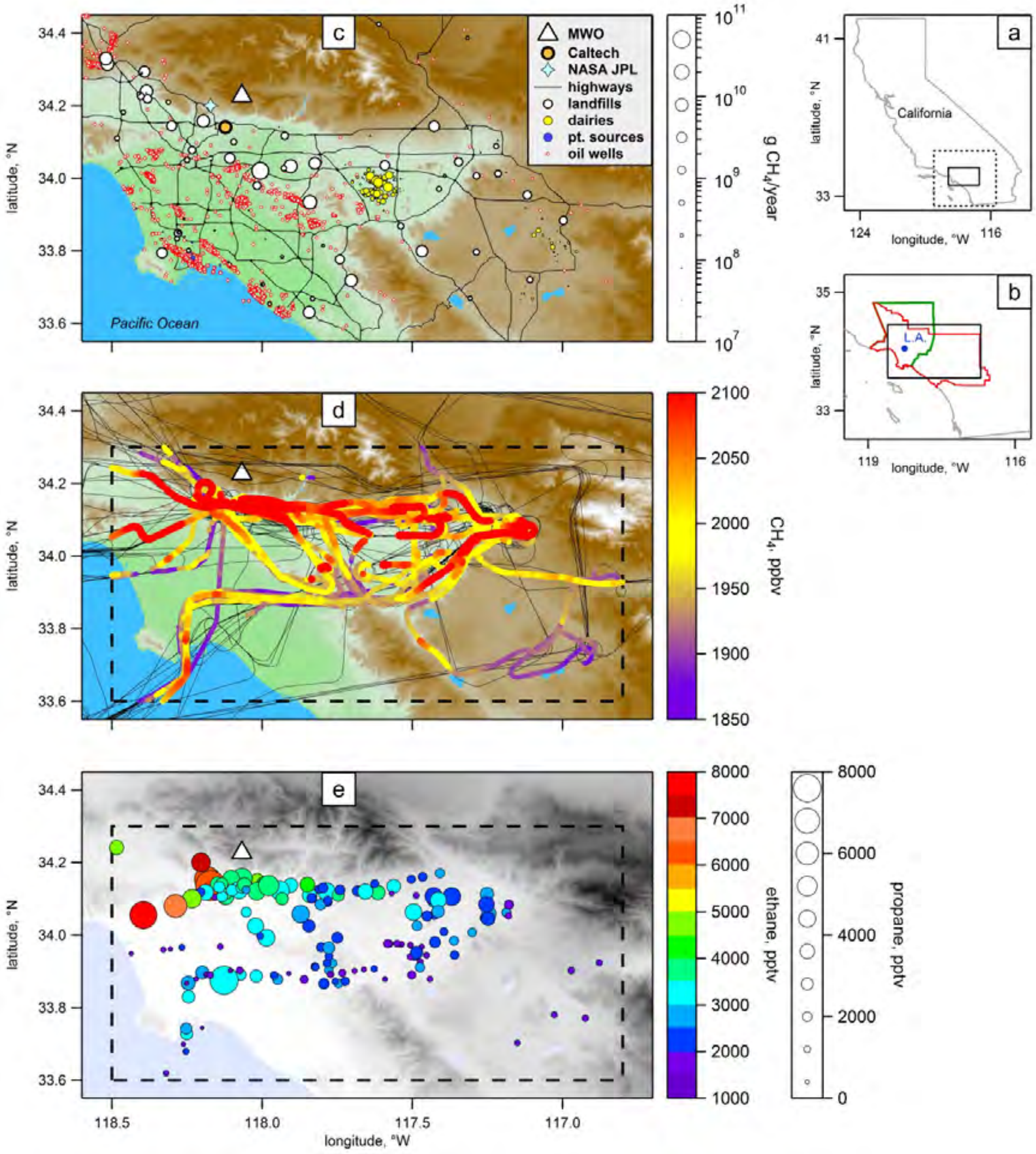

Figure 1. **a)** Map of California. The dashed box shows the inset for panel (b), the solid box shows the extent of the map boundaries for panels (c) – (e). **b)** Map of southern California showing the location of downtown L.A. (blue dot), the Los Angeles County boundary (green), the South Coast Air Basin boundary (red), and the extent of the map boundaries for panels (c) – (e) (black box). **c)** Map of the L.A. region showing known sources of CH$_4$ in the L.A. basin. The white triangle shows the location of the Mt. Wilson Observatory, where ground-based measurements were made by *Hsu et al.* [2010] and in this study. The light blue star shows the location of the Jet Propulsion Laboratory, where *Wunch et al.* [2009] made their measurements. The CalNex Pasadena ground site was located on the California Institute of Technology (Caltech) campus, located at the orange filled circle. Landfills (white circles) and CH$_4$ point sources (filled blue circles; negligibly small) are sized by emissions in the 2008 CARB greenhouse gas inventory. Dairies (filled yellow circles) are sized by the estimated emissions from the number of cows from *Salas et al.* [2008] multiplied by the 2009 CARB GHG inventory annual CH$_4$ emission per cow from enteric fermentation. **d)** Same map of the Los Angeles region as in (c), with flight tracks from 16 daytime flights of the NOAA P-3 (thin black lines). CH$_4$ measurements from the daytime boundary layer are color-coded atop these tracks according to the legend to the right. **e)** Locations of whole air samples in the L.A. basin, colored by ethane mixing ratio and sized by propane mixing ratio as indicated in the legends to the right.

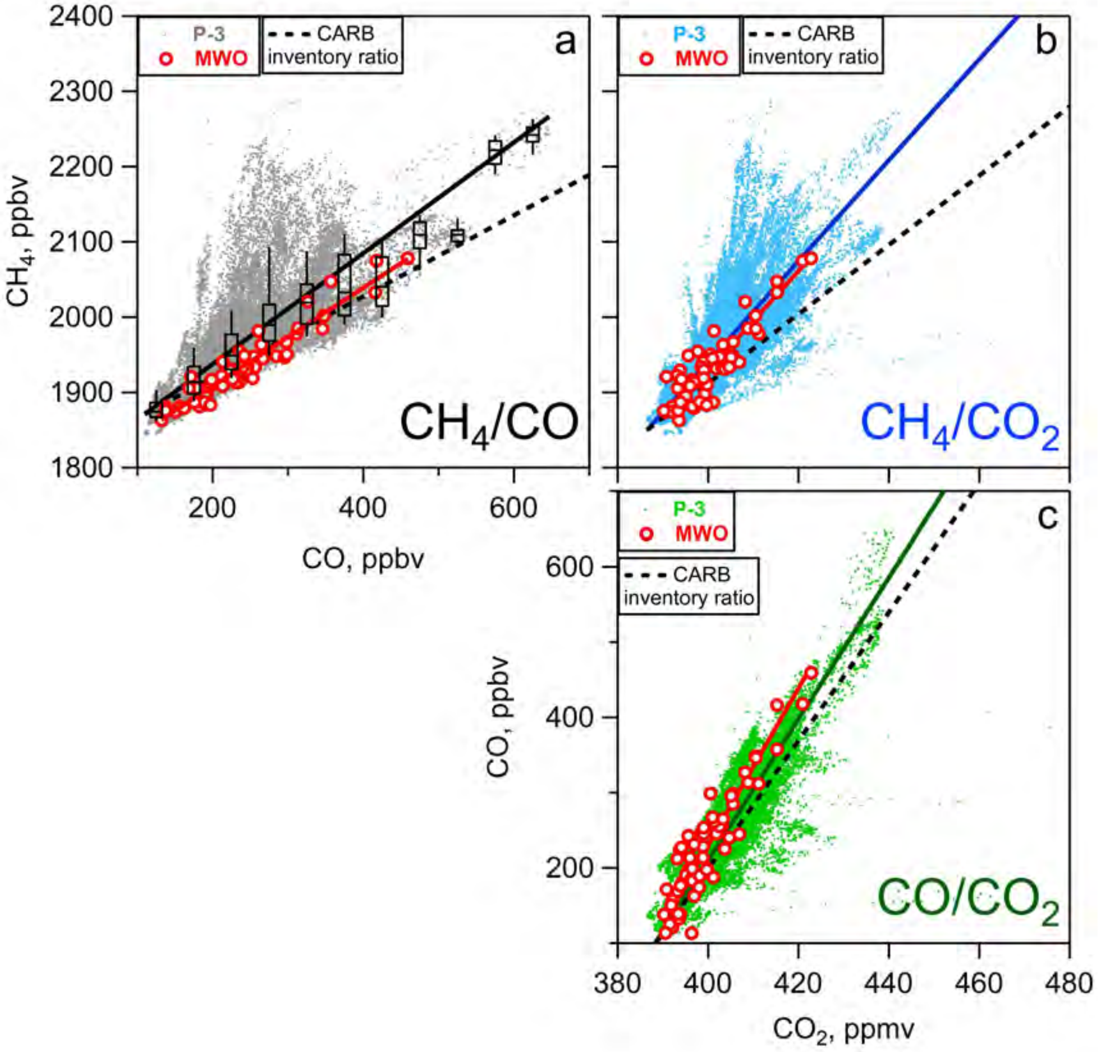

Figure 2. Scatter plots of CH_4, CO_2, and CO from all 1-second data points along flight track highlighted in Figure 1. Dots are from the NOAA P-3, while red circles are from NOAA GMD flask samples taken at the Mt. Wilson Observatory during CalNex. Weighted ODRs (solid lines) result in slopes of **(a)** 0.74 ± 0.04 and 0.68 ± 0.04 ppb CH_4/ppb CO; **(b)** 6.70 ± 0.01 and 6.60 ± 0.04 ppb CH_4/ppm CO_2; and **(c)** 9.4 ± 0.5 and 10.4 ± 0.5 ppb CO/ppm CO_2 from the NOAA P-3 and Mt. Wilson Observatory, respectively. The black dotted lines represent molar ratios of the CARB inventories listed in Table 2: CH_4:CO = 0.54, CH_4:CO_2 = 4.64×10^{-3}, and CO:CO_2 = 8.5×10^{-3}, where the background values used are the same as those determined from the fitted slopes. Also plotted in Figure 2a are boxes (25th–75th percentiles), whiskers (10th–90th percentiles), and the median (horizontal line) for distributions of CH_4 data calculated for 50 ppbv-wide bins from the NOAA P-3 CO data.

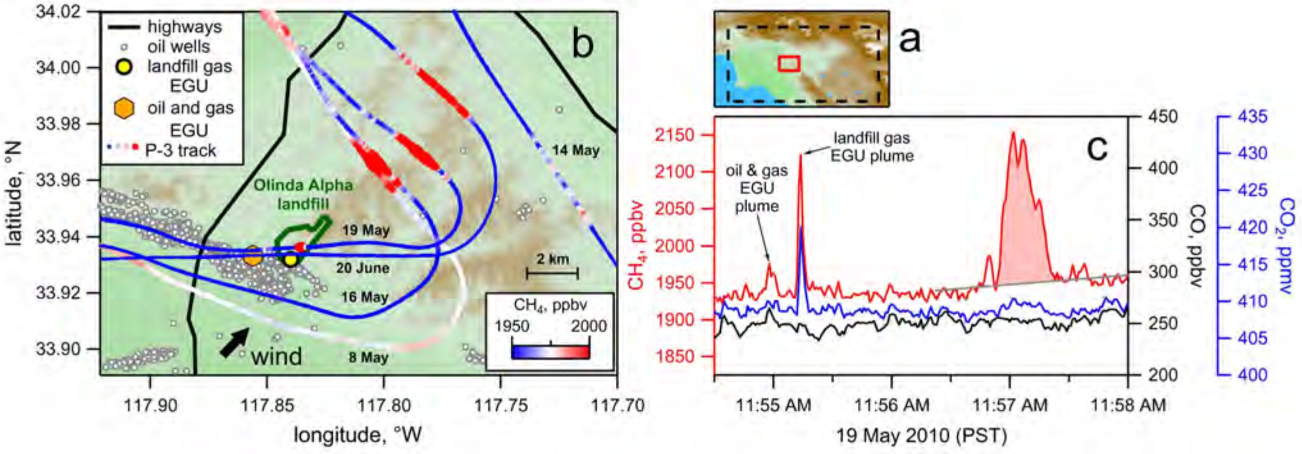

Figure 3. **a)** The map from Figure 1c–e shows the inset for part (b) in red. **b)** Five downwind transects, sized and colored by CH_4 mixing ratio, showing enhancements in CH_4 downwind of the Olinda Alpha landfill (green outline). Winds were from the southwest, except on 14 May, when they were from the west-southwest. **c)** Example of integration of the CH_4 plume from the 19 May flight. The filled pink area is integrated above the surrounding background (gray line). The upwind transect on this day passed downwind of two power plant (EGU) plumes.

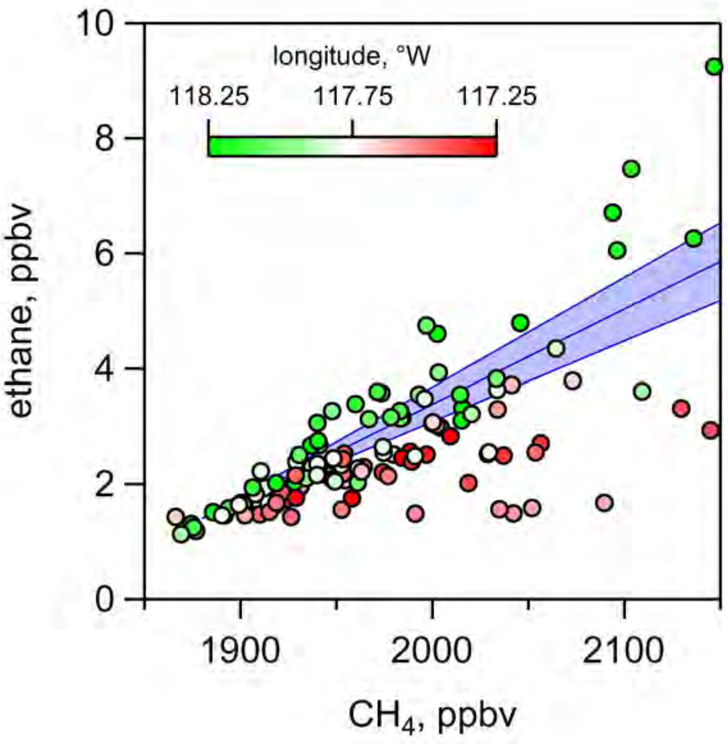

Figure 4. Scatter plot of ethane vs. CH_4 from the NOAA P-3 data in the L.A. basin. Data points are colored by longitude to show the different distributions of ethane to CH_4 in the eastern (red) and western (green) parts of the basin. The blue line represents the slope of 1.65 ± 0.25 % used by *Wennberg et al.* [2012] to represent the estimated ethane/CH_4 ratio of pipeline-quality dry natural gas from the Southern California Gas Company's pipelines.

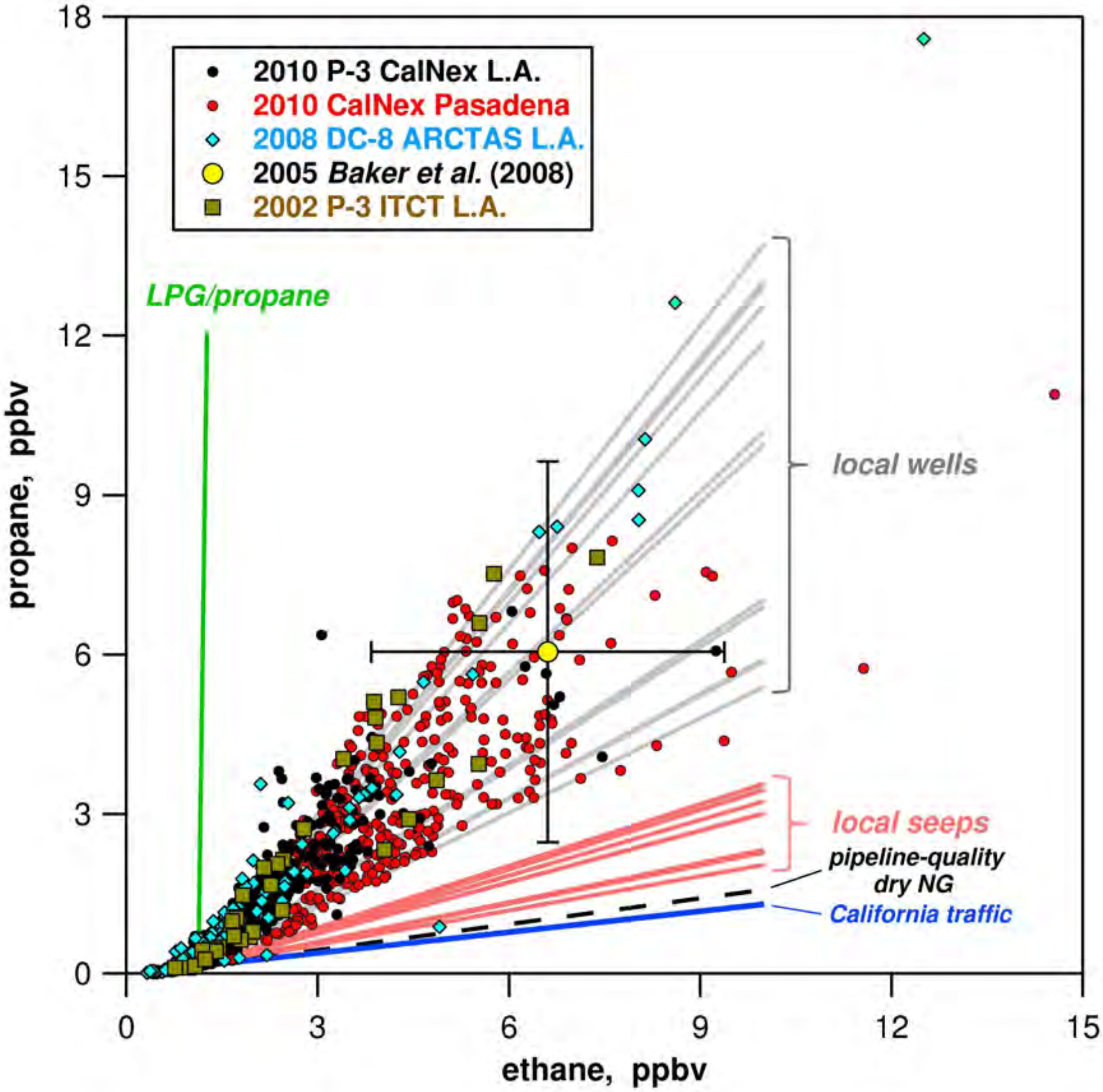

Figure 5. Correlation plot of propane vs. ethane from four Los Angeles datasets. Also plotted are composition ratios of local wells (gray lines) and local seeps (salmon lines) reported by *Jeffrey et al.* [1991], the composition ratio of pipeline-quality dry natural gas (black dashed line), the propane/ethane emission ratio from a San Francisco Bay-area tunnel study reported by *Kirchstetter et al.* [1996], and the average composition ratio of liquefied petroleum gas (LPG), or propane (green line).

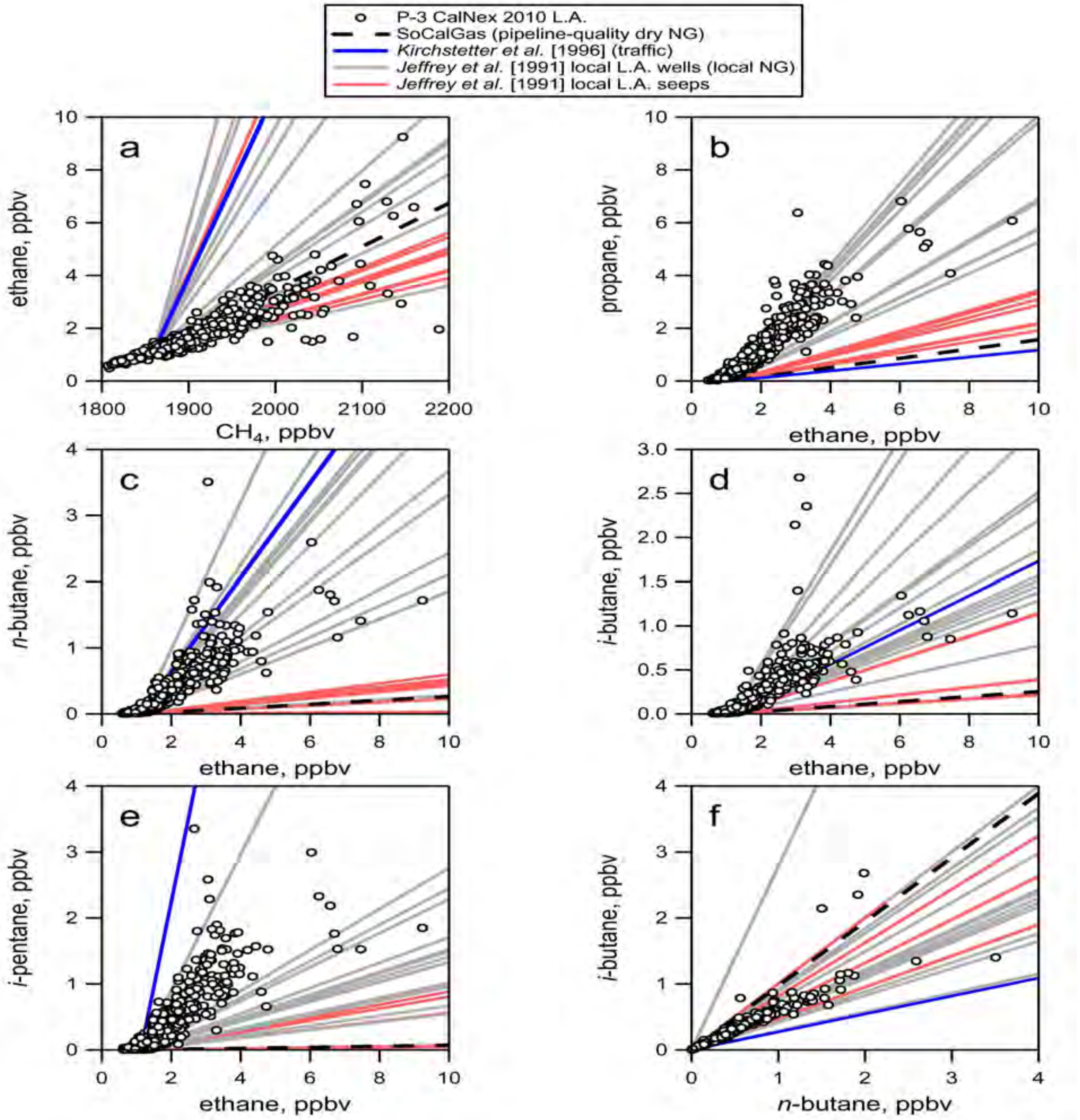

Figure 6. Plots of CH$_4$ and C$_2$–C$_5$ alkanes from the NOAA P-3 CalNex data set, selected for the SoCAB (black circles). Nighttime and high-altitude data are included. Also included for reference are the emission ratios of mobile sources from *Kirchstetter et al.* [1996] (blue line), composition ratios measured by *Jeffrey et al.* [1991] for local natural gas (gray lines) and local geologic seeps (salmon lines), and composition ratios from pipeline-quality dry natural gas (NG) delivered by SoCalGas (dashed black line). These ratios were plotted from daytime background levels.

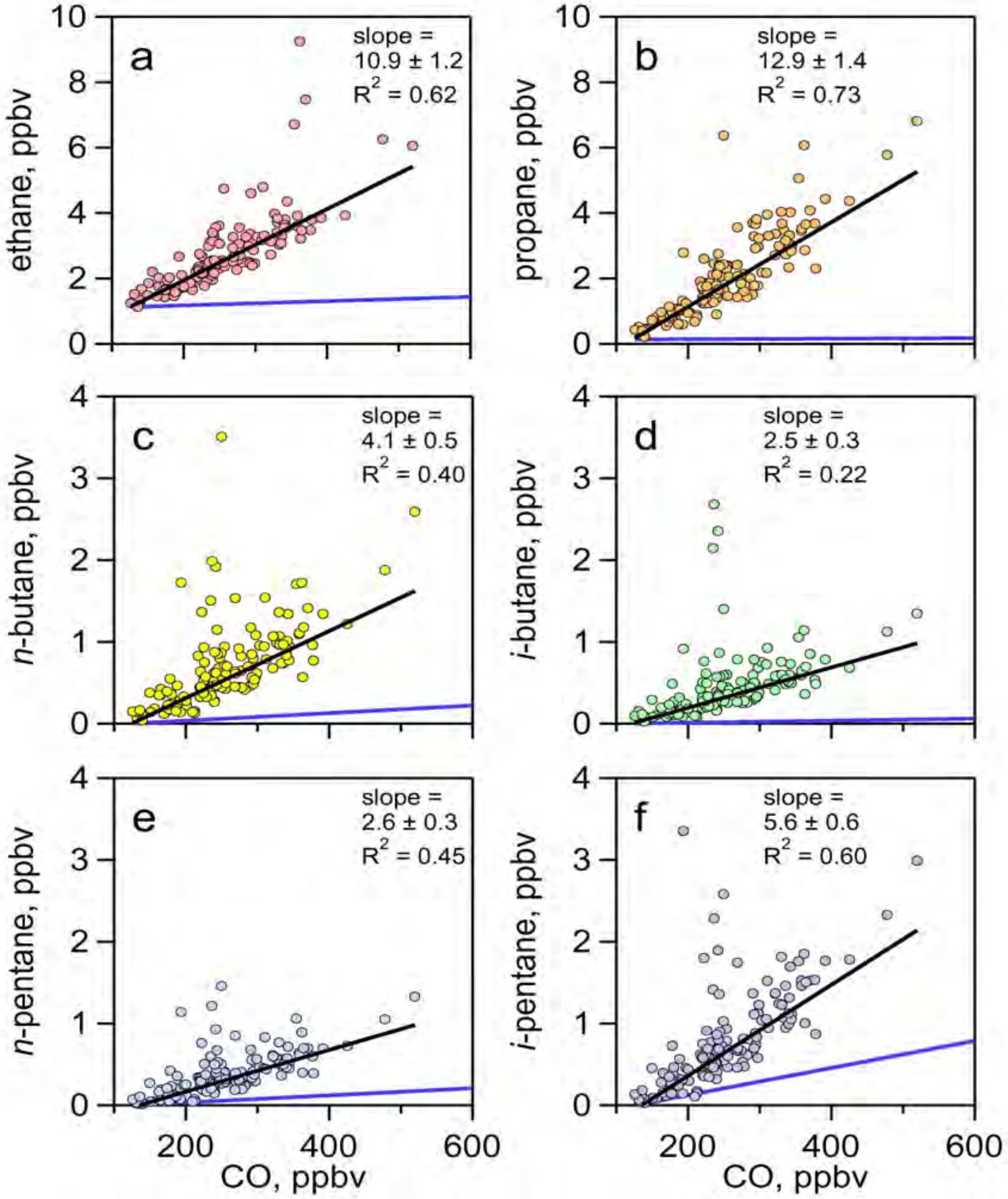

Figure 7. a–f) Daytime measurements of alkanes vs. CO from the NOAA P-3 in the L.A. basin during CalNex are plotted as filled circles. For comparison, the alkane/CO emission ratios from a San Francisco Bay-area tunnel study [*Kirchstetter et al.*, 1996] are plotted as a solid blue line, which extends to the edge of the right axis. The slope from a weighted ODR (given as ppt alkane/ppb CO), total slope uncertainty, and R^2 are given in each panel.

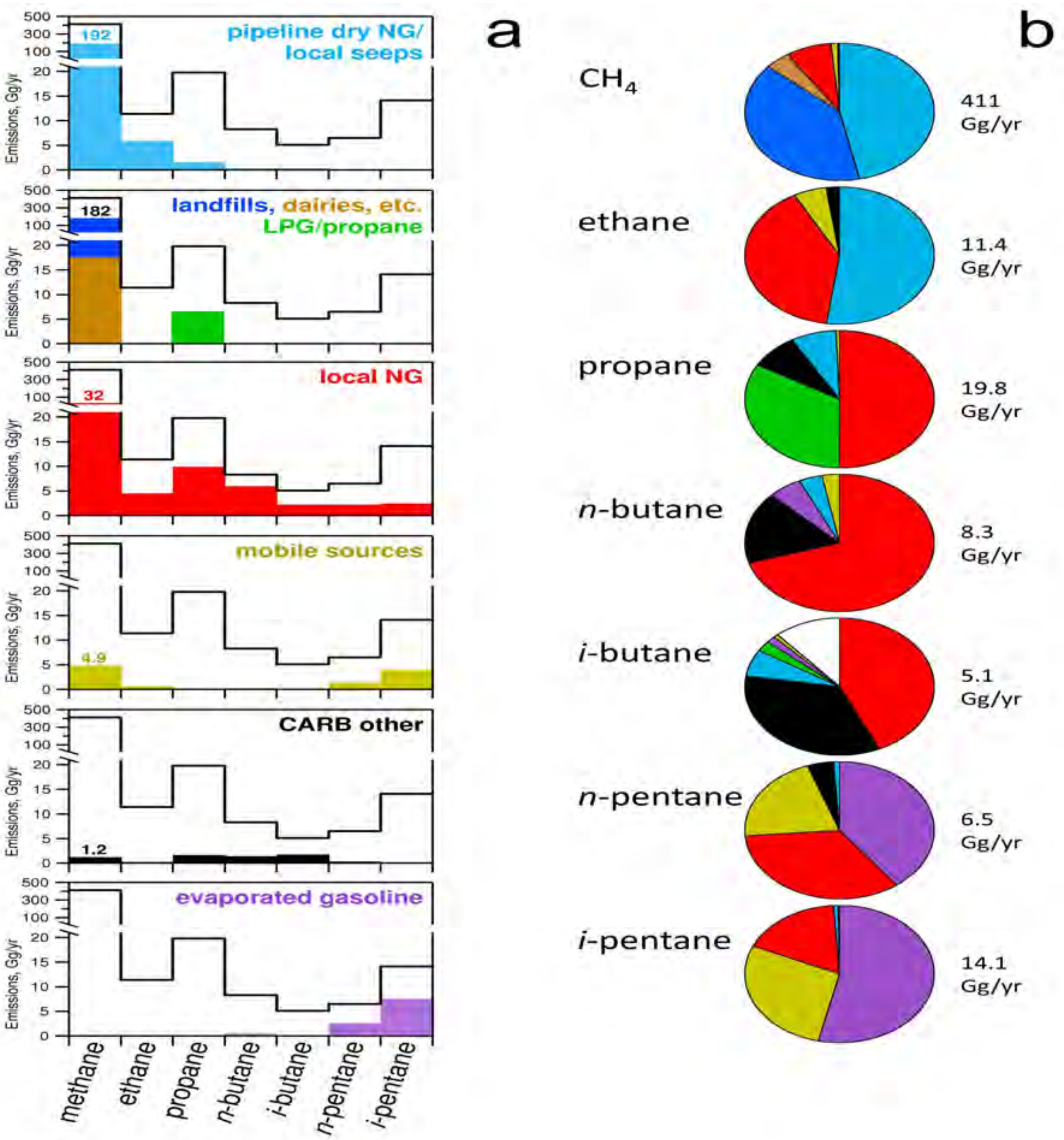

Figure 8. **a)** Results from a linear least squares solution to a combination of six sources and seven trace gas species in the SoCAB. The thick black line represents the estimated total annual emission to the SoCAB for seven hydrocarbons (CH_4 and C_2–C_5). The colored bars represent the fraction of the total contributed by each of the six source sectors used in the linear analysis. CH_4 emissions are written above the bar. **b)** Pie charts for the same data in (a) showing the relative contributions from each source for each of seven alkanes, colored as in part (a). The white region in the *i*-butane pie chart represents the 11% shortfall between our source attribution and our estimated emission to the SoCAB, though it is within the uncertainties of these two values. The total emission of the alkane to the SoCAB is given to the right of each pie chart.

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Rapid communication

Mapping urban pipeline leaks: Methane leaks across Boston

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ABSTRACT

Natural gas is the largest source of anthropogenic emissions of methane (CH_4) in the United States. To assess pipeline emissions across a major city, we mapped CH_4 leaks across all 785 road miles in the city of Boston using a cavity-ring-down mobile CH_4 analyzer. We identified 3356 CH_4 leaks with concentrations exceeding up to 15 times the global background level. Separately, we measured $\delta^{13}CH_4$ isotopic signatures from a subset of these leaks. The $\delta^{13}CH_4$ signatures (mean $= -42.8‰ \pm 1.3‰$ s.e.; $n = 32$) strongly indicate a fossil fuel source rather than a biogenic source for most of the leaks; natural gas sampled across the city had average $\delta^{13}CH_4$ values of $-36.8‰$ ($\pm 0.7‰$ s.e., $n = 10$), whereas CH_4 collected from landfill sites, wetlands, and sewer systems had $\delta^{13}CH_4$ signatures $\sim 20‰$ lighter ($\mu = -57.8‰$, $\pm 1.6‰$ s.e., $n = 8$). Repairing leaky natural gas distribution systems will reduce greenhouse gas emissions, increase consumer health and safety, and save money.

1. Introduction

Methane (CH_4) is a greenhouse gas more potent molecule for molecule than carbon dioxide (Shindell et al., 2012). In the United States, leaks of CH_4 from natural gas extraction and pipeline transmission are the largest human-derived source of emissions (EPA, 2012). However, CH_4 is not just a potent greenhouse gas; it also influences air quality and consumer health. CH_4 reacts with NO_x to catalyze ozone formation in urban areas (West et al., 2006). Incidents involving transmission and distribution pipelines for natural gas in the U. S. cause an average of 17 fatalities, 68 injuries, and $133 M in property damage each year (PHMSA, 2012). A natural gas pipeline explosion in San Bruno, CA, for instance, killed eight people and destroyed 38 homes in 2010. Detecting and reducing pipeline leaks of CH_4 and other hydrocarbons in natural gas are critical for reducing greenhouse gas emissions, improving air quality and consumer safety, and saving consumers money (West et al., 2006; Han and Weng, 2011; Shindell et al., 2012; Alvarez et al., 2012).

To assess CH_4 emissions in a major urban metropolis, we mapped CH_4 emissions over the entire 785 centerline miles of Boston's streets. To evaluate the likely source of the street-level CH_4 emissions, we also measured the $\delta^{13}C{-}CH_4$ carbon isotope composition, which can differentiate between biogenic (e.g., landfill, wetland, sewer) and thermogenic (e.g., natural gas) sources (Schoell, 1980).

2. Materials and methods

We conducted 31 mobile surveys during the period 18 August, 2011–1 October, 2011, covering all 785 road miles within Boston's city limits. We measured CH_4 concentration ([CH_4], ppm) using a mobile Picarro G2301 Cavity Ring-Down Spectrometer equipped with an A0491 Mobile Plume Mapping Kit (Picarro, Inc, Santa Clara, CA). This instrument was factory-calibrated on 15 August 2011, immediately prior to use in this study, and follow-up tests of the analyzer were made during 11–21 August, 2012, comparing analyzer output to a National Oceanic and Atmospheric Administration (NOAA) primary standard tank. In both pre- and post-checks, the analyzer output was found to be within 2.7 parts per billion of known [CH_4] in standard tanks, three orders of magnitude below typical atmospheric concentrations. Spectrometer and mobile GPS data were recorded every 1.1 s. To correct for a short time lag between instantaneous GPS location and a delay in [CH_4] measurement due to inlet tube length ($\sim$3 m), we used an auxiliary pump to increase tubing flow throughput to within 5 cm of the analyzer inlet; we also adjusted the time stamp on the [CH_4] readings based on a 1-s delay observed between analyzer response to a standard CH_4 source that we injected into the instrument while driving, and the apparent GPS location. We also checked the GPS-based locations of leaks with dozens of street-level sampling to confirm specific leak locations and the estimated sampling delay. Air was sampled through a 3.0 um Zefluor filter and Teflon tubing placed $\sim$30 cm above road surfaces.

For our mobile survey data, we defined a "leak" as a unique, spatially contiguous group of [CH_4] observations, all values of which exceed a concentration threshold of 2.50 ppm. This was used as a threshold because it corresponded to the 90th

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percentile of the distribution of data from all road miles driven, and, relative to global background, is ~37% above 2011 mean mixing ratios observed at Mauna Loa (NOAA, 2012).

Independently of mobile street sampling of CH_4, we measured $\delta^{13}CH_4$ from a subset of the leaks with a Picarro G2112i Cavity Ring-Down Spectrometer (Crosson, 2008). This instrument is calibrated monthly using isotopic standards from Isometric Instruments (Victoria, BC, Canada). The instrument was checked at least once daily to ensure analyzer output was within 1‰ of a tank of CH_4 with $\delta^{13}CH_4$ measured by a private lab (Isotech Labs, IL). Samples were collected in 1-L Tedlar sampling bags with valve and septa fittings, manufactured by Environmental Supply Company (Durham, NC). A Gas Sentry CGO-321 handheld gas detector (Bascom-Turner, MA) was used to identify the area of highest ambient $[CH_4]$ at each site sampled for $\delta^{13}CH_4$. Sampling bags were pre-evacuated and filled at the area of highest ambient concentration at the sampling site using a hand pump. $\delta^{13}CH_4$ was analyzed using a Picarro G2112i with a sample hold time typically of a few days and always less than two weeks.

At a subset of sampling sites ($n = 12$), we collected duplicate samples in glass vials to assess potential leaking or fractionation by the Tedlar sampling bags. We also sent duplicate samples from a different subset of sampling sites ($n = 5$) to a private lab (Isotech Labs, IL) for independent $\delta^{13}CH_4$ analysis. These analyses suggest no significant fractionation or bias either from the sampling bags or the Picarro G2112i analyzer. Most samples were analyzed at less than the maximum hold time of two weeks, at which bag diffusion could account for a 1.2‰ drift in our measurements of $\delta^{13}CH_4$.

We compared $\delta^{13}CH_4$ of these locations with samples taken from area landfills, wetlands, and the Deer Island Water Treatment Facility. Sampling equipment and procedures, as well as laboratory analyses, for landfill and wetland sites were similar to those for $\delta^{13}CH_4$ sampling locations described above. Samples were collected from three capped, inactive landfills (there are currently no active landfills in the Boston area). At one former landfill site, samples were collected at approximately three-month intervals between September, 2011 and April, 2012. The $\delta^{13}CH_4$ signature of the landfill was consistent over this period (±3.4‰ s.e.). At all wetland sampling sites, a plastic chamber (10 cm × 25 cm × 5 cm) connected to a sampling tube was placed over the surface of exposed moist sediment or shallow (>5 cm) water. Sediment below the chamber was disturbed gently before drawing air samples from the headspace within the chamber. The sample from the Deer Island Treatment Facility was drawn from the headspace of a sample bottle of anaerobic sludge, collected onsite by Deer Island staff for daily monitoring of the facility's anaerobic sludge digesters.

3. Results and discussion

We identified 3356 CH_4 leaks (Figs. 1 and 2) exceeding 2.50 parts per million. Surface concentrations corresponding to these leaks ranged up to 28.6 ppm, 14-times above a surface background concentration of 2.07 ppm (the statistical mode of the entire concentration distribution). Across the city, 435 and 97 independent leaks exceeded 5 and 10 ppm, respectively.

Based on their $\delta^{13}CH_4$ signatures, the CH_4 leaks strongly resembled thermogenic rather than biogenic sources (Fig. 3). Samples of natural gas from the gateway pipelines to Boston and from other consumer outlets in the city were statistically indistinguishable, with an average $\delta^{13}CH_4$ signature of −36.8‰ (±0.7‰ s.e., $n = 10$; ‰ vs. Vienna Pee Dee Belemnite). In contrast, CH_4 collected from landfill sites, wetlands, and sewer systems reflected a greater fractionation from microbial activity and $\delta^{13}CH_4$ signatures ~20‰ lighter. Biogenic values ranged from −53.1‰ to −64.5‰ ($\mu = -57.8$‰, ±1.6‰ s.e., $n = 8$) for samples collected in four wetlands, three capped landfills, and the primary sewage facility for the city, Deer Island Sewage Treatment Plant, which had the heaviest sample observed for non-natural-gas sources (−53.1‰). Our results for biogenic CH_4 carbon isotope signatures are consistent with other studies of the $\delta^{13}CH_4$ signature of CH_4 from landfills (Bergamaschi et al., 1998; Borjesson et al., 2001) and wetlands (Hornibrook et al., 2000).

Peaks of $[CH_4]$ detected in the road surveys strongly reflected the signature of natural gas rather than biogenic sources (Table 1). The average $\delta^{13}CH_4$ value for peaks was −42.8‰ ± 1.3‰ ($n = 32$), reflecting a dominant signal from natural gas, likely altered in some cases by minor fractionation of natural gas traveling through soils and by mixing with background air ($\delta^{13}CH_4 = -47$‰; Dlugokencky et al., 2011). A minority of samples had $\delta^{13}CH_4$ more negative than that of background air, reflecting apparent influence of biogenic CH_4. Most samples emitted a distinct odor of the mercaptan additive associated with natural gas, including those with a larger apparent biogenic influence on $\delta^{13}CH_4$.

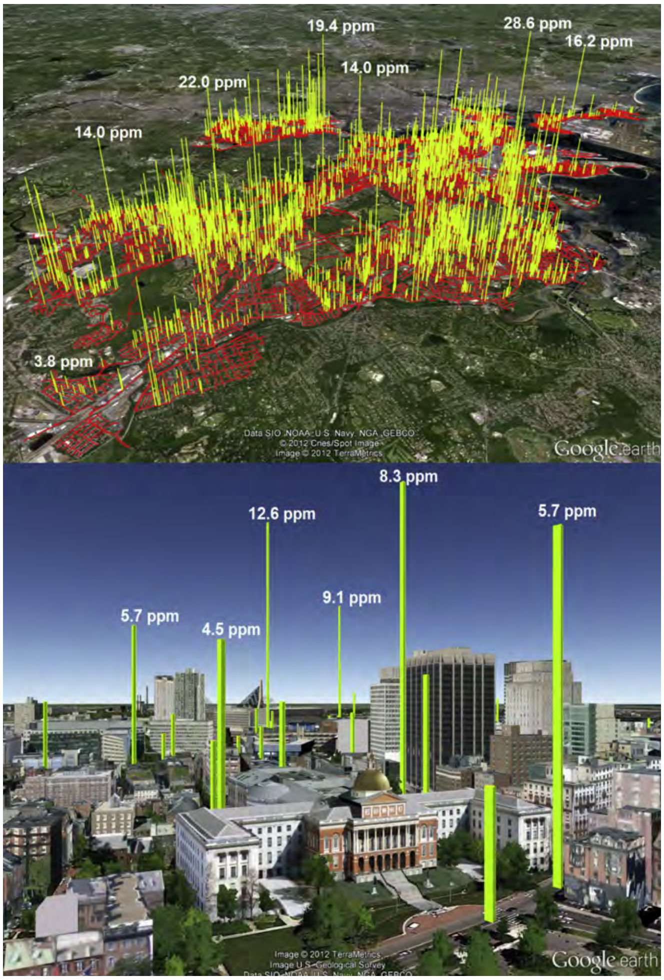

Fig. 1. Upper Panel: Methane leaks (3356 yellow spikes > 2.5 ppm) mapped on Boston's 785 road miles (red) surveyed in this study. Lower Panel: Leaks around Beacon Hill and the Massachusetts State House. Sample values of methane concentrations (ppm) are shown for each panel. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

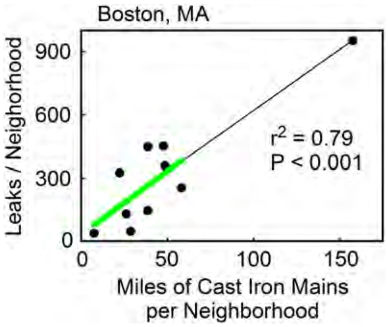

Fig. 2. Leak prevalence is associated with old cast iron pipes across ten Boston neighborhoods. (The combined line is the regression across all ten neighborhoods ($P < 0.001$); the green regression line [$r^2 = 0.34$; $P = 0.08$], which eliminates the influence of the leverage point [Dorchester neighborhood], has a slope and intercept indistinguishable ($P > 0.10$) from the combined regression.). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

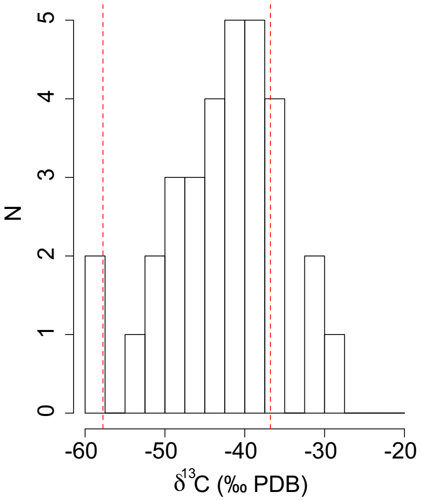

Fig. 3. $\delta^{13}CH_4$ of [CH_4] peaks detected in road surveys ($n = 32$). Red lines represent means of thermogenic (−36.8‰, ±0.7‰ s.e., $n = 10$) and biogenic (−57.8‰, ±1.6‰ s.e., $n = 8$) sources, respectively. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Leaks across Boston (Fig. 1), were associated primarily with cast iron mains that were sometimes over a century old (Fig. 2). Across ten Boston neighborhoods, leak frequency was linearly related to number of miles of cast iron mains ($r^2 = 0.79$, $P < 0.001$; Fig. 2), but only marginally to miles of non-cast-iron piping ($r^2 = 0.27$; $P = 0.12$, data not shown). Leak counts did not differ statistically by neighborhood or by socio-economic indicators for the neighborhoods obtained from the 2010 US Census ($P > 0.1$ for number of housing units and ethnicity) or the 2000 US Census ($P > 0.1$ for median income and poverty rate).

Table 1
Locations and isotopic values from discrete street leak samples.

Latitude	Longitude	$\delta^{13}CH_4$ (‰ PDB)
42.3654	−71.0612	−53.959
42.3439	−71.2628	−47.898
42.3493	−71.2265	−57.590
42.3583	−71.1749	−40.818
42.3411	−71.2440	−37.323
42.3543	−71.2441	−38.241
42.3559	−71.1898	−39.412
42.3513	−71.2092	−41.978
42.3515	−71.2081	−39.531
42.3614	−71.2314	−41.796
42.3426	−71.1012	−44.100
42.3443	−71.0949	−41.848
42.3328	−71.0761	−37.516
42.3360	−71.0738	−46.414
42.3441	−71.0673	−45.490
42.3303	−71.0569	−37.476
42.3409	−71.0542	−40.029
42.3524	−71.0445	−43.127
42.3799	−71.0272	−48.182
42.3722	−71.0361	−57.693
42.3785	−71.0681	−48.429
42.3730	−71.0632	−37.471
42.3593	−71.0629	−42.689
42.3584	−71.0644	−52.033
42.3546	−71.1271	−47.241
42.2943	−71.1891	−52.028
42.2793	−71.1514	−37.648
42.2887	−71.1428	−32.467
42.3285	−71.0792	−28.251
42.3215	−71.0692	−36.214
42.3269	−71.0796	−30.662
42.3553	−71.0573	−43.836
	Mean	**−42.793**
	Standard error	**1.259**

Reducing CH_4 leaks will promote safety and help save money. Although our study was not intended to assess explosion risks, we observed six locations where gas concentrations in manholes exceeded an explosion threshold of 4% [CH_4] at 20 °C (concentrations measured using a Gas Sentry CGO-321 handheld gas detector; Bascom-Turner, MA). Moreover, because CH_4, ethane (C_2H_6), and propane (C_3H_8) interact with NO_x to catalyze ozone formation, reducing these hydrocarbon concentrations should help reduce urban ozone concentrations and respiratory and cardiopulmonary disease (West et al., 2006; Shindell et al., 2012). CH_4 is also a potent greenhouse gas, with an estimated 20-year global warming potential 72 times greater than CO_2 (Alvarez et al., 2012; Townsend-Small et al., 2012). Replacing failing natural gas mains will reduce greenhouse gas emissions, thereby providing an additional benefit to the fewer mercury, SO_2 and particulate emissions that natural-gas burning emits compared to coal (Shindell et al., 2012). Finally, leaks contribute to $3.1 B of lost and unaccounted natural gas annually in the United States (EIA, 2012; 2005–2010 average).

Our ongoing and future research evaluates how surface [CH_4] values correspond to individual, and city-wide, urban leak rates and greenhouse-gas emissions. Two approaches to this question are useful: "bottom-up" chamber measurements taken on representative samples of individual leaks, and "top-down" atmospheric mass-balance estimates from rooftops of the collective urban leak rate that exploit the known isotopic signature of natural gas versus that of biogenic sources and other fossil fuel sources. The instrumentation used in this study is well-suited for both approaches.

We propose that a coordinated campaign to map urban pipeline leaks around the world would benefit diverse stakeholders, including companies, municipalities, and consumers. Repairing the leaks will bring economic, environmental, and health benefits to all.

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Greater focus needed on methane leakage from natural gas infrastructure

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Natural gas is seen by many as the future of American energy: a fuel that can provide energy independence and reduce greenhouse gas emissions in the process. However, there has also been confusion about the climate implications of increased use of natural gas for electric power and transportation. We propose and illustrate the use of technology warming potentials as a robust and transparent way to compare the cumulative radiative forcing created by alternative technologies fueled by natural gas and oil or coal by using the best available estimates of greenhouse gas emissions from each fuel cycle (i.e., production, transportation and use). We find that a shift to compressed natural gas vehicles from gasoline or diesel vehicles leads to greater radiative forcing of the climate for 80 or 280 yr, respectively, before beginning to produce benefits. Compressed natural gas vehicles could produce climate benefits on all time frames if the well-to-wheels CH_4 leakage were capped at a level 45–70% below current estimates. By contrast, using natural gas instead of coal for electric power plants can reduce radiative forcing immediately, and reducing CH_4 losses from the production and transportation of natural gas would produce even greater benefits. There is a need for the natural gas industry and science community to help obtain better emissions data and for increased efforts to reduce methane leakage in order to minimize the climate footprint of natural gas.

With growing pressure to produce more domestic energy and to reduce greenhouse gas (GHG) emissions, natural gas is increasingly seen as the fossil fuel of choice for the United States as it transitions to renewable sources. Recent reports in the scientific literature and popular press have produced confusion about the climate implications of natural gas (1–5). On the one hand, a shift to natural gas is promoted as climate mitigation because it has lower carbon per unit energy than coal or oil (6). On the other hand, methane (CH_4), the prime constituent of natural gas, is itself a more potent GHG than carbon dioxide (CO_2); CH_4 leakage from the production, transportation and use of natural gas can offset benefits from fuel-switching.

The climatic effect of replacing other fossil fuels with natural gas varies widely by sector (e.g., electricity generation or transportation) and by the fuel being replaced (e.g., coal, gasoline, or diesel fuel), distinctions that have been largely lacking in the policy debate. Estimates of the net climate implications of fuel-switching strategies should be based on complete fuel cycles (e.g., "well-to-wheels") and account for changes in emissions of relevant radiative forcing agents. Unfortunately, such analyses are weakened by the paucity of empirical data addressing CH_4 emissions through the natural gas supply network, hereafter referred to as CH_4 leakage.* The U.S. Environmental Protection Agency (EPA) recently doubled its previous estimate of CH_4 leakage from natural gas systems (6).

In this paper, we illustrate the importance of accounting for fuel-cycle CH_4 leakage when considering the climate impacts of fuel-technology combinations. Using EPA's estimated CH_4 emissions from the natural gas supply, we evaluated the radiative forcing implications of three U.S.-specific fuel-switching scenarios: from gasoline, diesel fuel, and coal to natural gas.

A shift to natural gas and away from other fossil fuels is increasingly plausible because advances in horizontal drilling and hydraulic fracturing technologies have greatly expanded the country's extractable natural gas resources particularly by accessing gas stored in shale deep underground (7). Contrary to previous estimates of CH_4 losses from the "upstream" portions of the natural gas fuel cycle (8, 9), a recent paper by Howarth et al. calculated upstream leakage rates for shale gas to be so large as to imply higher lifecycle GHG emissions from natural gas than from coal (1). (*SI Text*, discusses differences between our paper and Howarth et al.) Howarth et al. estimated CH_4 emissions as a percentage of CH_4 produced over the lifecycle of a well to be 3.6–7.9% for shale gas and 1.7–6.0% for conventional gas. The EPA's latest estimate of the amount of CH_4 released because of leaks and venting in the natural gas network between production wells and the local distribution network is about 570 billion cubic feet for 2009, which corresponds to 2.4% of gross U.S. natural gas production (1.9–3.1% at a 95% confidence level) (6).† EPA's reported uncertainty appears small considering that its current value is double the prior estimate, which was itself twice as high as the previously accepted amount (9).

Comparing the climate implications of CH_4 and CO_2 emissions is complicated because of the much shorter atmospheric lifetime of CH_4 relative to CO_2. On a molar basis, CH_4 produces 37 times more radiative forcing than CO_2.‡ However, because CH_4 is oxidized to CO_2 with an effective lifetime of 12 yr, the integrated, or cumulative, radiative forcings from equi-molar releases of CO_2 and CH_4 eventually converge toward the same value. Determining whether a unit emission of CH_4 is worse for the climate than a unit of CO_2 depends on the time frame considered. Because accelerated rates of warming mean ecosystems and humans have less time to adapt, increased CH_4 emissions due to substitution of natural gas for coal and oil may produce undesirable climate outcomes in the near-term.

The concept of global warming potential (GWP) is commonly used to compare the radiative forcing of different gases relative

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*Challenges also exist in the quantification of CH_4 emissions from the extraction of coal. We use the term "leakage" for simplicity and define it broadly to include all CH_4 emissions in the natural gas supply, both fugitive leaks as well as vented emissions.

†This represents an uncertainty range between −19% and +30% of natural gas system emissions. For CH_4 from petroleum systems (35% of which we assign to the natural gas supply) the uncertainty is −24% to +149%; however, this is only a minor effect because the portion of natural gas supply that comes from oil wells is less than 20%.

‡One-hundred-two times on a mass basis. This value accounts for methane's direct radiative forcing and a 40% enhancement because of the indirect forcing by ozone and stratospheric water vapor (10).

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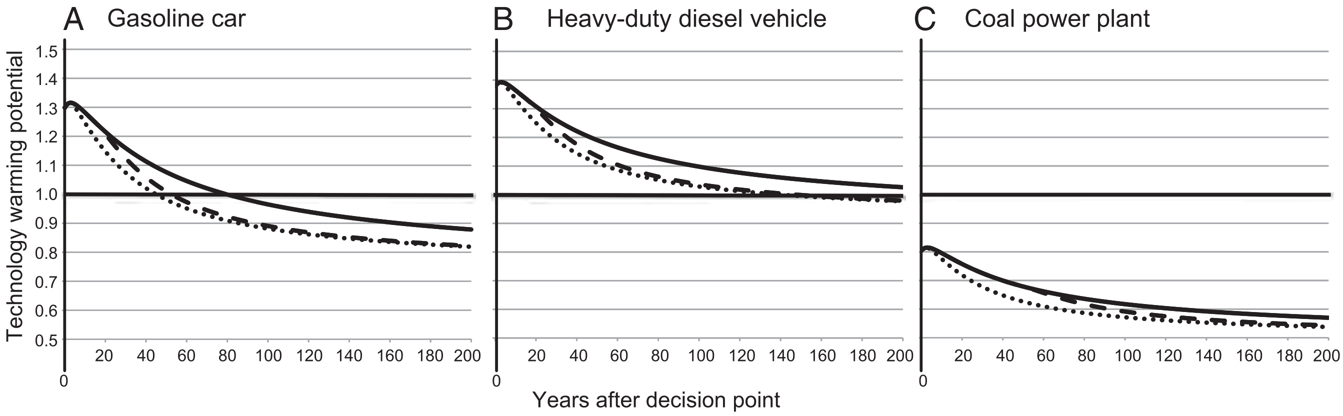

Fig. 1. Technology warming potential (TWP) for three sets of natural gas fuel-switching scenarios. (*A*) CNG light-duty cars vs. gasoline cars; (*B*) CNG heavy-duty vehicles vs. diesel vehicles; and (*C*) combined-cycle natural gas plants vs. supercritical coal plants using low-CH_4 coal. The three curves within each frame simulate real-world choices, including a single emissions pulse (dotted lines); emissions for the full service life of a vehicle or power plant (15 and 50 years, respectively, dashed lines); and emissions from a converted fleet continuing indefinitely (solid lines). For the pulse and service life analyses, our scenarios assume that the natural gas choice reverts back to the incumbent choice before the switch took place; for the fleet conversion analysis we assume that a natural gas vehicle or power plant is replaced by an identical unit at the end of its service life.

to CO_2 and represents the ratio of the cumulative radiative forcing *t* years after emission of a GHG to the cumulative radiative forcing from emission of an equivalent quantity of CO_2 (10). The Intergovernmental Panel on Climate Change (IPCC) typically uses 100 yr for the calculation of GWP. Howarth et al. (1) emphasized the 20-year GWP, which accentuates the large forcing in early years from CH_4 emissions, whereas Venkatesh et al. (2) adopted a 100-yr GWP and Burnham et al. (4) utilized both 20- and 100-yr GWPs.

GWPs were established to allow for comparisons among GHGs at one point in time after emission but only add confusion when evaluating environmental benefits or policy tradeoffs over time. Policy tradeoffs like the ones examined here often involve two or more GHGs with distinct atmospheric lifetimes. A second limitation of GWP-based comparisons is that they only consider the radiative forcing of single emission pulses, which do not capture the climatic consequences of real-world investment and policy decisions that are better simulated as emission streams.

To avoid confusion and enable straightforward comparisons of fuel-technology options, we suggest that plotting as a function of time the relative radiative forcing of the options being considered would be more useful for policy deliberations than GWPs. These technology warming potentials (TWP) require exactly the same inputs and radiative forcing formulas used for GWP but reveal time-dependent tradeoffs inherent in a choice between alternative technologies. We illustrate the value of our approach by applying it to emissions of CO_2 and CH_4 from vehicles fueled with CNG compared with gasoline or diesel vehicles and from power plants fueled with natural gas instead of coal.

Wigley also analyzed changes in the relative benefits over time of switching from coal to natural gas, but that was done in the context of additional complexities including specific assumptions about the global pace of technological substitution, emissions of sulfur dioxide and black carbon, and a specific model of global warming due to radiative forcing (5). We compare our results with Wigley's in the next section.

Results and Discussion

We focus on the TWPs of real-world choices faced by individuals, corporations, and policymakers about fuel-switching in the transport and power sectors. Each of the three curves within the panels of Fig. 1 represents a distinct choice and its associated emission duration: for example, whether to rent a CNG or a gasoline car for a day (Pulse TWP); whether to purchase and operate a CNG or gasoline car for a 15-yr service life (Service-Life TWP); and whether a nation should adopt a policy to convert the gasoline fleet of cars to CNG (Fleet Conversion TWP). In each of these cases, a TWP greater than 1 means that the cumulative radiative forcing from choosing natural gas today is higher than a current fuel option after *t* yr. Our results for pulse TWP at 20 and 100 yr are identical to fuel-cycle analyses using 20-year or 100-year GWPs for CH_4.

Given EPA's current estimates of CH_4 leakage from natural gas production and delivery infrastructure, in addition to a modest CH_4 contribution from the vehicle itself (for which few empirical data are available), CNG-fueled vehicles are not a viable mitigation strategy for climate change.[§] Converting a fleet of gasoline cars to CNG increases radiative forcing for 80 yr before any net climate benefits are achieved; the comparable cross-over point for heavy-duty diesel vehicles is nearly 300 yr.

Stated differently, converting a fleet of cars from gasoline to CNG would result in numerous decades of more rapid climate change because of greater radiative forcing in the early years after the conversion. This is eventually offset by a modest benefit. After 150 yr, a CNG fleet would have produced about 10% less cumulative radiative forcing than a gasoline fleet—a benefit equivalent to a fuel economy improvement of 3 mpg in a 30 mpg fleet. CNG vehicles fare even less favorably in comparison to heavy-duty diesel vehicles.

In contrast to the transportation cases, a fleet of new, combined-cycle natural gas power plants reduces radiative forcing on all time frames, relative to new coal plants burning low-CH_4 coal—assuming current estimates of leakage rates (Fig. 1*C*). The conclusions differ primarily because of coal's higher carbon content relative to petroleum fuels; however, fuel-cycle CH_4 leakage can also affect results. (As discussed elsewhere in this paper, our analysis considered only the emissions of CH_4 and CO_2. In *SI Text*, we examine the effect of different CH_4 leak rates in the coal and natural gas fuel cycles for the electric power scenario.)

To provide guidance to industry and policymakers, we also determined the maximum well-to-wheels or well-to-burner-tip leakage rate needed to ensure net climate benefits on all time frames after fuel-switching to natural gas (see Fig. 2). For example, if the well-to-wheels leakage was reduced to an effective leak rate of 1.6% of natural gas produced (approximately 45% below our estimate of current leakage of 3.0%), CNG cars would result

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[§] The CH_4 from operation of a CNG automobile was estimated to be 20 times the value for gasoline vehicles (11), which is approximately 20% of the well-to-pump CH_4 leakage on a kg/mmBtu basis. This assumption deserves much further scrutiny.

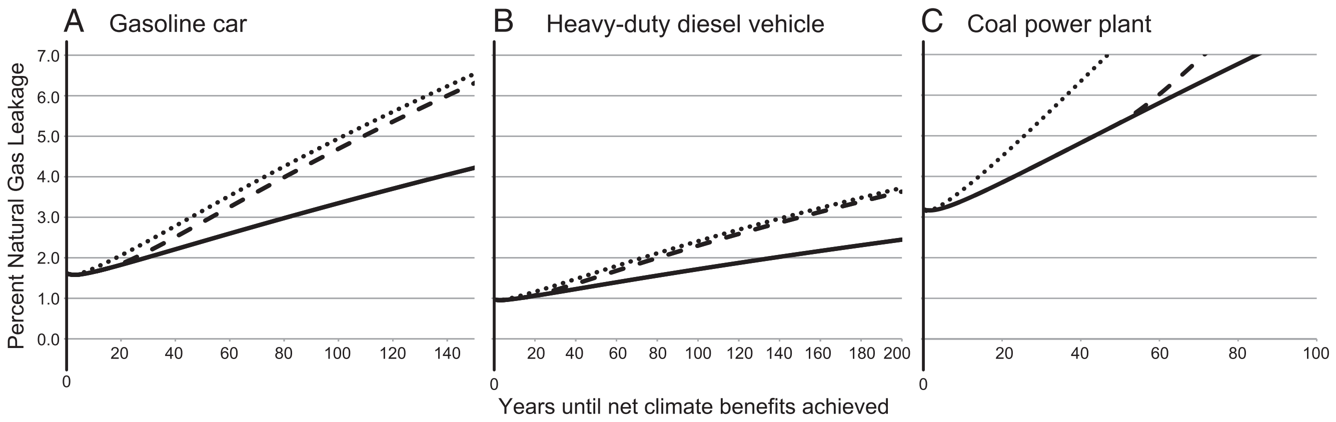

Fig. 2. Maximum "well-to-wheels" natural gas leak rate as a function of the number of years needed to achieve net climate benefits after choosing a CNG option in lieu of (*A*) gasoline cars; (*B*) heavy-duty diesel vehicles; and (*C*) coal power plants. For *A* and *B*, the maximum leakage is the sum of losses from the well through the distribution system plus losses from the CNG vehicle itself (well-to-wheels); for *C*, the maximum leakage is from the well through the transmission system where most power plants receive their fuel. When leak rates are less than the y-intercept, a fuel switch scenario would result in net climate benefits beginning immediately. The three curves within each frame follow the conventions outlined in Fig. 1 and represent: single emissions pulses (dotted lines); the service life of a vehicle or a power plant, 15 or 50 years, respectively (dashed lines); and a permanent fleet conversion (solid lines).

in climate benefits immediately and improve over time.[¶] For CNG to immediately reduce climate impacts from heavy-duty vehicles, well-to-wheels leakage must be reduced below 1%. Fig. 2*C* shows that new natural gas power plants produce net climate benefits relative to efficient, new coal plants using low-gassy coal on all time frames as long as leakage in the natural gas system is less than 3.2% from well through delivery at a power plant. Fig. 2 also shows, for a range of leakage rates, the number of years needed to reach the "cross-over point" when net climate benefits begin to occur after a fuel-technology choice is made.

We emphasize that our calculations assume an average leakage rate for the entire U.S. natural gas supply (as well for coal mining). Much work needs to be done to determine actual emissions with certainty and to accurately characterize the site-to-site variability in emissions. However, given limited current evidence, it is likely that leakage at individual natural gas well sites is high enough, when combined with leakage from downstream operations, to make the total leakage exceed the 3.2% threshold beyond which gas becomes worse for the climate than coal for at least some period of time.[||] Our analysis of reported routine emissions for over 250 well sites with no compressor engines in Barnett Shale gas well sites in Fort Worth, Texas, in 2010 revealed a highly skewed distribution of emissions, with 10% of well sites accounting for nearly 70% of emissions (see *SI Text*).[**] Natural gas leak rates calculated based on operator-reported, daily gas production data at these well sites ranged from 0% to 5%, with six sites out of 203 showing leak rates of 2.6% or greater due to routine emissions alone.[††]

Our analysis of coal-to-natural gas fuel-switching does not consider potential changes in sulfate aerosols and black carbon, short-lived climate forcers previously shown to affect the climate implications of such fuel-switching scenarios (5, 13). Recently, Wigley concluded that coal-to-gas switching on a global scale would result in increased warming on a global scale in the short term, based on examining a set of scenarios with a climate model that included both the increased warming produced by CH_4 losses from the natural gas fuel cycle and the additional cooling that occurs due to SO_2 emissions and the sulfate aerosols they form as a result of burning coal (5). The applicability of Wigley's global conclusion to the United States or any other individual country is limited due to the reliance on global emissions scenarios. Analyses such as Wigley's, which model the climate impacts of all climate forcing emissions, are useful to evaluate specific fuel-switching scenarios; however, their ultimate relevance to policymakers and fleet owners will be determined by the fidelity with which they reflect actual emissions from all phases of each fuel cycle at the relevant geographic scale (e.g., national, continental, or global). The SO_2 emissions that Wigley assumed are much higher than those of the current fleet of coal electrical generation plants in the United States, where SO_2 emissions declined by more than 50% between 2000 and 2010.[‡‡] Moreover, due to state and federal pollution abatement requirements, U.S. SO_2 emissions are projected to continue declining, to roughly 30% of 2000 levels by 2014 (see *SI Text*). This means that by 2014 the projected sulfur emissions from the U.S. coal electrical generation plant fleet, 3 TgS/GtC, will approach the emission factor that Wigley assumed the global fleet would reach in 2060 (2 TgS/GtC), when he projected the climate benefits of fuel-switching might begin, and significantly lower than Wigley's estimated 2010 value of 12 TgS/GtC. Accounting for the lower SO_2 from U.S. coal plants in an integrated way will result in greater net climate impacts of using coal than reported by Wigley and in turn the net benefits of fuel-switching will occur much sooner than he projected.

Increasingly, this will also be the case globally. The production of sulfur aerosols as a result of coal combustion causes such negative impacts on human and ecosystem health that it is prudent to assume that policies will continue to be rapidly implemented in many, if not most, countries to reduce such emissions at a much faster pace than assumed by Wigley. Indeed, it has been reported that China has already installed SO_2 scrubbers on power plants accounting for over 70% of the nation's installed coal power capacity (14), such that SO_2 emissions from power plants in 2010 were 58% below 2004 levels (15). The SO_2 emissions factor from

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[¶] Our estimate that current well-to-wheels leakage is 3.0% of gas produced assumes that 2.4% of gas produced is lost between the well and the local distribution system (based on EPA's 2011 GHG emission inventory) and that 0.6% is due to emissions during refueling and from the vehicle itself. For further discussion of the climatic implication of natural gas vehicles see (12).

[||] EPA's GHG inventory suggests leakage from natural gas processing and transmission is 0.6% of gas produced, meaning production leakage must be greater than 2.6% for the total fuel cycle leakage of a power plant receiving fuel from a transmission pipeline to exceed 3.2%.

[**] Sites with compressor engines were excluded due to the contractor's assumption that all engines in the City were uncontrolled, which leads to erroneous emission estimates.

[††] Routine emissions do not include such occasional events as well completions and blowdowns. Only 203 of the 254 sites had data for gas production. An Excel spreadsheet containing the Fort Worth data and our calculations is provided in Dataset S1.

[‡‡] Emissions query performed on December 5, 2011, using the Data and Maps feature of the U.S. Environmental Protection Agency's Clean Air Markets Web page (http://camddataandmaps.epa.gov/gdm/).

Chinese coal plants in 2010 has been estimated to be 204 g/GJ, comparable to the 2010 value of 229 g/GJ (4.7 TgS/GtC) for U.S. coal plants (*SI Text*).

Little work appears to have been done to evaluate fuel-switching in on-road transportation with methods that consider the implications of all climate forcing emissions, including sulfur aerosols and black carbon, although the effect of short-lived climate forcers on individual transport sectors has been studied (16, 17). One study reports that the influence of negative radiative forcing due to emissions from on-road transport is much lower than for the power generation sector in both the United States and globally (18). This implies that our approach, which considers CO_2 and CH_4 emissions alone, provides a reasonable first-order estimate of changes in radiative forcing from fuel-switching scenarios for the on-road transport sector.

Conclusions

The TWP Approach Proposed Here Offers Policymakers Greater Insights than Conventional GWP Analyses. GWPs are a valuable tool to compare the radiative forcing of different gases but are not sufficient when thinking about fuel-switching scenarios. TWPs provide a transparent, policy-relevant analytical approach to examine the time-dependent climate influence of different fuel-technology choices.

Improved Science and Data Are Needed. Despite recent changes to EPA's methodology for estimating CH_4 leakage from natural gas systems, the actual magnitude remains uncertain and estimates could change as methods are refined. Ensuring a high degree of confidence in the climate benefits of natural gas fuel-switching pathways will require better data than are available today. EPA's rule requiring natural gas industry disclosure of GHG emissions should begin to produce data in 2012, though it is unlikely that most uncertainties will be resolved and possible systematic biases eliminated. Specific challenges include confirming the primary sources of emissions and determining drivers of variance in leakage rates. Greater direct involvement of the scientific community could help improve estimates of CH_4 leakage and identify approaches that enable independent validation of industry-reported emissions.

Reductions in CH_4 Leakage Are Needed to Maximize the Climate Benefits of Natural Gas. While CH_4 leakage from natural gas infrastructure and use remains uncertain, it appears that current leakage rates are higher than previously thought. Because CH_4 initially has a much higher effect on radiative forcing than CO_2, maintaining low rates of CH_4 leakage is critical to maximizing the climate benefits of natural gas fuel-technology pathways. Significant progress appears possible given the economic benefits of capturing and selling lost natural gas and the availability of proven technologies. (EPA's Natural Gas STAR program shows many examples: www.epa.gov/gasstar/tools/recommended.html.)

Table 2. Radiative efficiency (RE) values used in this paper

	Direct RE ($W\ m^{-2}\ ppb^{-1}$)	Relative direct + indirect RE (per ppb or molar basis)	Relative direct + indirect RE (per kg basis)*
CO_2	1.4×10^{-5}	1	1
CH_4	3.7×10^{-4}	37	102

*Obtained by multiplying the molar radiative efficiency by the ratio of molecular weights of CH_4 and CO_2.

Methods

Our approach of using TWPs to compare the cumulative radiative forcing of fuel-technology combinations is a straightforward extension of the calculation of GWP, which is given by Eq. **1** over a time horizon, TH, for a pulse emission of 1 kg of a generic GHG producing time-dependent radiative forcing given by $RF_{GHG}(t)$:

$$\mathrm{GWP} = \frac{\int_0^{\mathrm{TH}} \mathrm{RF}_{\mathrm{GHG}}(t)dt}{\int_0^{\mathrm{TH}} \mathrm{RF}_{\mathrm{CO_2}}(t)dt}. \quad [1]$$

SI Text shows the analytical solution of Eq. **1** (i.e., GWP as a function of time horizon). Plotting the entire curve enables one to see the GWP values for all time horizons.

Our TWP approach extends the standard GWP calculation in two ways: by combining the effects of CH_4 and CO_2 emissions from technology-fuel combinations and by considering streams of emissions in addition to single pulses. Considering streams of emissions is more reflective of real-world scenarios that involve activities that occur over multiyear time frames.

Eq. **2** is our extension of the GWP formula Eq. **1** to calculate TWPs, with the following definitions. We label as Technology-1 the alternative that combusts natural gas and has CO_2 emissions E_{1,CO_2} and CH_4 emissions from the production, processing, storage, delivery, and use of the fuel: E_{1,CH_4}. If L_{REF} is the percent of gross natural gas produced that is currently emitted to the atmosphere over the relevant fuel cycle (e.g., electric power or transportation), then Technology-1's CH_4 emissions at leakage rate L would be: $(L/L_{REF})E_{1,CH_4}$. The calculations of TWP in this paper assume that the leakage rate L is at the national average value L_{REF} (and thus $L/L_{REF} = 1$). The scaling factor L/L_{REF} is included to allow calculations about changes in the national leakage rate or about individual wells and distribution networks that deviate from the national average. The values we used for L_{REF} are derived in *SI Text* using EPA's estimated emissions with one exception and are equal to 2.1% for a natural gas power plant and 3.0% for CNG vehicles. The exception to the last statement is that we estimated CH_4 from the operation of a CNG automobile to be 20 times that from a gasoline vehicle (11), which is approximately 20% of the well-to-pump CH_4 leakage on a kg/mmBtu basis. This assumption deserves much further scrutiny. Technology-2 combusts gasoline, diesel fuel, or coal and produces CO_2 emissions E_{2,CO_2} and methane emissions E_{2,CH_4}. Estimates of the Es for each of the technologies considered are reported in Table 1 and are explained in *SI Text*. The TWPs at each point in time can be obtained by substituting the total radiative forcing values, $TRF_{CH_4}(t)$ and $TRF_{CO_2}(t)$ for CH_4 and CO_2, respectively, and emission factors, $E_{n,GHG}$ from Table 1 into Eq. **2**:

Table 1. Emission factors used for TWP calculations in this paper

	Power Plants		Vehicles			
	Natural gas combined cycle* (kg/MW*h*)	Supercritical pulverized coal† (kg/MW*h*)	Light-duty CNG car (kg/mmBtuHHV)‡	Light-duty gasoline car (kg/mmBtuHHV)	Heavy-duty CNG truck (mg/ton-mile)	Heavy-duty diesel truck (mg/ton-mile)
Upstream CH_4	3.1	0.65	0.51	0.1	590	100
Upstream CO_2	36	7	9.4	15.9	10,000	15,000
In-Use CH_4	0	0	0.11	0.0056	15	0
In-Use CO_2	361	807	53.1	70.3	80,000	85,000
Fuel cycle CH_4	3.1	0.65	0.62	0.11	605	100
Fuel cycle CO_2	397	814	62.5	86.2	90,000	100,000

*Heat rate = 6,798 Btu/kWh.

†Heat rate = 8,687 Btu/kWh.

‡1 mmBtu = 10^6 Btu = 1.055 GJ.

Table 3. Total radiative forcing (TRF) functions for CH_4 and CO_2 used in calculation of TWP in Eq. 2 for three distinct emissions profiles

Case	$\mathrm{TRF}_{CH_4}(t)$	$\mathrm{TRF}_{CO_2}(t)$
Pulse TWP	$\mathrm{RE}\{\tau_M(1-e^{-t/\tau_M})\}$	$a_0 t + \sum_{i=1}^{3} a_i\tau_i(1-e^{-t/\tau_i})$
Service Life TWP for $t \le \mathrm{AMAX}$	$\mathrm{RE}\{\tau_M t-\tau_M^2(1-e^{-t/\tau_M})\}$	$a_0\frac{t^2}{2}+\sum_{i=1}^{3} a_i(\tau_i t-\tau_i^2(1-e^{-t/\tau_i}))$
Service Life TWP for $t > \mathrm{AMAX}$	$\mathrm{RE}\{\tau_M \mathrm{AMAX}-\tau_M^2 e^{-t/\tau_M}(e^{\mathrm{AMAX}/\tau_M}-1)\}$	$a_0[\mathrm{AMAX}\,t-\frac{\mathrm{AMAX}^2}{2}]+\sum_{i=1}^{3} a_i(\tau_i \mathrm{AMAX}-\tau_i^2 e^{-t/\tau_i}(e^{\mathrm{AMAX}/\tau_i}-1))$
Fleet Conversion TWP	$\mathrm{RE}\{\tau_M t-\tau_M^2(1-e^{-t/\tau_M})\}$	$a_0\frac{t^2}{2}+\sum_{i=1}^{3} a_i(\tau_i t-\tau_i^2(1-e^{-t/\tau_i}))$

RE in these formulas is the radiative efficiency of CH_4 relative to CO_2 and equals 102.

$$\mathrm{TWP}(t)=\frac{\frac{L}{L_{\mathrm{REF}}}E_{1,\mathrm{CH_4}}\mathrm{TRF}_{\mathrm{CH_4}}(t)+E_{1,\mathrm{CO_2}}\mathrm{TRF}_{\mathrm{CO_2}}(t)}{E_{2,\mathrm{CH_4}}\mathrm{TRF}_{\mathrm{CH_4}}(t)+E_{2,\mathrm{CO_2}}\mathrm{TRF}_{\mathrm{CO_2}}(t)}. \qquad [2]$$

The TRF values needed for Eq. **2** are derived as follows. Let $f(t, t_E)$ be the mass of a gas left in the atmosphere at time t if 1 kg of the gas was emitted at time t_E. The cumulative radiative forcing function, $\mathrm{CRF}(t)$ (in units of $\mathrm{J\,m^{-2}\,kg^{-1}}$), at a later time t, due to emission of 1 kg of the gas at time t_E, is then:

$$\mathrm{CRF}(t)\equiv\int_{t_E}^{t}\mathrm{RE}\,f(x,t_E)dx, \qquad [3]$$

where RE is the radiative efficiency of the gas. The integral in Eq. **3** sums radiative forcing for the $t-t_E$ years from the year in which the gas was emitted, $x=t_E$, to year $x=t$. For simplicity, we adopt units which make the RE of CO_2 equal to one, and so the RE of CH_4 is expressed as a multiple of the RE of CO_2. In these units, the RE of CH_4 is determined to be 102, using the values in Table 2 taken from the IPCC (10) and following the IPCC convention that methane's direct radiative efficiency be enhanced by 25% and 15% to account for indirect forcing due to ozone and stratospheric water, respectively.

Now suppose that instead of a single pulse, the gas is emitted continuously at a rate of 1 kg/yr from $t=0$ until some maximum time $t_{\max}$, as would occur, for example, if emissions were to continue over the service life of a vehicle, power plant, or fleet. For such cases we define the total radiative forcing (TRF) in year t to be:

$$\mathrm{TRF}(t)\equiv\int_{0}^{t_{\max}}\int_{t_E}^{t}\mathrm{RE}\,f(x,t_E)dx\,dt_E. \qquad [4]$$

In the special case of a single emission pulse, $\mathrm{TRF}(t)=\mathrm{CRF}(t)$. Our use of Eq. **4** assumes a constant, unit emission rate; a more general formulation could be employed to reflect potential technology improvements over time.

For CH_4, $f(t, t_E)$ is an exponential decay:

$$f(t,t_E)=e^{-\frac{t-t_E}{\tau_M}}, \qquad [5]$$

where τ_M is 12 yr. For CO_2, we follow the IPCC and use the Bern carbon cycle model (10):

$$f(t,t_E)=a_0+\sum_{i=1}^{3}a_i e^{-\frac{t-t_E}{\tau_i}} \qquad [6]$$

where $\tau_1=172.9$, $\tau_2=18.51$, $\tau_3=1.186$, $a_0=0.217$, $a_1=0.259$, $a_2=0.338$, and $a_3=0.186$. Our calculations do not consider the CO_2 produced from the oxidation of CH_4, an approximation which introduces a small underestimation of the radiative forcing from a fuel cycle's CH_4 leakage.

If calculating the TWP for a single pulse of emissions (pulse TWP), then $t_E=0$; $\mathrm{TRF}_{CH_4}(t)$ is given by Eq. **3** with $f(t, t_E)$ given by Eq. **5**; and $\mathrm{TRF}_{CO_2}(t)$ is given by Eq. **3** with $f(t, t_E)$ given by Eq. **6**. If calculating the TWP for a permanent fuel conversion of a fleet (fleet conversion TWP) then $\mathrm{TRF}_{CH_4}(t)$ is given by Eq. **4** with $t_{\max}=t$ and $f(t, t_E)$ given by Eq. **5**. Similarly, $\mathrm{TRF}_{CO_2}(t)$ is given by Eq. **4** with $t_{\max}=t$ and $f(t, t_E)$ given by Eq. **6**. If calculating the TWP for emissions over the service life of a vehicle or power plant (service life TWP) and $t\le\mathrm{AMAX}$, where AMAX is the average age at which the asset ceases to emit, then $TRF_{CH_4}(t)$ and $\mathrm{TRF}_{CO_2}(t)$ are the same as in the fleet conversion TWP calculations. However, if $t>\mathrm{AMAX}$, then $\mathrm{TRF}_{CH_4}(t)$ is given by Eq. **4** with $t_{\max}=\mathrm{AMAX}$ and $f(t, t_E)$ given by Eq. **5**. Similarly, $\mathrm{TRF}_{CO_2}(t)$ is given by Eq. **4** with $t_{\max}=\mathrm{AMAX}$ and $f(t, t_E)$ given by Eq. **6**. The solutions for all of these cases are in Table 3. We use $\mathrm{AMAX}=15$ yr for vehicles and $\mathrm{AMAX}=50$ yr for power plants.

By rearranging terms in Eq. **2** when $\mathrm{TWP}=1$ to bring L to the left hand side, we obtain an equation for the relationship between the cross-over time (t^*—the time at which the two technologies have equal cumulative radiative forcing) and the percent leakage that makes this happen (L^*):

$$L^*=L_{\mathrm{REF}}\left\{\frac{E_{2,\mathrm{CH_4}}}{E_{1,\mathrm{CH_4}}}+\frac{E_{2,\mathrm{CO_2}}-E_{1,\mathrm{CO_2}}}{E_{1,\mathrm{CH_4}}}\,\frac{\mathrm{TRF}_{\mathrm{CO_2}}(t^*)}{\mathrm{TRF}_{\mathrm{CH_4}}(t^*)}\right\}. \qquad [7]$$

Taking the limit of L^* as the cross-over time t^* goes to zero, we obtain an expression for the critical leakage rate L_0, which serves as an approximation of the leakage rate below which the natural gas-burning technology causes less radiative forcing on all time frames.

$$L_0=L_{\mathrm{REF}}\left\{\frac{E_{2,\mathrm{CH_4}}}{E_{1,\mathrm{CH_4}}}+\frac{E_{2,\mathrm{CO_2}}-E_{1,\mathrm{CO_2}}}{\mathrm{RE}\ \mathrm{E}_{1,\mathrm{CH_4}}}\right\} \qquad [8]$$

where $\mathrm{RE}=102$. Eq. **8** must be viewed as an approximation because L^* is a nonmonotonic function of t^* for small values of t^* (see Fig. 2, which plots L^* as a function of cross-over time t^*). The small decrease in L^* for small t^* is caused by the fact that 18.6% of the emitted CO_2 decays faster than CH_4 in the Bern carbon cycle model (time scales of 1.186 vs. 12 yr). The large increase in L^* for $t_* > 3$ years is caused by the rapid decay of CH_4 relative to the remaining 81.4% of the CO_2. The decay curves for CO_2 and CH_4 are shown in *SI Text*. Calculated values of L_o using Eq. **8** are within 2–3% of the absolute minima for L^*. Calculations of TWP and L^* using Eq. **2** and Eq. **8** were performed with an Excel spreadsheet and are available in Dataset S1.

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Simultaneously Mitigating Near-Term Climate Change and Improving Human Health and Food Security

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Tropospheric ozone and black carbon (BC) contribute to both degraded air quality and global warming. We considered ~400 emission control measures to reduce these pollutants by using current technology and experience. We identified 14 measures targeting methane and BC emissions that reduce projected global mean warming ~0.5°C by 2050. This strategy avoids 0.7 to 4.7 million annual premature deaths from outdoor air pollution and increases annual crop yields by 30 to 135 million metric tons due to ozone reductions in 2030 and beyond. Benefits of methane emissions reductions are valued at $700 to $5000 per metric ton, which is well above typical marginal abatement costs (less than $250). The selected controls target different sources and influence climate on shorter time scales than those of carbon dioxide–reduction measures. Implementing both substantially reduces the risks of crossing the 2°C threshold.

Tropospheric ozone and black carbon (BC) are the only two agents known to cause both warming and degraded air quality. Although all emissions of BC or ozone precursors [including methane (CH_4)] degrade air quality, and studies document the climate effects of total anthropogenic BC and tropospheric ozone (*1*–*4*), published literature is inadequate to address many policy-relevant climate questions regarding these pollutants because emissions of ozone precursors have multiple cooling and warming effects, whereas BC is emitted along with other particles that cause cooling, making the net effects of real-world emissions changes obscure. Such information is needed, however, because multiple stakeholders are interested in mitigating climate change via control of non–carbon dioxide (CO_2)–forcing agents such as BC, including the G8 nations (L'Aquila Summit, 2009) and the Arctic Council (Nuuk Declaration, 2011). Here, we show that implementing specific practical emissions reductions chosen to maximize climate benefits would have important "win-win" benefits for near-term climate, human health, agriculture, and the cryosphere, with magnitudes that vary strongly across regions. We also quantify the monetized benefits due to health, agriculture, and global mean climate change per metric ton of CH_4 and for the BC measures as a whole and compare these with implementation costs.

Our analysis proceeded in steps. Initially, ~400 existing pollution control measures were screened with the International Institute for Applied Systems Analysis Greenhouse Gas and Air Pollution Interactions and Synergies (IIASA GAINS) model (*5*, *6*). The model estimated potential worldwide emissions reductions of particulate and gaseous species on the basis of available real-world data on reduction efficiencies of these measures where they have been applied already and examined the impact of full implementation everywhere by 2030. Their potential climate impact was assessed by using published global warming potential (GWP) values for each pollutant affected. All emissions control measures are assumed to improve air quality. We then selected measures that both mitigate warming and improve air quality, ranked by climate impact. If enhanced air quality had been paramount, the selected measures would be quite different [for example, measures primarily reducing sulfur dioxide (SO_2) emissions improve air quality but may increase warming]. The screening revealed that the top 14 measures realized nearly 90% of the maximum reduction in net GWP (table S1 and fig. S2). Seven measures target CH_4 emissions, covering coal mining, oil and gas production, long-distance gas transmission, municipal waste and landfills, wastewater, livestock manure, and rice paddies. The others target emissions from incomplete combustion and include technical measures (set "Tech"), covering diesel vehicles, clean-burning biomass stoves, brick kilns, and coke ovens, as well as primarily regulatory measures (set "Reg"), including banning agricultural waste burning, eliminating high-emitting vehicles, and providing modern cooking and heating. We refer to these seven as "BC measures," although in practice, we consider all co-emitted species (*7*).

We then developed future emissions scenarios to investigate the effects of the emissions control measures in comparison with both a reference and a potential low-carbon future: (i) a reference scenario based on energy and fuel projections of the International Energy Agency (IEA) (*8*) regional and global livestock projections (*9*) and incorporating all presently agreed policies affecting emissions (*10*); (ii) a CH_4 measures scenario that follows the reference but also adds the CH_4 measures; (iii) CH_4+BC measures scenarios that follow the reference but add the CH_4 and one or both sets of BC measures; (iv) a CO_2 measures scenario under which CO_2 emissions follow the IEA's "450 CO_2-equivalent" scenario (*8*) as implemented in the GAINS model (affecting CO_2 and co-emissions of SO_2 but not other long-lived gases); and (v) a combined CO_2 plus CH_4 and BC measures scenario. Measures are phased in linearly from 2010 through 2030, after which only trends in CO_2 emissions are included, with other emissions kept constant.

Emissions from these scenarios were then used with the ECHAM5-HAMMOZ (*11*) and GISS-PUCCINI (*12*) three-dimensional composition-climate models to calculate the impacts on atmospheric concentrations and radiative forcing (*7*). Changes in surface $PM_{2.5}$ (particles of less than 2.5 micrometers) and tropospheric ozone were used with published concentration-response relationships (*13*–*15*) to calculate health and agricultural impacts. CH_4 forcing was calculated from the modeled CH_4 concentrations. Direct ozone and aerosol radiative forcings were produced by using the fraction of total anthropogenic direct radiative forcing removed by the emission control measures, as calculated in the two models, multiplied by the best estimate and uncertainty range for direct forcing, which was determined from a literature assessment. Albedo forcing was similarly estimated on the basis of the fractional decrease of BC deposition to snow and ice surfaces. Indirect and semidirect forcings were estimated by simply assuming that these had the same fractional changes as the direct forcings (*16*). Initially, analytic equations representing rapid and slow components of the climate system

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(*17*) were used to estimate global and regional (*18*) mean temperature response to the forcings.

This analytic analysis shows that the measures substantially reduce the global mean temperature increase over the next few decades by reducing tropospheric ozone, CH_4, and BC (Fig. 1). The short atmospheric lifetime of these species allows a rapid climate response to emissions reductions. In contrast, CO_2 has a very long atmospheric lifetime (hence, growing CO_2 emissions will affect climate for centuries), so that the CO_2 emissions reductions analyzed here hardly affect temperatures before 2040. The combination of CH_4 and BC measures along with substantial CO_2 emissions reductions [a 450 parts per million (ppm) scenario] has a high probability of limiting global mean warming to <2°C during the next 60 years, something that neither set of emissions reductions achieves on its own [which is consistent with (*19*)].

Work to this stage was largely in support of the Integrated Assessment of Black Carbon and Tropospheric Ozone (*20*). Here, we present detailed climate modeling and extend impact analyses to the national level, where regulations are generally applied and which provides detailed spatial information that facilitates regional impact analyses. We also provide cost/benefit analyses.

Climate modeling. We performed climate simulations driven by the 2030 CH_4 plus BC measures, by greenhouse gas changes only, and by reference emissions using the GISS-E2-S model; the same GISS atmosphere and composition models were coupled to a mixed-layer ocean (allowing ocean temperatures, but not circulation, to adjust to forcing). Direct, semidirect (aerosol effects on clouds via atmospheric heating), indirect (aerosol effects on clouds via microphysics), and snow/ice albedo (by BC deposition) forcings were calculated internally (*7*). We analyzed the equilibrium response 30 to 50 years after imposition of the measures, which is comparable with the latter decades in the analytic analysis.

The global mean response to the CH_4 plus BC measures was –0.54 ± 0.05°C in the climate model. The analytic equations yielded –0.52°C (–0.21 to –0.80°C) for 2070, which is consistent with these results. Climate model uncertainty only includes internal variations, whereas the analytic estimate includes uncertainties in forcing and climate sensitivity (but has no internal variability).

We also examined individual forcing components. Direct global mean aerosol forcings in the ECHAM and GISS models are almost identical (Table 1), despite large uncertainties generally present in aerosol forcing and the two aerosol models being fundamentally different [for example, internal versus external mixtures (*7*)]. CH_4 and ozone responses to CH_4 emissions changes are also quite similar. Ozone responses to changes in CO, volatile organic compounds, and NO_x associated with the BC measures are quite different, however. This is consistent with the nonlinear response of ozone to these precursors (*21*).

The combined indirect and semidirect radiative forcing by all aerosols in the GISS model is negative for the BC Tech and Reg measures. Although sulfate increases slightly—largely because of increases in the oxidant H_2O_2—in all emissions control scenarios, the BC measures primarily decrease BC and organic carbon (OC). The negative forcing suggests that a decreased positive semidirect effect may outweigh decreased negative indirect effects of BC and OC in this model [studies differ on the magnitude of these effects (*22*–*24*)]. Indirect effects are much larger than net direct effects for the Tech measures.

Global mean BC albedo forcing in the model is very small (Table 1), but we assume its

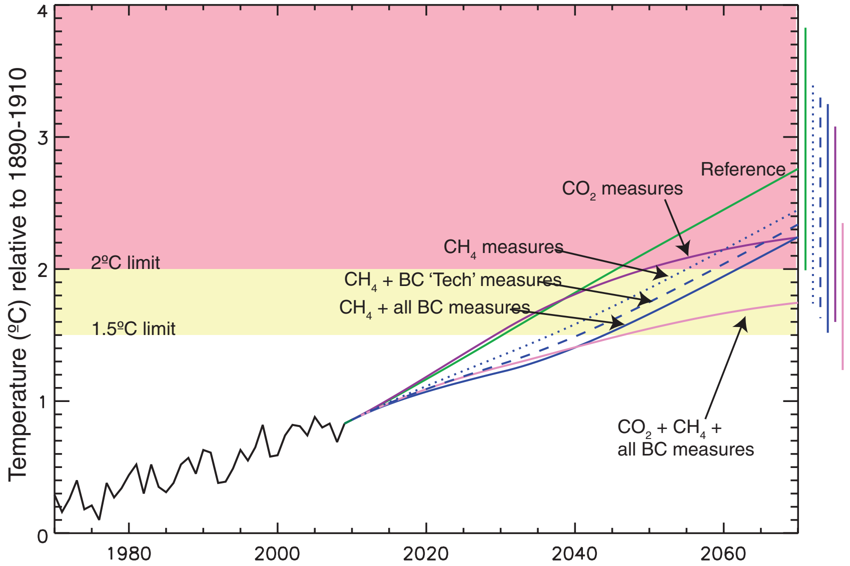

Fig. 1. Observed temperatures (*42*) through 2009 and projected temperatures thereafter under various scenarios, all relative to the 1890–1910 mean. Results for future scenarios are the central values from analytic equations estimating the response to forcings calculated from composition-climate modeling and literature assessments (*7*). The rightmost bars give 2070 ranges, including uncertainty in radiative forcing and climate sensitivity. A portion of the uncertainty is systematic, so that overlapping ranges do not mean there is no significant difference (for example, if climate sensitivity is large, it is large regardless of the scenario, so all temperatures would be toward the high end of their ranges; see www.giss.nasa.gov/staff/dshindell/Sci2012).

Table 1. ECHAM and GISS forcing (W/m^2) at 2030 due to the measures relative to the reference. Dashes indicate forcing not calculated.

	CH_4 measures	CH_4+BC Tech measures	All measures
ECHAM ozone	–0.09	–0.10	–0.10
GISS ozone	–0.10	–0.17	–0.19
ECHAM direct aerosols*	–0.01	–0.06	–0.15
GISS direct aerosols*	–0.01	–0.06	–0.17
(BC, OC, sulfate, nitrate)	(0.00, 0.00, –0.02, 0.00)	(-0.10, 0.06, –0.02, 0.01)	(–0.22, 0.07, –0.02, 0.01)
ECHAM CH_4†	–0.22	–0.22	–0.20
GISS CH_4†	–0.20	–0.20	–0.18
GISS indirect and semidirect aerosols	—	–0.14 ± 0.03	–0.16 ± 0.04
GISS BC albedo	—	–0.010	–0.017
(effective forcing ×5)		(–0.05)	(–0.09)
GISS net‡	–0.32	–0.60	–0.77

*For aerosols, the value for ECHAM is the sum of the direct BC+OC+sulfate forcing. For GISS, the same sum is presented first, and individual components are listed afterward (the ECHAM model has more realistic internally mixed aerosols, so components are not separable). †CH_4 forcing at 2030 is roughly 75% of the forcing that is eventually realized from CH_4 emission changes through 2030. ‡The net forcing given here includes the effective value for BC albedo forcing. Uncertainties due to internal variability in the models are 0.01 W/m^2 or less for direct forcings and 0.001 W/m^2 for BC albedo forcing.

"effective" forcing is five times the instantaneous value (*25*, *26*). Albedo forcing can be important regionally (Fig. 2), especially in the Arctic and the Himalayas, where the measures decrease forcing up to 4 W/m^2 (not including the factor of 5). Such large regional impacts are consistent with other recent studies (*27*, *28*) and would reduce snow and ice melting.

Roughly half the forcing is relatively evenly distributed (from the CH_4 measures). The other half is highly inhomogeneous, especially the strong BC forcing, which is greatest over bright desert and snow or ice surfaces. Those areas often exhibit the largest warming mitigation, making the regional temperature response to aerosols and ozone quite distinct from the more homogeneous response to well-mixed greenhouse gases (Fig. 2) [although the impact of localized forcing extends well beyond the forcing location (*29*)]. BC albedo and direct forcings are large in the Himalayas, where there is an especially pronounced response in the Karakoram, and in the Arctic, where the measures reduce projected warming over the next three decades by approximately two thirds and where regional temperature response patterns correspond fairly closely to albedo forcing (for example, they are larger over the Canadian archipelago than the interior and larger over Russia than Scandinavia or the North Atlantic).

The largest precipitation responses to the CH_4 plus BC measures are seen in South Asia, West Africa, and Europe (Fig. 2). The BC measures greatly reduce atmospheric forcing—defined as top-of-the-atmosphere minus surface forcing—in those parts of Asia and Africa (fig. S4), which can strongly influence regional precipitation patterns (*30*–*32*). In comparison with a semiempirical estimate (*33*), the two composition-climate models represent present-day atmospheric forcing reasonably well (fig. S4). The response to greenhouse gases alone shows different spatial structure over South Asia and Europe and is much weaker everywhere (per unit of global mean forcing). The BC measures moderate a shift in the monsoon westward away from Southeast Asia into India seen during 20th- and 21st-century GISS-E2 simulations, with especially strong impacts at the Indian west coast and from Bengal to the northwest along the Himalayan foothills. Climate models' simulations of monsoon responses to absorbing aerosols vary considerably (*30*–*32*). The results suggest that the BC measures could reduce drought risk in Southern Europe and the

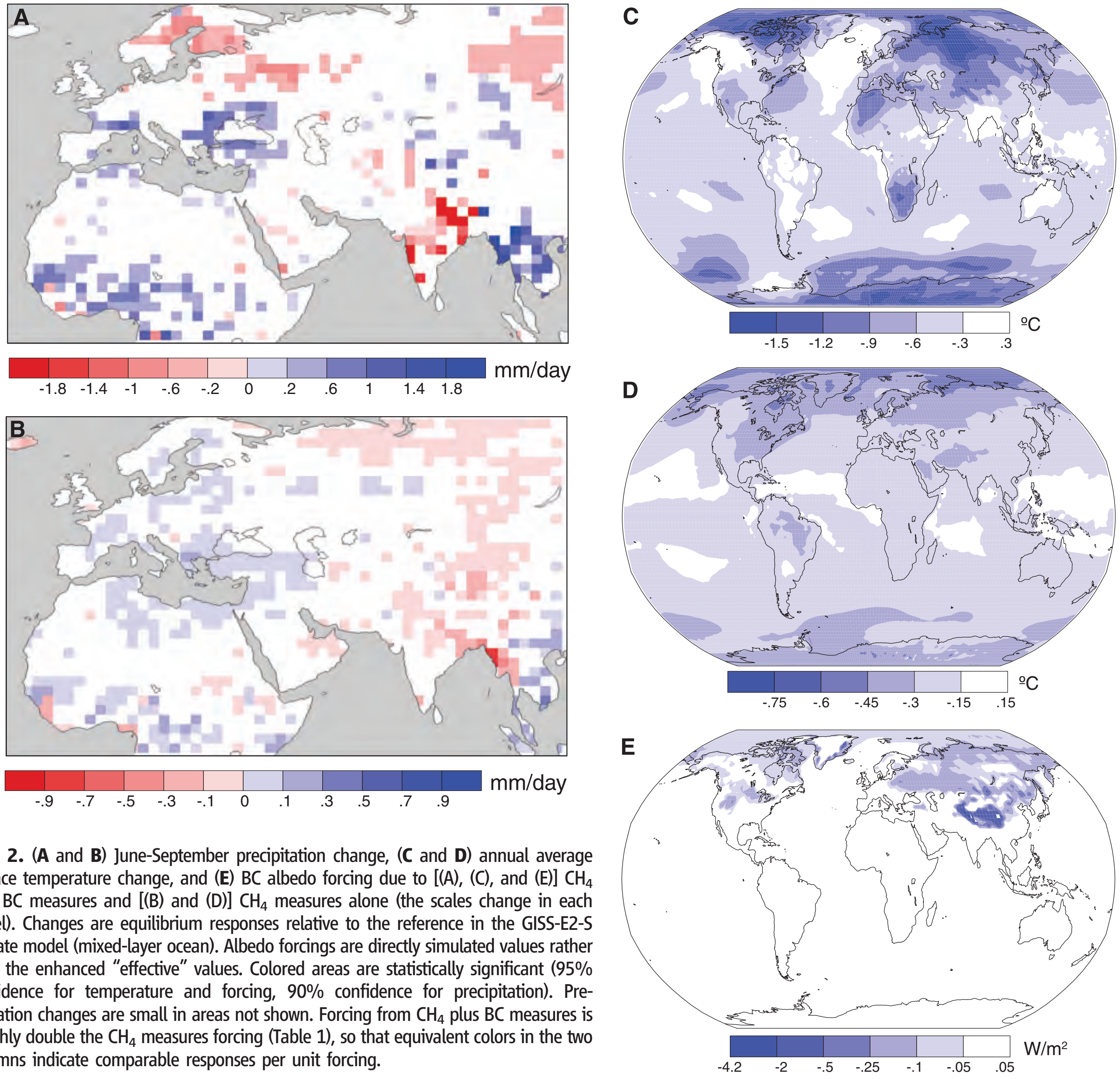

Fig. 2. (**A** and **B**) June-September precipitation change, (**C** and **D**) annual average surface temperature change, and (**E**) BC albedo forcing due to [(A), (C), and (E)] CH_4 plus BC measures and [(B) and (D)] CH_4 measures alone (the scales change in each panel). Changes are equilibrium responses relative to the reference in the GISS-E2-S climate model (mixed-layer ocean). Albedo forcings are directly simulated values rather than the enhanced "effective" values. Colored areas are statistically significant (95% confidence for temperature and forcing, 90% confidence for precipitation). Precipitation changes are small in areas not shown. Forcing from CH_4 plus BC measures is roughly double the CH_4 measures forcing (Table 1), so that equivalent colors in the two columns indicate comparable responses per unit forcing.

Sahel while reversing shifting monsoon patterns in South Asia.

Global mean impacts of packages of measures. Having established the credibility of the analytic climate calculations at the global scale [air quality simulations were shown to be realistic in (*20*)], we now briefly compare the global effects of the separate packages of measures (Table 2). The CH_4 measures contribute more than half the estimated warming mitigation and have the smallest relative uncertainty. BC Tech measures have a larger climate impact and a substantially smaller fractional uncertainty than that of the Reg measures because aerosols contribute a larger portion of the total forcing in the Reg case (and uncertainty in radiative forcing by BC or OC is much larger than for CH_4 or ozone). In the Reg case, the temperature range even includes the possibility of weak global warming, although the distribution shows a much larger probability of cooling.

For yield losses of four staple crops due to ozone, the mean values for CH_4 and BC Tech measures are comparable, whereas BC Reg measures have minimal impact. The health benefits from BC measures are far larger that those from the CH_4 measures because health is more sensitive to reduced exposure to $PM_{2.5}$ than to ground-level ozone. The large ranges for health impacts stem primarily from uncertainty in concentration-response relationships. The estimated 0.7 to 4.7 million annually avoided premature deaths are substantial in comparison with other causes of premature death projected for 2030, including tuberculosis (0.6 million), traffic accidents (2.1 million), or tobacco use (8.3 million) (*34*). There would also be large health benefits from improved indoor air quality. Because of limited data, we only estimated these for India and China, where implementation of all BC measures leads to an additional 373,000 annually avoided premature deaths (*7*).

Cost and benefit valuation. Economic analyses use the value of a statistical life (VSL) for health, world market prices for crops, and the social cost of carbon (SCC) along with global mean impacts relative to CO_2 for climate (*7*). Valuation is dominated by health effects and hence by the BC measures (Table 2). Climate valuation is large for the CH_4 measures, although it depends strongly on the metrics used. If instead of the 100-year GWP, the 100-year global temperature potential (GTP) of CH_4 is used (*35*), the value becomes $159 billion. Similarly, benefits scale with differing choices for the SCC. Climate benefits for the BC measures are based on the CH_4 measures' climate benefits times the relative global mean climate impact of the BC measures because published GWP or GTP values do not cover all species and ignore some factors affecting climate (such as aerosol indirect effects), and the ratio of the temperature responses is similar to the ratio of the integrated forcing due to a single year's emissions (GWP). This method still neglects regional effects of these pollutants on temperatures, precipitation, and sunlight available for photosynthesis.

Because the CH_4 measures largely influence CH_4 emissions alone, and CH_4 emissions anywhere have equal impact, it is straightforward to value CH_4 reductions by the metric ton. Climate benefits dominate, at $2381 per metric ton, with health second and crops third. The climate benefit per metric ton is again highly dependent on metrics. For example, instead of a $265 SCC (*36*)—a typical value assuming a near-zero discount rate—a value of $21 consistent with a ~3% discount rate could be used. Because discounting emphasizes near-term impacts, we believe a 20-year GWP or GTP should be used with the $21 SCC, in which case the valuation is $599 or $430 per metric ton, respectively. Health and agricultural benefits could also be discounted to account for the time dependence of the ozone response. Using a 5% discount rate, the mean health and agricultural benefits decrease relative to the undiscounted Table 2 values to $659 and $18 per metric ton, respectively. Climate benefits always exceed the agricultural benefits per metric ton, but climate values can be less or more than health benefits depending on the metric choices (the health benefits are similarly dependent on the assumed VSL).

A very conservative summation of benefits, using $430 for climate and discounted health and agricultural values, gives a total benefit of ~$1100 per metric ton of CH_4 (~$700 to $5000 per metric ton, using the above analyses). IEA estimates (*37*) indicate roughly 100 Tg/year of CH_4 emissions can be abated at marginal costs below $1100, with more than 50 Tg/year costing less than 1/10 this valuation (including the value of CH_4 captured for resale). Analysis using more recent cost information in the GAINS model (*38*, *39*) finds that the measures analyzed here could reduce 2030 CH_4 emissions by ~110 Tg at marginal costs below $1500 per metric ton, with 90 Tg below $250. The full set of measures reduce emissions by ~140 Tg, indicating that most would produce benefits greater than—and for approximately two-thirds of reductions far greater than—the abatement costs. Of course, the benefits would not necessarily accrue to those incurring costs.

Prior work valued CH_4 reductions at $81 ($48 to $116) per metric ton, including agriculture (grains), forestry, and nonmortality health benefits using 5% discounting (*40*). Their agricultural valuation was ~$30 ($1 to $42) per metric ton. Hence, our agriculture values are smaller but well within their large range. Those results suggest that forestry and nonmortality health effects contribute another ~$50 per metric ton of CH_4. Nonlinearities imply all valuations may shift somewhat as the background atmospheric composition changes.

GAINS estimates show that improved efficiencies lead to a net cost savings for the brick kiln and clean-burning stove BC measures. These account for ~50% of the BC measures' impact. The regulatory measures on high-emitting vehicles and banning of agricultural waste burning, which require primarily political rather than economic investment, account for another 25%. Hence, the bulk of the BC measures could probably be implemented with costs substantially less than the benefits given the large valuation of the health impacts (Table 2).

CH_4 measures by sector and region. It is also straightforward to separate the impact of CH_4 reductions in each region and sector on forcing. Because CH_4 is relatively well mixed globally, other impacts (such as crop yields) have the same proportionality as forcing. Emissions reductions in the coal mining and oil/gas production sectors

Table 2. Global impacts of measures on climate, agriculture, and health and their economic valuation. Valuations are annual values in 2030 and beyond, due to sustained application of the measures, which are nearly equal to the integrated future valuation of a single year's emissions reductions (without discounting). Climate valuations for CH_4 use GWP100 and an SCC of $265 per metric ton (*36*). Crop and health valuations use 95% confidence intervals, whereas climate valuations use ~67% uncertainty range. All values are in 2006 dollars.

	CH_4 measures	BC Tech measures	BC Reg measures
Physical Impacts			
Avoided warming in 2050 (°C)	0.28 ± 0.10	0.12 (+0.06/–0.09)	.07 (+.04/–0.09)
Annually avoided crop yield losses (millions metric tons; sum of wheat, rice, maize, and soy)	27 (+42/–20)	24 (+72/–21)	2 (+13/–3)
Annually avoided premature deaths (thousands)	47 (+40/–34)	1720 (+1529/–1188)	619 (+639/–440)
Valuation			
Climate, billions $US	331 ± 118	142 (+71/–106)	83 (+47/–106)
($US per metric ton CH_4)	(2381 ± 850)		
Crops, billions $US	4.2 ± 1.2	3.6 ± 2.6	0.4 ± 0.6
($US per metric ton CH_4)	(29 ± 8)		
Health, billions $US	148 ± 99	3717 (+3236/–2563)	1425 (+1475/–1015)
($US per metric ton CH_4)	(1080 ± 721)		

have the largest impacts, with municipal waste third (Fig. 3). Globally, sectors encompassing fossil fuel extraction and distribution account for nearly two thirds of the benefits because technology to control emissions from these sectors is readily available.

Examining benefits by sector and region, the largest by a considerable amount are from coal mining in China (Fig. 3). Oil and gas production in Central Africa, the Middle East, and Russia are next, followed by coal mining in South Asia, gas transmission in Russia (in high-pressure mains), and municipal waste in the United States and China. Ranking is obviously quite sensitive to regional groupings and country size, and there is substantial uncertainty in emissions from certain sectors in some regions. In particular, using national emission factors (instead of the Intergovernmental Panel on Climate Change default methodology) would lower the coal-mining potential from India and Southern Africa substantially. Nonetheless, those eight regional/sectoral combinations alone represent 51% of the total impact from all CH_4 measures.

Regional and national impacts. Upon examination of impacts of the CH_4 plus BC measures, avoided warming is greatest in central and northern Asia, southern Africa, and around the Mediterranean (Fig. 4, fig. S5, and table S5). Three of the top four national-level responses are in countries with strong BC albedo forcing (Tajikistan, Kyrgyzstan, and Russia). In contrast, the atmospheric forcing linked to regional hydrologic cycle disruption is reduced most strongly in south Asia and west Africa, where the measures greatly decrease BC emissions. Total numbers of avoided premature deaths are greatest in developing nations in Asia and Africa with large populations and high PM concentrations (and large emissions changes). Turning to per capita impacts, premature deaths are reduced most strongly in countries of south Asia, followed by central Africa, then east and southeast Asia, in a pattern quite similar to the atmospheric forcing impacts.

For crop production, China, India, and the United States, followed by Pakistan and Brazil, realize the greatest total metric tonnage gains. Looking instead at percentage yield changes, impacts are largest in the Middle East, with large changes also in central and south Asia. There is a large impact on percentage crop yields in Mexico that is quite distinct from neighboring countries, reflecting the influence of local emission changes. Impacts vary greatly between crops for changes in total production (fig. S6), with largest impacts occurring where the distribution and seasonal timing of crop production coincide with high ozone concentrations (*7*). Percentage yield changes are more consistent, however. Additional crop yield benefits would result from the avoided climate change, but they are not considered here.

Avoided warming is spread much more evenly over the Earth than other impacts. Both climate benefits in terms of reductions in regional atmospheric forcing and air quality–related human health benefits are typically largest in the countries of south Asia and central Africa, whereas agricultural benefits are greatest in the Middle East, where ozone reductions are large. Because many nations in these areas face great development challenges, realization of these benefits would be especially valuable in those areas.

Discussion. The results clearly demonstrate that only a small fraction of air quality measures provide substantial warming mitigation. Nonetheless, the CH_4 and BC emissions reduction measures examined here would have large benefits to global and regional climate, as well as to human health and agriculture. The CH_4 measures lead to large global climate and agriculture benefits and relatively small human health benefits, all with high confidence and worldwide distribution. The BC measures are likely to provide substantial global climate benefits, but uncertainties are much larger. However, the BC measures cause large regional human health benefits, as well as reduce regional hydrology cycle disruptions and cryosphere melting in both the Arctic and the Himalayas and improve regional agricultural yields. These benefits are more certain and are typically greatest in and near areas where emissions are reduced. Results are robust across the two composition-climate models. Protecting public health and food supplies may take precedence over avoiding climate change in most countries, but knowing that these measures also mitigate climate change may help motivate policies to put them into practice.

We emphasize that the CH_4 and BC measures are both distinct from and complementary to CO_2 measures. Analysis of delayed implemen-

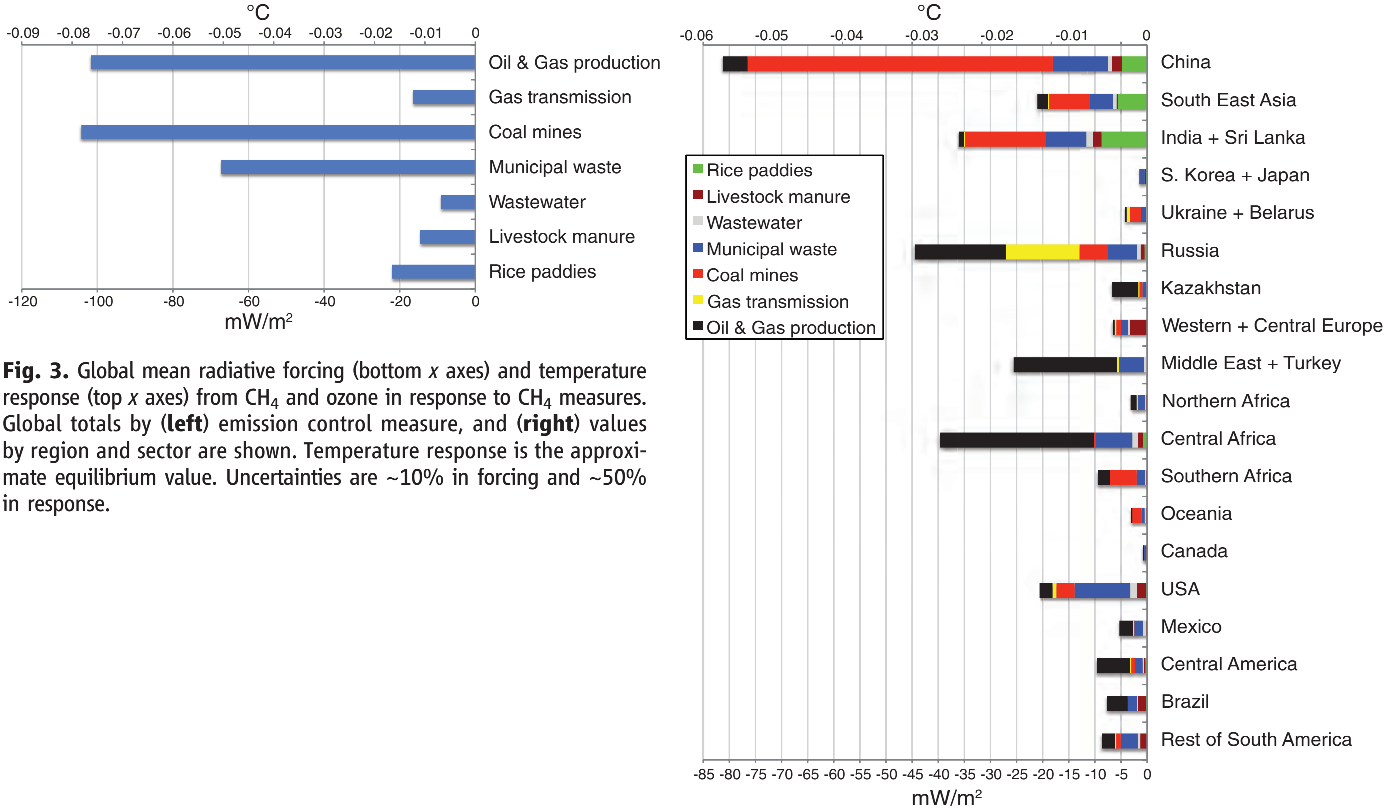

Fig. 3. Global mean radiative forcing (bottom *x* axes) and temperature response (top *x* axes) from CH_4 and ozone in response to CH_4 measures. Global totals by (**left**) emission control measure, and (**right**) values by region and sector are shown. Temperature response is the approximate equilibrium value. Uncertainties are ~10% in forcing and ~50% in response.

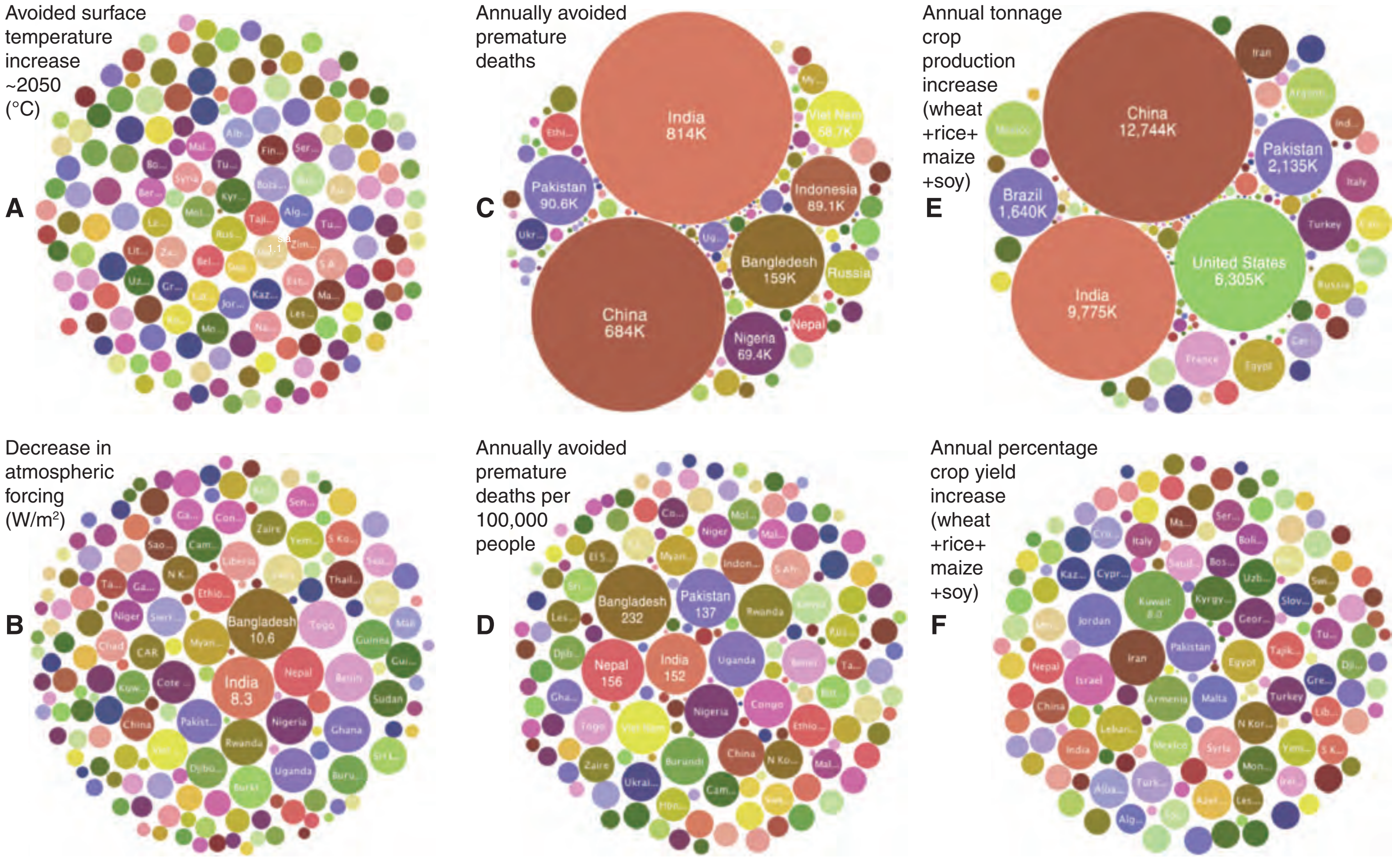

Fig. 4. National benefits of the CH_4 plus BC measures versus the reference scenario. Circle areas are proportional to values for (**A** and **B**) climate change, (**C** and **D**) human health (values for population over age 30), and (**E** and **F**) agriculture. Surface temperature changes are from the GISS-E2-S simulation. Health, agriculture, and atmospheric forcing impacts are based on the average of GISS and ECHAM concentration changes and are for 2030 and beyond. Uncertainties are ~60% for global mean temperatures, with national scale uncertainties likely greater, ~60% for atmospheric forcing, ~70% for health, and roughly –70%/+100% for crops [see (*7*) for factors included in uncertainties, most of which are systematic for atmospheric forcing, health, and agriculture so that much smaller differences between regions are still significant]. Interactive versions providing values for each country are at www.giss.nasa.gov/staff/dshindell/Sci2012, whereas alternate presentations of these data are in fig. S5 and table S5.

tation of the CH_4 and BC measures (fig. S3) shows that early adoption provides much larger near-term benefits but has little impact on long-term temperatures (*20*). Hence, eventual peak warming depends primarily on CO_2 emissions, assuming air quality–related pollutants are removed at some point before peak warming.

Valuation of worldwide health and ecosystem impacts of CH_4 abatement is independent of where the CH_4 is emitted and usually outweighs abatement costs. These benefits are therefore potentially suitable for inclusion in international mechanisms to reduce CH_4 emissions, such as the Clean Development Mechanism under the United Nations Framework Convention on Climate Change or the Prototype Methane Financing Facility (*41*). Many other policy alternatives exist to implement the CH_4 and BC measures, including enhancement of current air quality regulations. The realization that these measures can slow the rate of climate change and help keep global warming below 2°C relative to preindustrial in the near term, provide enhanced warming mitigation in the Arctic and the Himalayas, and reduce regional disruptions to traditional rainfall patterns—in addition to their local health and local-to-global agricultural benefits—may help prompt widespread and early implementation so as to realize these manifold benefits.

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Supporting Online Material

www.sciencemag.org/cgi/content/full/335/6065/183/DC1
Materials and Methods
Figs. S1 to S6
Tables S1 to S5
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Periodic Emission from the Gamma-Ray Binary 1FGL J1018.6–5856

The Fermi LAT Collaboration*

Gamma-ray binaries are stellar systems containing a neutron star or black hole, with gamma-ray emission produced by an interaction between the components. These systems are rare, even though binary evolution models predict dozens in our Galaxy. A search for gamma-ray binaries with the Fermi Large Area Telescope (LAT) shows that 1FGL J1018.6–5856 exhibits intensity and spectral modulation with a 16.6-day period. We identified a variable x-ray counterpart, which shows a sharp maximum coinciding with maximum gamma-ray emission, as well as an O6V((f)) star optical counterpart and a radio counterpart that is also apparently modulated on the orbital period. 1FGL J1018.6–5856 is thus a gamma-ray binary, and its detection suggests the presence of other fainter binaries in the Galaxy.

Two types of interacting binaries containing compact objects are expected to emit gamma rays (*1*): microquasars—accreting black holes or neutron stars with relativistic jets (*2*)—and rotation-powered pulsars interacting with the wind of a binary companion (*3*). Microquasars should typically be powerful x-ray sources when active, and hence such gamma ray–emitting systems may already be known x-ray binaries. Indeed, the bright x-ray source Cygnus X-3 is now known to be such a source (*4*, *5*). The existence of pulsars interacting with stellar companions of early spectral types is predicted as an initial stage in the formation of high-mass x-ray binaries (HMXBs) containing neutron stars (*6*). These interacting pulsars are predicted to be much weaker x-ray emitters and may not yet be known or classified x-ray sources. Gamma-ray binaries may thus not be as rare as they appear to be, and many systems may await detection.

A gamma-ray binary is expected to show orbitally modulated gamma-ray emission due to a combination of effects, including changes in viewing angle and, in eccentric orbits, the degree of the binary interaction, both of which depend on binary phase. Periodic gamma-ray modulation has indeed been seen in LS 5039 (period 3.9 days), LS I +61° 303 (26.5 days), and Cygnus X-3 (4.8 hours) (*4*, *7*, *8*), and gamma-ray emission is at least orbital phase–dependent for the PSR B1259–63 system (3.4 years) (*9*). However, the putative gamma-ray binary HESS J0632+057, for which a 321-day x-ray period is seen, has not yet been shown to exhibit periodic gamma-ray emission (*10*). PSR B1259–63 contains a pulsar, and LS 5039 and LS I +61° 303 are suspected, but not proved, to contain pulsars, whereas Cygnus X-3 is a black hole candidate. A search for periodic modulation of gamma-ray flux from LAT sources may thus lead to the detection of further gamma-ray binaries, potentially revealing the predicted HMXB precursor population. The first Fermi LAT (*11*) catalog of gamma-ray sources ("1FGL") contains 1451 sources (*12*), a large fraction of which do not have confirmed counterparts at other wavelengths and thus are potentially gamma-ray binaries.

To search for modulation, we used a weighted photon method to generate light curves for all 1FGL sources in the energy range 0.1 to 200 GeV (*13*). We then calculated power spectra for all sources. From an examination of these, in addition to modulation from the known binaries LS I +61° 303 and LS 5039, we noted the presence of a strong signal near a period of 16.6 days from 1FGL J1018.6–5856 (Fig. 1). 1FGL J1018.6–5856 has a cataloged 1- to 100-GeV flux of 2.9×10^{-8} photons cm^{-2} s^{-1}, making it one of the brighter LAT sources. The source's location at right ascension (R.A.) = 10^h 18.7^m, declination (decl.) = –58° 56.30′ (J2000; ±1.8′, 95% uncertainty) means that it lies close to the galactic plane ($b = -1.7°$), marking it as a good candidate for a binary system. 1FGL J1018.6–5856 has been noted to be positionally coincident with the supernova remnant G284.3–1.8 (*12*) and the TeV source HESS J1018–589 (*14*), although it has not been shown that these sources are actually related.

The modulation at a period of 16.6 days has a power more than 25 times the mean value of the power spectrum and has a false-alarm probability of 3×10^{-8}, taking into account the number of statistically independent frequency bins. From both the power spectrum itself (*15*) and from fitting the light curve, we derived a period of 16.58 ± 0.02 days. The folded light curve (Fig. 1) has a sharp peak together with additional broader modulation. We modeled this to determine the epoch of maximum flux T_{max} by fitting a function consisting of the sum of a sine wave and a Gaussian function, and obtained T_{max} = modified Julian date (MJD) 55403.3 ± 0.4.

The gamma-ray spectrum of 1FGL J1018.6–5856 shows substantial curvature through the LAT passband. To facilitate discussion of the lower-energy (<1 GeV) and higher-energy (>1 GeV) gamma rays, we adopted as our primary model a broken power law with photon indices $\Gamma_{0.1-1}$ and Γ_{1-10} for energies below and above 1 GeV, respectively. The best-fit values (*13*) are $\Gamma_{0.1-1} = 2.00 \pm 0.04_{stat} \pm 0.08_{syst}$ and $\Gamma_{1-10} = 3.09 \pm 0.06_{stat} \pm 0.12_{syst}$, along with an integral energy flux above 100 MeV of $(2.8 \pm 0.1_{stat} \pm 0.3_{syst}) \times 10^{-10}$ erg cm^{-2} s^{-1}. A power law with exponential cutoff (*7*, *8*), $dN/dE = N_0(E/\mathrm{GeV})^{-\Gamma} \exp(-E/E_c)$, gives an acceptable fit with $\Gamma = 1.9 \pm 0.1$ and $E_c = 2.5 \pm 0.3$ GeV (statistical errors only). Although this spectral shape is qualitatively similar to that of pulsars and of LS I +61° 303 and LS 5039, so far no detection of pulsed gamma-ray emission has been reported (*16*).

To investigate variability on the 16.6-day period, we folded the data into 10 uniform bins in orbital phase and then refit the broken power-law parameters within each phase bin. The resulting

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LETTER

Coal to gas: the influence of methane leakage

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Abstract Carbon dioxide (CO_2) emissions from fossil fuel combustion may be reduced by using natural gas rather than coal to produce energy. Gas produces approximately half the amount of CO_2 per unit of primary energy compared with coal. Here we consider a scenario where a fraction of coal usage is replaced by natural gas (i.e., methane, CH_4) over a given time period, and where a percentage of the gas production is assumed to leak into the atmosphere. The additional CH_4 from leakage adds to the radiative forcing of the climate system, offsetting the reduction in CO_2 forcing that accompanies the transition from coal to gas. We also consider the effects of: methane leakage from coal mining; changes in radiative forcing due to changes in the emissions of sulfur dioxide and carbonaceous aerosols; and differences in the efficiency of electricity production between coal- and gas-fired power generation. On balance, these factors more than offset the reduction in warming due to reduced CO_2 emissions. When gas replaces coal there is additional warming out to 2,050 with an assumed leakage rate of 0%, and out to 2,140 if the leakage rate is as high as 10%. The overall effects on global-mean temperature over the 21st century, however, are small.

Hayhoe et al. (2002) have comprehensively assessed the coal-to-gas issue. What has changed since then is the possibility of substantial methane production by high volume hydraulic fracturing of shale beds ("fracking") and/or exploitation of methane reservoirs in near-shore ocean sediments. Fracking, in particular, may be associated with an increase in the amount of attendant gas leakage compared with other means of gas production (Howarth et al. 2011). In Hayhoe et al., the direct effects on global-mean temperature of differential gas leakage between coal and gas production are very small (see their Fig. 4). Their estimates of gas

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leakage, however, are less than more recent estimates. Here, we extend and update the analysis of Hayhoe et al. to examine the potential effects of gas leakage on the climate, and on uncertainties arising from uncertainties in leakage percentages.

We begin with a standard "no-climate-policy" baseline emissions scenario, viz. the MiniCAM Reference scenario (MINREF below) from the CCSP2.1a report (Clarke et al. 2007). (Hayhoe et al. used the MiniCAM A1B scenario, Nakićenović and Swart 2000.) We chose MINREF partly because it is a more recent "no-policy" scenario, but also because there is an extended version of MINREF that runs beyond 2,100 out to 2,300 (Wigley et al. 2009). The longer time horizon is important because of the long timescales involved in the carbon cycle where changes to CO_2 emissions made in the 21st century can have effects extending well into the 22nd century. (A second baseline scenario, the MERGE Reference scenario from the CCSP2.1a report, is considered in the Electronic Supplementary Material).

In MINREF, coal combustion provides from 38% (in 2010) to 51% (in 2100) of the emissions of CO_2 from fossil fuels. (The corresponding percentages for gas are 19 to 21%, and for oil are 43 to 28%.) For our coal-to-gas scenario we start with their contributions to energy. It is important here to distinguish between primary energy (i.e., the energy content of the resource) and final energy (the amount of energy delivered to the user at the point of production). For a transition from coal to gas, we assume that there is no change in final energy. As electricity generation from gas is more efficient than coal-fired generation, the increase in primary energy from gas will be less than the decrease in primary energy from coal — the differential depends on the relative efficiencies with which energy is produced.

To calculate the change in fossil CO_2 emissions for any transition scenario we use the following relationship relating CO_2 emissions to primary energy (P)…

$$\mathrm{ECO2} = \mathrm{A\ Pcoal} + \mathrm{B\ Poil} + \mathrm{C\ Pgas} \tag{1}$$

where A, B and C are representative emissions factors (emissions per unit of primary energy) for coal, oil and gas. The emissions factors relative to coal that we use are 0.75 for oil and 0.56 for gas, based on information in EPA's AP-42 Report (EPA 2005). Using the MINREF emissions for CO_2 and the published primary energy data give a best fit emissions factor for coal of 0.027 GtC/exajoule, well within the uncertainty range for this term.

To determine the change in CO_2 emissions in moving from coal to gas under the constraint of no change in final energy we use the equivalent of Eq. (1) expressed in terms of final energy (F). This requires knowing the efficiencies for energy production from coal, oil and gas (i.e., final energy/primary energy). If F=P×(efficiency), then we have

$$\mathrm{ECO2} = (\mathrm{A/a})\mathrm{Fcoal} + (\mathrm{B/b})\mathrm{Foil} + (\mathrm{C/c})\mathrm{Fgas} \tag{2}$$

where a, b and c are the efficiencies for energy production from coal, oil and gas. For changes in final energy (ΔF) in the coal-to-gas case, ΔFoil is necessarily zero. To keep final energy unchanged, therefore, we must have ΔFgas = $-\Delta$Fcoal. Hence, from Eq. (2) …

$$\mathit{\Delta}\mathrm{ECO2} = (\mathit{\Delta}\mathrm{Fcoal})(\mathrm{A/a} - \mathrm{C/c}) \tag{3}$$

or …

$$\mathit{\Delta}\mathrm{ECO2} = \mathrm{A}\ \mathit{\Delta}\mathrm{Pcoal}[1 - (\mathrm{C/A})/(\mathrm{c/a})] \tag{4}$$

As ΔPcoal is negative, the first term here is the reduction in CO_2 emissions from the reduction in coal use, while the second term is the partially compensating increase in CO_2

emissions from the increase in gas use. Our best-fit value for A is 0.027 GtC/exajoule, and C/A=0.56. To apply Eq. (4) we need to determine a reasonable value for the relative gas-to-coal efficiency ratio (c/a), which we assume does not change appreciably over time. For electricity generation, the primary sector for coal-to-gas substitution, Hayhoe et al. (2002, Table 2) give representative efficiencies of 32% for coal and 60% for gas. Using these values, Eq. (4) becomes …

$$\Delta ECO2 = 0.027\ \Delta Pcoal[1 - 0.299] \tag{5}$$

for ΔECO2 in GtC and ΔP in exajoules. Thus, for a unit reduction in coal emissions, there is an increase in emissions from gas combustion of about 0.3 units.

To complete our calculations, we need to estimate the changes in methane, sulfur dioxide and black carbon emissions that would follow the coal-to-gas conversion. Consider methane first. Methane is emitted to the atmosphere as a by-product of coal mining and gas production. Although these fugitive emissions are relatively small, they are important because methane is a far more powerful forcing agent per unit mass than CO_2.

For coal mining we use information from Spath et al. (1999; Figs. C1 and C4). A typical US coal-fired power plant emits 1,100 gCO2/kWh, with an attendant release of methane of 2.18 gCH4/kWh, almost entirely from mining. Thus, for each GtC of CO_2 emitted from a coal-fired power plant, 7.27 TgCH4 are emitted from mining. Spath et al. give other information that can used to check the above result. They give values of 1.91 gCH4 released per ton of coal mined from surface mines, and 4.23 gCH4 per ton from deep mines. As 65% of coal comes from deep mines, the weighted average release is 3.42 gCH4/ton. Since 1 ton of coal, when burned, typically produces 1.83 kgCO2, the amount of fugitive methane per GtC of CO_2 emissions from coal-fired power plants is 6.85 TgCH4/GtC, consistent with the previous result. For our calculations we use the average of these two results, 7.06 TgCH4/GtC; i.e., if CO_2 emissions from coal-fired power generation are reduced by 1 GtC, we assume a concomitant decrease in CH_4 emissions of 7.06 TgCH4. We assume that this value for the USA is applicable for other countries.

For leakage associated with gas extraction and transport we note that every kg of gas burned produces 12/16 kgC of CO_2. If the leakage rate is "p" percent, then, for any given increase in CO_2 emissions from gas combustion, the amount of fugitive methane released is (p/100) (16/12) 1000=13.33 (p) TgCH4/GtC. For a leakage rate of 2.5%, for example (roughly the present leakage rate for conventional gas extraction), this is 33.3 TgCH4/GtC. Because the CO_2 emissions change from gas combustion is much less than that for coal (about 30%; see Eq. (5)), for the 2.5% leakage case this would make the coal mining and gas leakage effects on CH_4 quite similar (but of opposite sign), in accord with Hayhoe et al. (2002, Table 1).

SO_2 emissions are important because coal combustion produces substantial SO_2, whereas SO_2 emissions from gas combustion are negligible. Reducing energy production from coal has compensating effects — reduced CO_2 emissions leads to reduced warming in the long term, but this is offset by the effects of reduced SO_2 emissions which lead to lower aerosol loadings in the atmosphere and an attendant warming (Wigley 1991). For CO_2 and SO_2, emissions factors for coal (from Hayhoe et al. 2002, Table 1) are 25 kgC/GJ and 0.24 kgS/GJ. For each GtC of CO_2 produced from coal combustion, therefore, there will be 19.2 TgS of SO_2 emitted. We can check this using emissions factors from Spath et al. (1999, Figs. C1 and C2). For a typical coal-fired power plant these are 7.3 gSO2/kWh and 1,100 gCO2/kWh. Hence, for each GtC of CO_2 produced from coal combustion, SO_2 emissions will be 12.17 TgS. Effective global emissions factors can also be obtained from published emissions scenarios. For example, for changes over 2000 to 2010 in the MINREF scenario, the emissions factor for coal combustion is approximately 11.6 TgS/GtC.

From these different estimates it is clear that there is considerable uncertainty in the SO_2 emissions factor, echoing in part the widely varying sulfur contents in coal. Furthermore, for future emissions from coal combustion the SO_2 emissions factor is likely to decrease markedly due to the imposition of SO_2 pollution controls (as explained, for example, in Nakićenović and Swart 2000). It is difficult to quantify this effect, a difficulty highlighted, for example, by the fact that, in the second half of the 21st century, many published scenarios show increasing CO_2 emissions, but decreasing SO_2 emissions — with large differences between scenarios in the relative changes.

For the coal-to-gas transition, it is not at all clear how to account for the effects that SO_2 pollution controls, that will likely go on in parallel with any transition from coal to gas, will have on the SO_2 emissions factor. However, future coal-fired plants will certainly employ such controls, so emissions factors for SO_2 will decrease over time. To account for this we assume a value of 12 TgS/GtC for the present (2010) declining linearly to 2 TgS/GtC by 2,060 and remaining at this level thereafter. This limit and the attainment date are consistent with the fact that many of the SRES scenarios tend to stabilize SO_2 emissions at a finite, non-zero value at around this time.

For black carbon (BC) aerosol emissions we use the relationship between BC and SO_2 emissions noted by Hayhoe et al. (2002, p. 125) and make BC forcing proportional to SO_2 emissions. Using best-estimate forcings from the IPCC Fourth Assessment Report, this means that the increase in sulfate aerosol forcing changes due to SO_2 emissions reductions are reduced by approximately 30% by the attendant changes in BC emissions. This is a larger BC effect than in Hayhoe et al. However, compared with the large overall uncertainty in aerosol forcing, the difference between what we obtain here and the results of Hayhoe et al. are relatively small.

For our coal-to-gas emissions scenario we assume that primary energy from coal is reduced linearly (in percentage terms) by 50% over 2010 to 2050 (1.25%/yr), and that the reduction in final energy is made up by extra energy from gas combustion. (A second, more extreme scenario is considered in the Electronic Supplementary Material). In this way, there are no differences in final energy between the MINREF baseline scenario and the coal-to-gas perturbation scenario. Hayhoe et al. consider scenarios where coal production reduces by 0.4, 1.0 and 2.0%/yr over 2000 to 2025. After 2050 we assume no further percentage reduction in coal-based energy (i.e., the reduction in emissions from coal relative to the baseline scenario remains at 50%). This is an idealized scenario, but it is sufficiently realistic to be able to assess the relative importance of different gas leakage rates. We consider leakage rates of zero to 10%,

Baseline and perturbed (coal to gas) primary energy scenarios for coal and gas are shown in Fig. 1, together with the corresponding fossil-fuel CO_2 emissions. The changes in primary energy breakdown are large: e.g., in 2100, primary energy from coal is 37% more than from gas in the baseline case, but 50% less than gas in the perturbed case. The corresponding reduction in emissions is less striking. In the perturbed case, 2100 emissions are reduced only by 19%. (Cases where there are larger emissions reductions are considered in the Electronic Supplementary Material).

To determine the consequences of the coal-to-gas scenario we use the MAGICC coupled gas-cycle/upwelling-diffusion climate model (Wigley et al. 2009; Meinshausen et al. 2011). These are full calculations from emissions through concentrations and radiative forcing to global-mean temperature consequences. We do not make use of Global Warming Potentials (as in Howarth et al. 2011, for example), which are a poor substitute for a full calculation

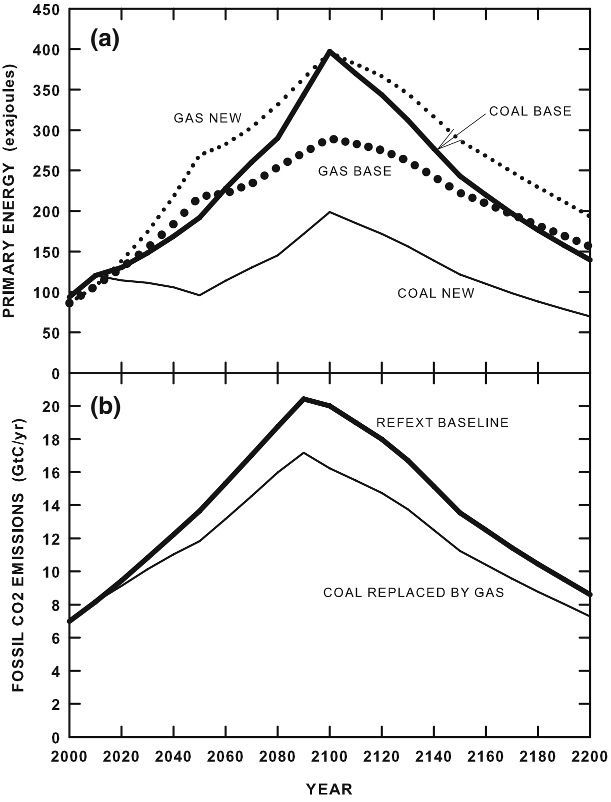

Fig. 1 **a** Primary energy scenarios. Baseline data to 2100 are from the CCSP2.1a MiniCAM Reference scenario. After 2100, baseline primary energy data have been constructed to be consistent with emissions data in the extended MiniCAM Reference scenario (Wigley et al. 2009 — REFEXT). Full lines are for coal, dotted lines are for gas. "NEW" data correspond to the coal-to-gas scenario. Under the final energy constraint that ΔFgas = −ΔFcoal, ΔPgas = −(a/c) ΔPcoal = −0.533 ΔPcoal. **b** Corresponding fossil CO_2 emissions data

(see, e,g., Smith and Wigley 2000a, b). MAGICC considers all important radiative forcing factors, and has a carbon cycle model that includes climate feedbacks on the carbon cycle. Methane lifetime is affected by atmospheric loadings on methane, carbon monoxide, nitrogen oxides (NOx) and volatile organic compounds. The effects of methane on tropospheric ozone and stratospheric water vapor are considered directly. For component forcing values we use central estimates as given in the IPCC Fourth Assessment Report (IPCC 2007, p.4). We also assume a central value for the climate sensitivity of 3°C equilibrium warming for a CO_2 doubling. (A second case using a higher sensitivity is considered in the Electronic Supplementary Material).

Figure 2 shows the relative and total effects of the coal-to-gas transition for a leakage rate of 5%. This is within the estimated leakage rate range (1.7–6.0%; Howarth et al. 2011) for conventional methane production (the effects of well site leakage, liquid uploading and gas processing, and transport, storage and processing). For methane from shale, Howarth et al. estimate an additional leakage of 1.9% (their Table 2) with a range of 0.6–3.2% (their Table 1). The zero to 10.0% leakage rate range considered here spans these estimates — although we note that the high estimates of Howarth et al. have been criticized (Ridley 2011, p. 30).

The top panel of Fig. 2 shows that the effects of CH_4 leakage and reduced aerosol loadings that go with the transition from coal to gas can appreciably offset the effect of reduced CO_2 concentrations, potentially (see Fig. 3) until well into the 22nd century. For the leakage rate ranges considered here, however, the overall effects of the coal to gas transition on global-mean temperature are very small throughout the 21st century, both in absolute and relative terms (see Fig. 2a). This is primarily due to the relatively small reduction in CO_2 emissions that is effected by the transition away from coal (see Fig. 1b). Cases where the CO_2 emissions reductions are larger (due to a more extreme substitution scenario, or a different baseline) are considered in the Electronic Supplementary Material. The relative contributions to temperature change are similar, but the magnitudes of temperature change scale roughly with the overall reduction in CO_2 emissions.

Figure 3 shows the sensitivity of the temperature differential to the assumed leakage rate. The CO_2 and aerosol terms are independent of the assumed leakage rate, so we only show the methane and total-effect results. These results are qualitatively similar to those of Hayhoe et al. who considered only a single leakage rate case (corresponding approximately to our 2.5% leakage case). For leakage rates of more than 2%, the methane leakage contribution is positive (i.e., replacing coal by gas produces higher methane concentrations) — see the "CH4 COMPONENT" curves in Fig. 3. Depending on leakage rate, replacing coal by gas leads, not to cooling, but to additional warming out to between 2,050 and 2,140. Initially, this is due mainly to the influence of SO_2 emissions changes, with the effects of CH_4 leakage becoming more important over time. Even with zero leakage from gas production, however, the cooling that eventually arises from the coal-to-gas transition is only a few tenths of a degC (greater for greater climate sensitivity — see Electronic Supplementary Material). Using climate amelioration as an argument for the

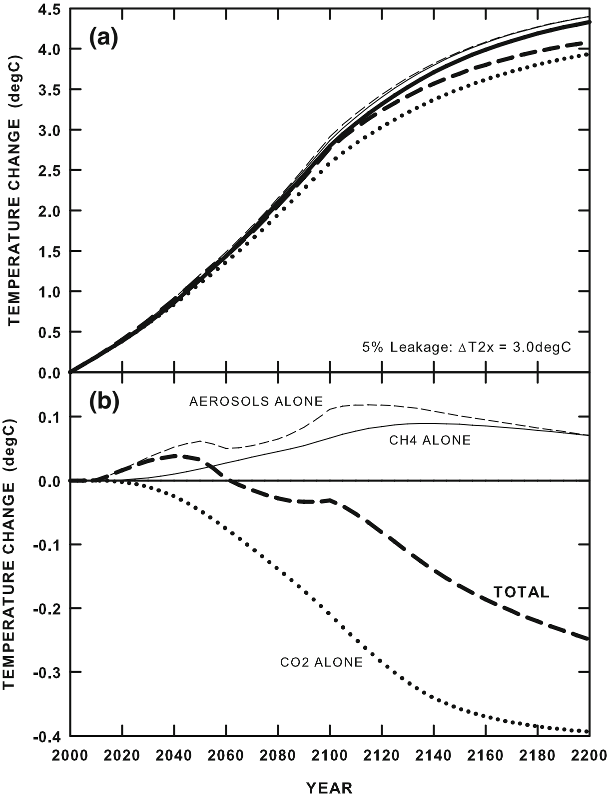

Fig. 2 **a** Baseline global-mean warming (*solid bold line*) from the extended CCSP2.1a Mini-CAM reference scenario together with the individual and total contributions due to reduced CO_2 concentrations, reduced aerosol loadings and increased methane emissions for the case of 5% methane leakage. The *bold dashed line* gives the result for all three components, the *dotted line* shows the effect of CO_2 alone. The *top two thin lines* show the CH_4 and aerosol components. **b** Detail showing differences from the baseline

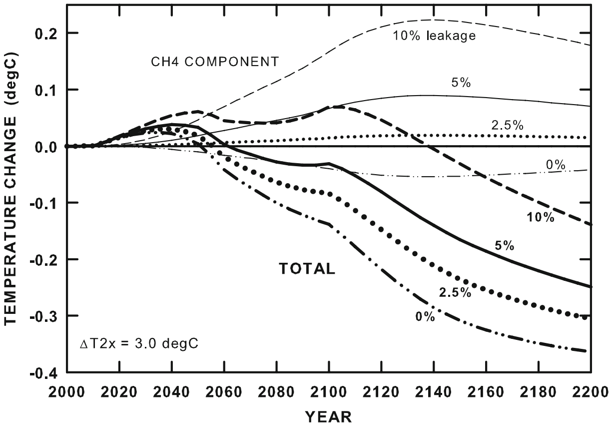

Fig. 3 The effects of different methane leakage rates on global-mean temperature. The *top four curves* (CH4 COMPONENT) show the effects of methane concentration changes, while the *bottom four curves* (TOTAL) show the total effects of methane changes, aerosol changes and CO_2 concentration changes. The latter two effects are independent of the leakage rate, and are shown in Fig. 2. Results here are for a climate sensitivity of 3.0°C

transition is, at best, a very weak argument, as noted by Hayhoe et al. (2002), Howarth et al. (2011) and others.

In summary, our results show that the substitution of gas for coal as an energy source results in increased rather than decreased global warming for many decades — out to the mid 22nd century for the 10% leakage case. This is in accord with Hayhoe et al. (2002) and with the less well established claims of Howarth et al. (2011) who base their analysis on Global Warming Potentials rather than direct modeling of the climate. Our results are critically sensitive to the assumed leakage rate. In our analysis, the warming results from two effects: the reduction in SO_2 emissions that occurs due to reduced coal combustion; and the potentially greater leakage of methane that accompanies new gas production relative to coal. The first effect is in accord with Hayhoe et al. In Hayhoe et al., however, the methane effect is in the opposite direction to our result (albeit very small). This is because our analyses use more recent information on gas leakage from coal mines and gas production, with greater leakage from the latter. The effect of methane leakage from gas production in our analyses is, nevertheless, small and less than implied by Howarth et al.

Our coal-to-gas scenario assumes a linear decrease in coal use from zero in 2010 to 50% reduction in 2050, continuing at 50% after that. Hayhoe et al. consider linear decreases from zero in 2000 to 10, 25 and 50% reductions in 2025. If these authors assumed constant reduction percentages after 2025, then their high scenario is very similar to our scenario.

In our analyses, the temperature differences between the baseline and coal-to-gas scenarios are small (less than 0.1°C) out to at least 2100. The most important result, however, in accord with the above authors, is that, unless leakage rates for new methane can be kept below 2%, substituting gas for coal is not an effective means for reducing the magnitude of future climate change. This is contrary to claims such as that by Ridley (2011) who states (p. 5), with regard to the exploitation of shale gas, that it will "accelerate the decarbonisation of the world economy". The key point here is that it is not decarbonisation *per se* that is the goal, but the attendant reduction of climate change. Indeed, the shorter-term effects are in the opposite direction. Given the small climate differences between the baseline and the coal-to-gas scenarios, decisions regarding further exploitation of gas reserves should be based on resource availability (both gas and water), the economics of extraction, and environmental impacts unrelated to climate change.

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