Groundwater Pollution Prevention in Southeast Minnesota's Karst Region

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Groundwater Pollution Prevention in Southeast Minnesota's Karst Region

Jeffrey St. Ores, E. Calvin Alexander, Jr., and Clifton F. Halsey*

Introduction

Approximately three-fourths of Minnesota's groundwater is contained in aquifers (water-bearing rock formations) underlying southeast Minnesota. Some of these aquifers underlie terrain classified as karst. Other aquifers, because of their cracked and jointed nature, can be considered karst aquifers.

Karst aquifers and aquifers underlying karst features are extremely susceptible to contamination. Reported cases of typhoid fever in Illinois, infectious hepatitis in Michigan, phenol poisoning in Wisconsin, and gastrointestinal illness in Missouri have all been tied to the rapid transmission of the particular disease agents through karst aquifers to the suspected water supplies.

S.P. Kingston, a former Minnesota health official, noted in 1943 that the regional groundwater system in southeast Minnesota is particularly vulnerable to contamination from many sources including surface runoff, domestic sewage, and industrial waste. Kingston, investigating an outbreak of typhoid fever in Fillmore County, concluded that infectious organisms were transmitted from the source of contamination to the wells of the infected individuals via cavernous and fissured underground limestone deposits (karst aquifers).

Many shallow wells in southeast Minnesota contain coliform bacteria and high nitrate levels—both indicators of possible contamination. Some southeast springs also contain these substances as well as traces of pesticides. Even aquifers hundreds of feet deep are considered in danger of contamination.

This publication describes the nature of karst areas and groundwaters, the extreme sensitivity of these groundwaters to many human everyday activities, and procedures which can reduce groundwater pollution potentials.

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Glossary

Agronomic rate: Amount of added nutrients (generally N, P, and K) necessary to sustain a "reasonable" anticipated crop yield. The supplemental source could be manure or inorganic fertilizer.

Aquifer: A geologic formation which yields useful amounts of groundwater. An aquifer must have an appreciable porosity and permeability and must contain drinkable water. In southeast Minnesota the bedrock aquifers are the sandstones and the karst limestones and dolomites. The alluvial sands and gravels may also yield useful amounts of groundwater —particularly in the valleys.

Aquitard: A geologic formation which does not yield useful amounts of groundwater and which retards the movement of groundwater between aquifers above and below it.

Blind valley: A valley which has no surface outlet. Blind valleys terminate in bedrock walls and are formed by disappearing streams.

Blowing well: A well which alternately blows air in and out. The movement of air indicates that the well has intersected a significant air-filled void in the subsurface.

Closed surface depression: A depression in the surface of the land surrounded by a closed contour. In a karst region such depressions often indicate the presence of a buried sinkhole.

Coarse (sandy) soils: Coarse-textured soils have a large proportion of sand-sized mineral particles. The soil is generally characterized by large pore (air) spaces and less total pore space area (relative to loams and clays). Large pores decrease the soil's ability to hold water. Reduced pore area decreases the quantity of water that can be stored at one time. Both characteristics result in rapid downward or lateral movement of water and some contaminants toward fractured limestone bedrock.

1) Coarse sands and gravels are extremely coarse.
2) Medium to fine sands and loamy sands are coarse.
3) Sandy loams and fine sandy loams are medium coarse.

Disappearing streams: A stream which sinks completely underground. The flow may sink at one or more discrete points, stream sinks, and/or it may disappear gradually over a length of the stream bed, a stream sieve. A disappearing stream is a direct connection between the surface and groundwaters.

Karst region: In this publication refers to the area underlain by carbonate bedrock. Includes, but is not limited to, that portion of southeast Minnesota exhibiting terrain classified as karst.

Losing stream: A stream which loses part of its flow into the subsurface. The loss can occur through stream sinks, or stream sieves, or both.

Normal household amounts: Refers to the amount of liquid wastes that can legally be placed in certified sanitary landfills. No absolute values have been established. But, for example, a partially full or full 5-gallon pesticide container is not a normal amount. An empty container of bleach would be a normal amount. Spent motor oil, antifreeze, and similar substances should be recycled rather than placed in landfills.

Permeability: In soil, refers to the ease with which gasses, liquid, or plant roots pass through a bulk mass of soil or a layer of soil (after Brady 1974. *The Nature and Property of Soils*).

Shallow or thin soils: Shallowness is a relative term depending on soil use. Twenty inches or less is generally considered shallow for taxonomic or soil naming purposes. However, the following definitions should be considered for use in karst aquifer protection relative to depth to limestone or water tables.

1) 50 feet or less is shallow if cesspools are being used and impermeable clay or hard bedrock layers do not separate limestone from the bottom of the cesspools.
2) 20 feet or less of coarse- to medium-textured soils is shallow if waste lagoons or holding ponds are used (measured from bottom of structures).
3) 5-10 feet of most soils is shallow if lagoons or holding ponds are used. 5-10 feet of extremely coarse- to coarse-textured soils is shallow when considering manure application, particularly waste irrigation, and manure storage methods other than lagoons or ponds.
4) 3-5 feet of coarse- to medium-coarse-textured soils are shallow when considering any activity.
5) Less than 3 feet of any soil texture is shallow for any potentially polluting activity.

Shallow well: A well which receives water from the near-surface aquifer. The aquifer tapped by each well is determined by the local geology, the depth of the well, and the construction of the well. A properly cased and grouted well only 100 feet deep may act as a deep well and avoid the surface aquifer. Conversely, an improperly constructed well 400 feet deep may be acting as a shallow well if it receives most of its water from the near surface aquifer.

Sinkhole: A closed, usually circular, depression which forms in karst areas. Sinkholes are formed by the removal of material from beneath by underground water flow. Sinkholes are *dug from the bottom* by groundwater. Sinkholes provide a direct conduit connecting surface waters with underground waters.

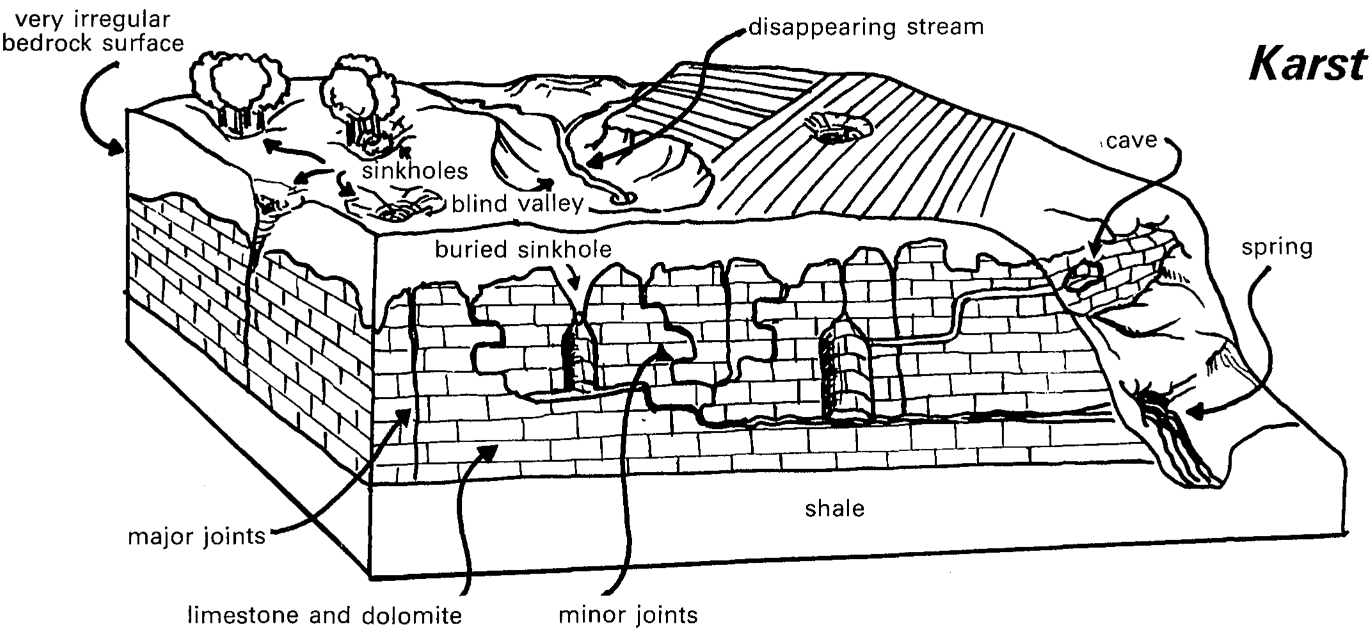
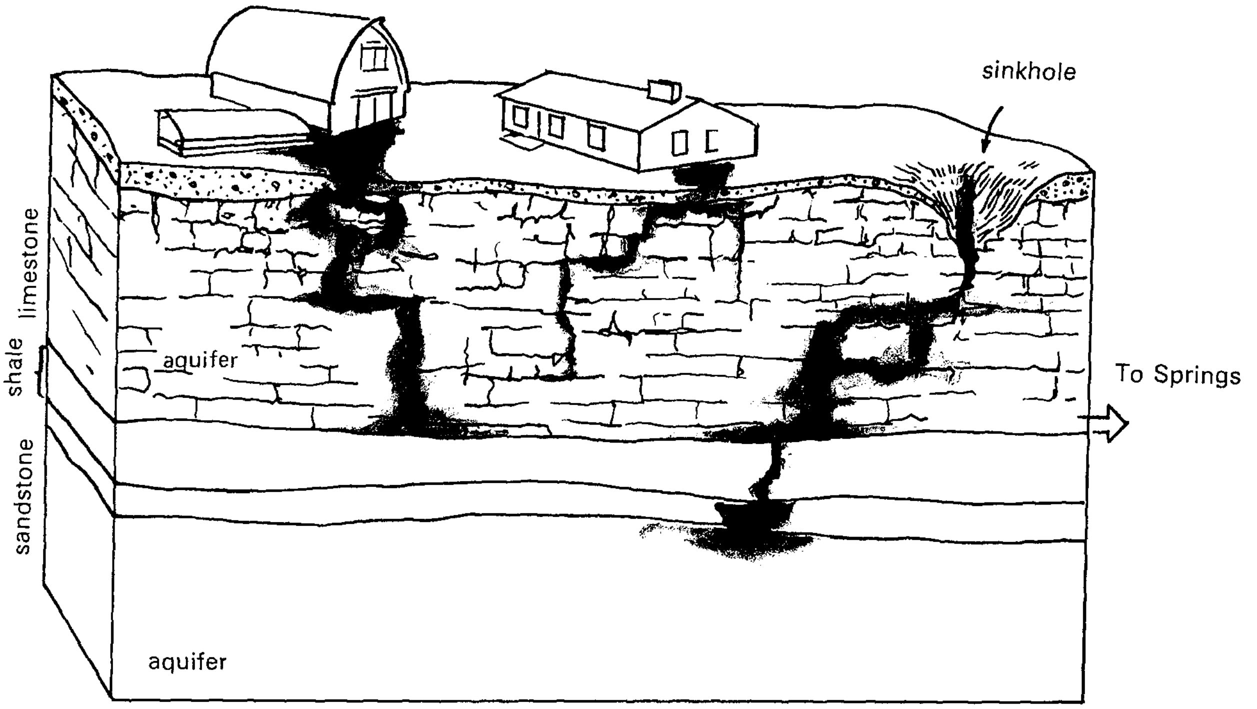


Figure 1. Block diagram showing terrain and subsurface features of karst.

Karst is a geologic term for a land area characterized by streams which disappear underground (disappearing streams) or which lose most of their flow into the ground (losing streams); valleys which have no surface outlet (blind valleys); caves, springs, and circular depressions in the earth referred to as sinkholes (figure 1). Karsts develop in areas where bedrock near the earth's surface is soluble in groundwater. The bedrock, generally limestone (calcium carbonate) or dolomite (calcium and magnesium carbonate), is normally fractured and contains numerous cracks, crevices, channels, and caves.

Karsts typically have very little flowing surface water. Most of the precipitation that starts running across the soil surface quickly disappears into underground drainage. After flowing underground for varying distances, the water will usually return to the surface in the form of springs. Runoff entering the ground via sinkholes, disappearing, and losing streams can become groundwater in hours or just minutes. Contaminants in this runoff, including soil and chemicals attached to soil, will also become part of the groundwater as evidenced by the number of shallow

Figure 2. Contaminant movement through shallow soils, limestone bedrock and sinkholes.



southeast Minnesota wells which yield soil-rich water after heavy rainfalls.

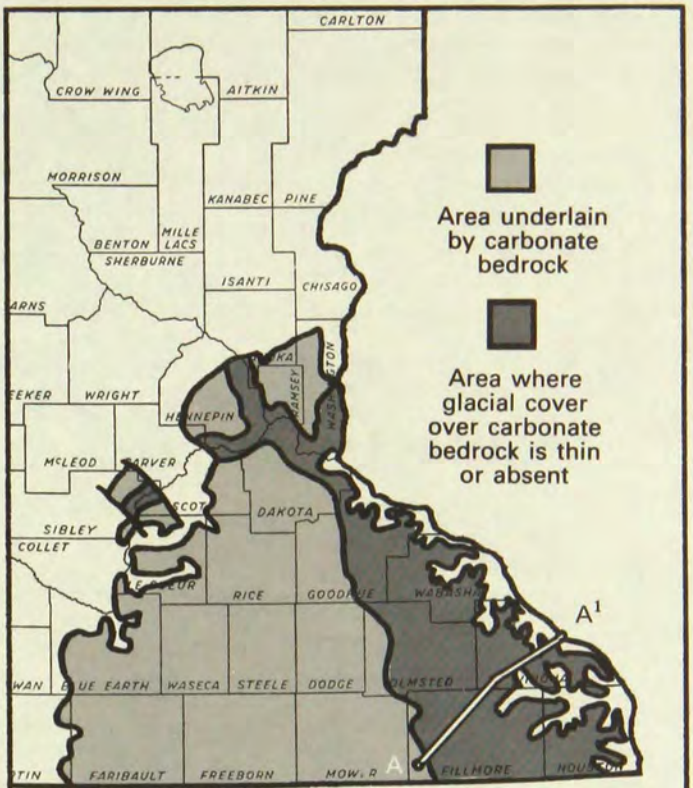
Karst aquifers are fractured and partially dissolved limestone or dolomite bedrock containing quantities of groundwater. Groundwater flowing through the cracks and channels of karst aquifers does not come in contact with as many mineral particles as does groundwater flowing through nonkarst aquifers such as sandstone. So, not only does karst aquifer groundwater flow rapidly (flows have been measured in miles per day versus the inches or feet per year common to sandstones), but contaminants in the groundwater are not readily filtered out. As a result, contaminants can reach domestic wells located miles from the source of contamination.

Karst aquifers can underlie both karst and areas not displaying karst features. Varying thicknesses of soil separate these aquifers from the ground surface. The overlying soil and soil organisms are natural filters of water and contaminants moving down toward the aquifers. But the thinner and coarser the soil, the less the amount of purification. Additionally, sinkholes and disappearing streams can bypass this natural purification process by creating direct links between the ground surface and aquifers. Consequently, karst aquifers underlying only a few feet of soil or aquifers underlying karst are easily contaminated (figure 2).

Figure 3 shows the areas in Minnesota underlain by limestones and dolomites (karst aquifers). A series of these aquifers as well as sands and muds were deposited on top of one another millions of years ago as a sequence of oceans advanced and retreated across southeast Minnesota. The sands became sandstone aquifers and the muds became shales, which now function as aquitards or confining bedrock layers which restrict water movement and partially protect underlying aquifers from contamination. The karst and sandstone aquifers and shale aquitards are not level but rise gently in several directions, including toward the Mississippi River. Figure 4 illustrates the series of aquifers and aquitards present in an area extending from Mower County northeast toward the Mississippi River. Note the rise of the formations and the division of the aquifers into upper, middle, and lower aquifers.

A few million years ago, giant ice sheets began to advance and retreat across part of southeast Minnesota. These glaciers left thick deposits of clay, sand and gravels covering the sandstones, shales, limestones, and dolomites. But the latest group of glaciers did not cover extreme southeast Minnesota (the figure 3 area indicated as glacial cover thin or absent). The absence of the glacial deposits in this area and centuries of erosion have resulted in a thin protective cover overlying aquifers. Additionally, the rising upper aquifers and aquitard have been completely worn away in many portions of the Mississippi River border counties (note the right side of figure 4). Karst has developed in areas (for example, Fillmore and Olmsted Counties) having deep river valleys and a relatively thin, but still present, soil layer covering *upper* aquifers.

Figure 3. Karst region.



A-A[1] is the line of cross section shown in figure 4

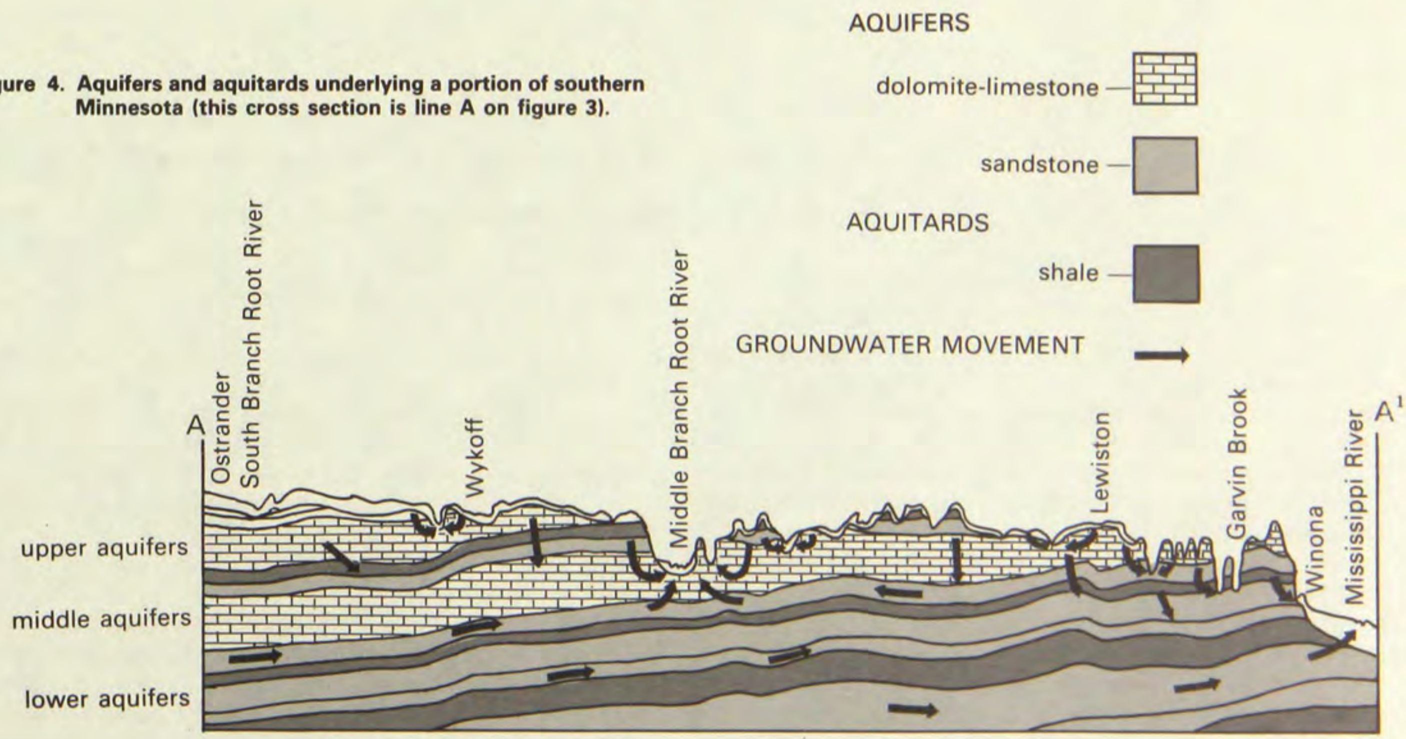
All of the area indicated in figure 3 as underlain by carbonate bedrock is sensitive to groundwater contamination. Sensitivity is lowest where both the protective glacial deposits and upper aquitard are present (the light shaded area in figure 3). Yet, scattered spots of high sensitivity occur in this western half. Pockets of shallow soils exist, and activities such as home site development and quarrying can strip away soil and decrease the distance between aquifers and the soil surface.

The eastern and northern portions of southeast Minnesota (dark shading, figure 3) are very susceptible to groundwater contamination. This high susceptibility is due in part to the occasional occurrence of karst terrain but is primarily due to the more frequent occurrence of shallow soils overlying karst aquifers. Shallow soils in parts of the Mississippi River border counties are particularly critical because they overlie *middle* karst aquifers (as noted, the upper aquifers and more important, the protective upper aquitard have disappeared).

In summary, the entire area underlain by carbonate bedrock is sensitive to groundwater pollution. But this sensitivity varies. Each piece of land (for example 40 acre segment) and underlying soil and rock formations should be examined, both to detect the presence of groundwater contamination and to determine the potential to contaminate groundwater at that particular spot. This publication cannot provide information based on such an intensive evaluation program.

However, table 1 summarizes southeast Minnesota features which indicate susceptibility to groundwater contamination. Table 2 lists human activities which

Figure 4. Aquifers and aquitards underlying a portion of southern Minnesota (this cross section is line A on figure 3).



(Source: Adapted from Hydrologic Investigations Atlas HA-548, 1975) can contribute to contamination. Use of the two tables can help in the initial evaluation of a rural area. The presence of any listed feature or activity indicates potential for pollution to take place at that particular spot and implies need for a closer look. However, only periodic well water sampling will determine the actual presence of groundwater contamination because activities occurring miles away can affect the quality of water in many wells.

Table 1. Karst features indicating high groundwater pollution potential

Indicators of direct connections from the soil surface to groundwaters.
- Sinkholes.
- Disappearing or losing streams.
- Blind valleys (see glossary).
- Closed surface depressions (see glossary).
- "Blowing" wells and wells that turn murky after storms.

Indicators of minimal separation between limestone or dolomite bedrock and the soil surface.
- Outcrops of bedrock.
- Shallow soils above bedrock (see glossary).
- Lack of surface drainage.

Table 2. Activities or structures that can contribute to groundwater pollution

1. Disposing of *any material* in sinkholes, streams, or drainageways leading to these features.
2. Cesspools.
3. Drywells (seepage pits) less than 50 feet above limestone bedrock or groundwater.
4. Drainfields with bottoms less than three feet above limestone bedrock.
5. Malfunctioning and poorly maintained septic tanks and drainfields.
6. Bypassing malfunctioning septic systems by pumping wastes into the nearest ravine, sinkhole, stream, or field.
7. Disposing of materials accumulating in septic tanks (septage) other than as called for by MPCA guidelines.
8. Improperly constructed and grouted active water wells.
9. Uncapped and unsealed abandoned water wells.
10. Pasturing animals in or near disappearing streams and sinkholes.
11. Manure storage areas and outdoor animal confinement areas not having a good soil surface seal or situated such that runoff carries pollutants from these areas to wells, sinkholes, streams or drainageways leading to wells, sinkholes, or streams.
12. Applying more manure and fertilizer than soils and crops can retain or use.
13. Applying manure and fertilizers at high runoff times to areas draining to sinkholes and disappearing streams.
14. Disposing of normal household amounts of flammable, toxic, and explosive "household" wastes in other than a certified sanitary landfill, recycling facility or waste recovery plant.
15. Runoff and erosion on crop and pastureland.
16. Disposal of full or partially full pesticide containers or contents of the containers in any area including landfills which has not been designed to contain or treat such chemicals.
17. Formulating pesticides and/or washing application equipment within 200 feet of wells, sinkholes and streams or drainageways leading to these features.
18. Failure to triple rinse "empty" pesticide containers followed by disposal of containers other than at certified sanitary landfills, drum reconditioners or recycling facilities.
19. Lack of anti-siphoning devices on pesticide applicator filling equipment.
20. Leaking above or below ground fuel, manure, silage or other storage facilities.
21. Others (see text).

Polluting Activities and Practices Which Reduce Groundwater Pollution Potential

Almost any human activity can result in groundwater contamination if the nature of karst and karst aquifers is not realized. Activities include those conducted by urbanites, suburbanites, units of government, and commerce and industry. However, this publication addresses activities associated primarily with rural residences and farms (table 2).

There are many well-known practices which can be used to minimize groundwater pollution potential in rural areas. These practices are discussed in the following pages. However, all the practices do not apply to every southeast Minnesota acre. Consultations with experts (see listing at the end of this publication) will help determine if and what practices are necessary in a particular area.

SINKHOLES

Sinkholes must not be used as disposal sites because sinkholes are direct conduits to groundwater. Placing anything in sinkholes or runoff entering sinkholes is almost like putting that material into wells. Unfortunately, garbage, herbicide cans, old railroad ties, debris from burned buildings, and other materials have been observed in sinkholes in southeast Minnesota. Feedlots draining to sinkholes have also been noted.

Attempts to eliminate sinkholes by plugging with sand and other fill materials can prove ineffective. Subsurface water and soil processes responsible for sinkhole formation may be accelerated by improper filling procedures. Contact university geologists trained in karst phenomenon and United States Department of Agriculture-Soil Conservation Service (USDA-SCS) staff for help in determining if sinkhole plugging will work.

Diverting potentially polluted runoff. Keeping runoff away from sinkholes is a pollution control practice, provided the diverted water does not trigger new sinkhole formation. Again, it is important for geologists and SCS or local Soil and Water Conservation District (SWCD) staff to help determine the feasibility of diversion.

Fencing around sinkholes. This practice protects animals from possible injury; discourages dumping of materials into holes; and may result in natural vegetation growing up around sinkholes.

Growing natural vegetation around a sinkhole. Natural vegetation creates a buffer zone which filters pollutants out of runoff. Guidelines for buffer zones have not been developed, but new guidelines applicable to feedlots may prove worthwhile. Alternatively, research indicates that forest or grass buffer strips from 50-100 feet wide greatly reduce nitrogen concentrations in runoff. Widths down to 13 feet have also proved effective. Perhaps 25 feet should be a minimum width around sinkholes.

HOME SEWAGE TREATMENT

Based on rural population, there could be at least 15,000 home sewage treatment systems just in Fillmore, Houston, Wabasha, and Winona Counties. A number of these systems were likely installed without knowledge of karst, and do not use sufficient soil for adequate treatment. Such systems may be a major source of groundwater contamination.

Agricultural Extension Service publications (see page 18) discuss in detail, system evaluation, design, and maintenance. The publications and local extension agents, SCS staff, zoning administrators, and regional Minnesota Pollution Agency (MPCA) staff should be consulted for specific information.

Systems. Common sewage systems are septic tanks and drywells, septic tanks and drainfield trenches or beds, and cesspools.

- *Cesspools* can no longer be legally installed. Raw sewage is discharged into a leaky tank. The soil around the cesspool eventually seals and the sewage surfaces, constituting a health hazard. Or the cesspool is in contact with fractured bedrock and the sewage discharges without treatment.

- *Drywells* (alternately called leaching pits or seepage pits and incorrectly called cesspools) are small confined areas receiving wastes from septic tanks. Dry wells can be a poor choice in karst areas because sewage from drywells encountering fractured bedrock can move directly into channels leading to groundwater. Individual Sewage Treatment System Standards (WPC-40) of the MPCA states that seepage pits shall not be installed "in areas where limestone or any geological formation characterized by similar fault patterns is covered by less than 50 feet of earth."

Additionally, drywells should not be installed in the following instances: where domestic water wells shallower than 50 feet are used; in soils having a percolation rate slower than 30 minutes per inch or where the percolation rate of any soil layer contacting the drywell side or bottom is faster than 0.1 minutes per inch; or when barrier rock such as clay and nonfractured bedrock or the known level of the groundwater table would be less than 3 feet below the drywell bottom.

- *Soil absorption fields* such as drainfield trenches or beds are subsurface systems which receive effluent from septic tanks. Drainfield trenches are 18-36-inch-wide excavations on the contour into which trench rock (¾-2½ inches) and a 4-inch distribution pipe are placed. The trench rock is backfilled with the removed topsoil. A slime layer of organisms, called an organic mat, forms at the contact point between the trench rock and the underlying soil. Both the organic mat and the soil treat the effluent. But at least 3 feet of aerated soil below the trench bottom is necessary for adequate treatment. Less than 3 feet of suitable soil between the trench and underlying fractured bedrock or sandstone can result in inadequate removal of pathogens (disease causing agents) from sewage and subsequent movement of those pathogens into the groundwater. Soils having percolation rates between 0.1 and 60 minutes per inch are generally considered suitable for efficient operation of a soil absorption field.

- *Mound systems* are options for use in shallow soil areas. Effluent from a septic tank is directed to a seepage bed elevated above the original ground surface by carefully selected fill materials which maintain acceptable separation distances between the bed and shallow fractured bedrock. NCR Bulletin 130 discusses mound systems, as well as other alternative systems to use in problem soil areas.

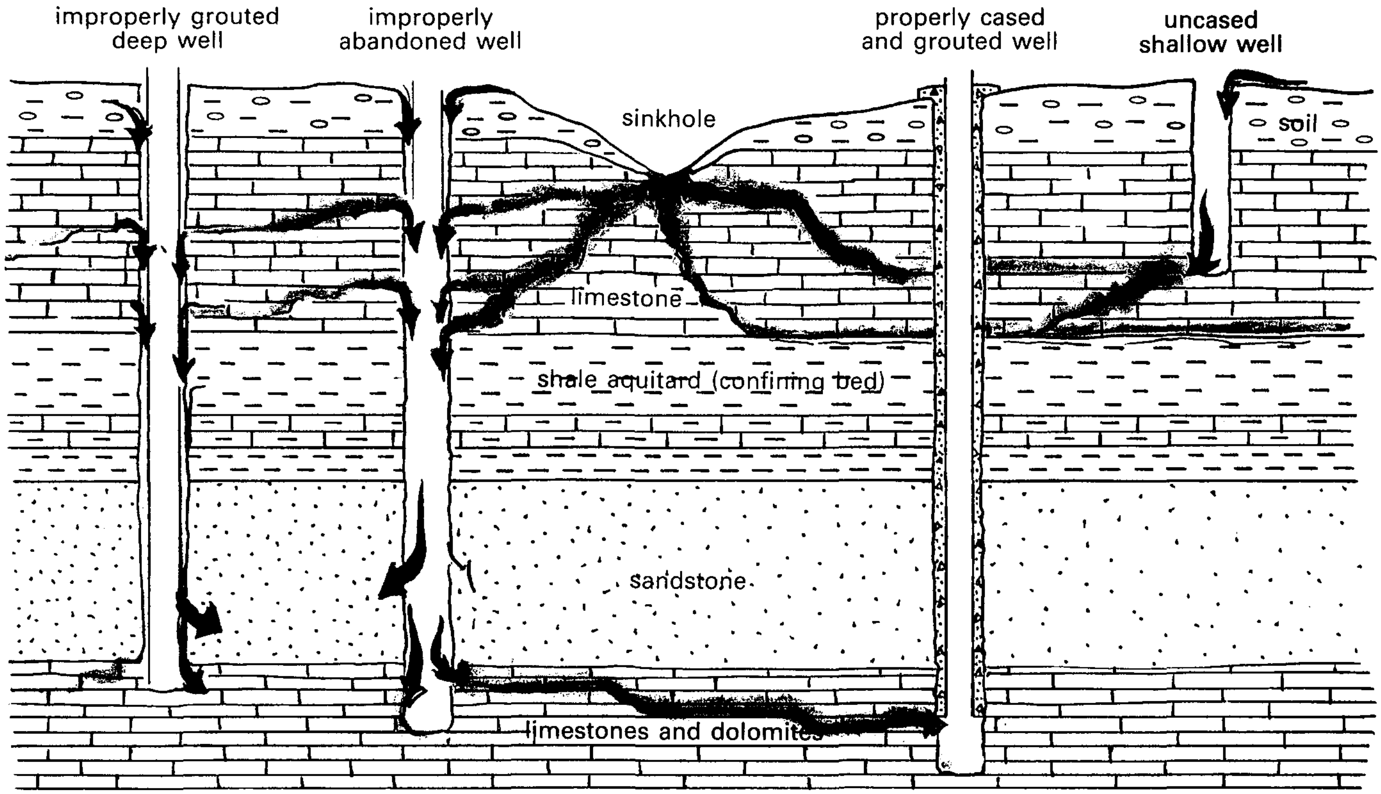
System use and maintenance. Garbage such as coffee grounds, cooking fats, disposable diapers, wet-strength paper towels, rags, and other materials which disintegrate slowly should not be put in sewage systems. These materials will rapidly fill septic tanks and if not removed periodically will flow to and clog drywells or soil absorption fields. Materials from sink garbage disposals can also clog a treatment system.

Septic tanks must be maintained and periodically cleaned out (preferably by professionals). Failure to remove accumulated materials (septage) from septic tanks can clog the system's soil absorption area. Waste may then be discharged to the ground surface and run into a stream or sinkhole if the system fails because of clogging.

Disposal of septic tank septage. Septage removed from septic tanks should be treated as a fertilizer and disposed of according to MPCA guidelines for septage disposal. Never discharge septage into quarries, ravines, sinkholes, and other karst features.

Malfunctioning systems. Have a malfunctioning treatment system immediately repaired. Running a pipe to the nearest field, ditch, or other area is not a solution to a plugged system.

Figure 5. Contamination of wells through improperly constructed or abandoned wells.



(Adapted from: Problems relating to safe water supply in southeastern Minnesota. Report to the Legislative Commission on Minnesota Resources from the Minnesota Department of Health)

WELLS

Proper well construction and abandonment procedures are essential in southeast Minnesota. Minnesota's Water Well Construction Code (7MCAR), instituted in the mid 1970s, addresses all aspects of proper well construction, maintenance, and abandonment. It further requires that wells be constructed only by drillers licensed by the Minnesota Department of Health.

Construction. Improperly constructed wells are a major pathway of pollutant movement to groundwater. Well boreholes are generally larger than well casings. A conduit is created linking the soil surface or upper soil formations to lower aquifers if the space between the wellhole walls and casings is not sealed or grouted properly (figure 5). Additionally, deteriorating and leaking casings allow materials to enter and move down the well itself. Contaminated runoff or contaminants in the upper soil layers can and will move toward wells and down the outside or inside of the well casing under the conditions just discussed.

There are, based on Minnesota Department of Health estimates, at least 14,000 active water wells in that portion of southeast Minnesota indicated on figure 3 dark shading. Estimates of the number of active wells, which need improvement or redrilling, range as high as 10,000 in the four county area of Fillmore, Houston, Wabasha, and Winona. These wells may have been drilled into shallow polluted aquifers, improperly grouted or sealed, or constructed with poor quality casing. Wells constructed prior to passage of the Water Well Construction Code are most suspect. Many existing wells should be evaluated for adequacy and, if necessary, repaired or replaced.

New well construction must comply with code requirements. Among other things, wells drilled through a number of aquifers must be sealed off from any contaminated aquifers encountered. Any spacing between boreholes and casing or between various casings must be adequately grouted and sealed. No openings should exist linking the ground surface to aquifers other than that through which water is produced. Only approved casing material should be used. Well tops should generally extend above ground and the site should be graded to divert runoff away from the well top.

Well location. Runoff, depending on site conditions, can drain toward well tops. Shallow subsurface water can also move toward wells. For these reasons, wells should be located away from potential contamination sources. At a minimum, insure that wells are located at least:

- 150 feet from a chemical preparation or storage area
- 100 feet or greater (depending on conditions) from below grade manure storage areas if these areas are in compliance with MPCA regulations
- 75 feet from cesspools, leaching pits, and drywells
- 50 feet from septic tanks, subsurface sewage disposal fields, graves, livestock yards and buildings, and manure storage piles

Wells with casings less than 50 feet deep and not encountering at least 10 feet of impervious soil should be located at least 150 feet from cesspools, leaching pits, or dry wells and at least 100 feet from a subsurface disposal field or manure storage pile.

Abandoned wells. Abandoned wells are another major source of concern. Any abandoned well which has not been filled, sealed, and covered properly is a potential pathway for pollutant movement to groundwater. Contaminants can move directly down the well itself.

Estimates of the number of abandoned wells in southeast Minnesota range as high as 9,000. Many of these have not been filled and sealed properly. The seriousness of the problem cannot be overstated. Persons knowing locations of abandoned wells should contact district or state health officials. An accurate count of abandoned wells will help officials assess the magnitude of the problem and develop programs to correct it.

Wells to be abandoned. Wells when being abandoned must be abandoned in accordance with the state code. This means doing the following:

- notifying health officials of abandonment procedures
- disconnecting the well from the system
- plugging the well hole according to the code procedures
- permanently sealing the top of the well according to code procedures

Well water testing. Have well water periodically tested for contaminants and record the results. Groundwater pollution trends may be detected before the water becomes undrinkable. Contact county community health service for well sampling instructions.

LIVESTOCK PRODUCTION

Local SWCDs and SCS technicians, and geologists, extension agents, and MPCA staff should be contacted for help in evaluating pollution potential of livestock production activities and selecting pollution control practices (including manure disposal plans).

Unconfined livestock. Animals are allowed free access to or may be pastured near disappearing and losing streams and sinkholes (figure 6). Wastes from these animals can move into the groundwater system. Practices to keep livestock away from streams and sinkholes should be used and include: well-located livestock watering facilities, vegetated buffer strips, and fencing.

Confined livestock. This section pertains to areas where animals are concentrated, including housed or partially housed animals and outdoor confinement areas such as beef feedlots, outdoor dairy feeding operations, and sow feeding pens.

Southeast Minnesota has a high density of animals (number of animals per square mile), a relatively large number of feedlots, and relatively great potential for runoff. Runoff can carry contaminants from feedlots and manure storage areas to sinkholes, disappearing and losing streams, and wells. High pollution potential exists when livestock are confined near these karst features and wells, and precautions have not been taken to prevent contaminants from entering the features. It is estimated that there are 480 total feedlots discharging wastes to streams and lakes in just Goodhue, Wabasha, Winona, Olmsted, and Houston Counties.

Feedlots and manure storage areas located on shallow sandy soils overlying fractured limestone can also pollute if the lot or storage area floors have not been sealed. Contaminants can move downward in the soil profile toward groundwater.

There are a number of practices which can reduce pollution potentials associated with confined animals.

- Runoff originating outside the lot can be diverted away from the lot or manure storage area.
- Down spouts and gutters on farm buildings can reduce the amount of runoff flowing across the lot.

Figure 6. Livestock pastured near a stream.

- Lot or manure storage area floors can be sealed. Paving may be necessary when limestone is only a few feet deep. Animal traffic can compact unpaved lot floors. This compaction reduces movement of water and contaminants into the soil and downward toward groundwater. Remove manure carefully from compacted unpaved lot floors. Avoid disturbing the lower 3-4 inch mixture of compacted soil and manure during manure scraping operations.
- Manure can be stored in storage tanks or above ground silos. These facilities when made of concrete or steel provide good assurance against leaching or runoff.
- Locate lots away from sinkholes, streams, and shallow sandy soils.
- Wastes from lots or animal housing can be collected, stored, and sometimes treated with holding ponds, settling basins, lagoons, and oxidation ditches. These structures should have sealed bottoms (either naturally or artificially sealed) particularly in areas where limestone is only a few feet deep. One group of scientists, however, (see Extension Handbook MWPS-18) suggests avoiding the use of lagoons when the lagoon bottom would be less than 20 feet above limestone (depending on soil type).

Proper land application of manure. This practice is as important as proper storage. Nitrates from manure can move downward (leach) toward fractured limestone if plants haven't used the nitrates and water is moving down in the soil. Disease bearing (pathogenic) organisms, if present in manure, can leach toward fractured limestone if the organisms are alive, soils are relatively sandy, and water is moving downward.

Additionally, nitrates and pathogens can move to sinkholes, wells, and disappearing streams by runoff and soil loss. Nitrate movement occurs primarily when manure is applied on actively melting snow or thawing ground or irrigated at a rate which causes runoff. Pathogen movement occurs when soil loss and runoff occur, provided the organisms are present and alive.

The potential for groundwater contamination from land-applied manure is real in karst areas. But this potential can be minimized by developing and following a sound manure disposal plan. Such a plan should recommend methods, timing, and amount of manure applications for individual fields based on characteristics of those fields.

The following recommendations should be considered when developing a manure disposal plan. The first three apply to all areas in southeast Minnesota and if followed, will greatly reduce pollution potential. The last six apply to especially critical areas which occur in some fields or portions of fields.

1. Apply at rates no greater than necessary to satisfy plant phosphorus (P) or potassium (K) or nitrogen (N) needs in a single year (agronomic rates). But do not exceed the agronomic rate for N. First, the N, P, and K nutrient need for the crop to be grown should be determined by the use of soil tests, with credit given for contributions from preceding legumes and past manure application. Then the amount of manure, and perhaps supplementary fertilizer, to meet this nutrient need, can be calculated based on the available nutrient content of manure after it has undergone collection, storage, and any treatment operations occurring on the farm. Periodic manure testing will help determine manure nutrient content. Publication MWPS-18 can also be consulted to obtain average nutrient values of manure.

Sometimes, areas may exist on the farm where manure can be applied at greater than agronomic N rates without the potential for excessive leaching or runoff to occur. But on-farm investigation will be necessary to locate such areas.

2. Incorporate manure soon after application (when soil depth, crop life stage, and tillage technique permit).

3. When irrigating animal wastes, apply light applications which do not exceed the soil's capabilities to retain the liquid (depth to limestone bedrock or local water tables and soil water holding capacities, percolation rates, and moisture content must be considered).

4. Limit or avoid applications including irrigated applications within 200 feet of wells, disappearing streams, and sinkholes (100 feet from sinkholes for non-irrigated wastes). Increase this distance to 300 feet (200 feet from sinkholes for non-irrigated wastes) on slopes greater than 6 percent.

5. Avoid applications on saturated soils, actively melting snow, or thawing ground on fields upslope from sinkholes, streams, and drainageways.

6. Limit or avoid applications on alfalfa fields or pastureland draining to sinkholes and disappearing streams.

7. Avoid applications on coarse sands and gravels which do not have fine clays or impermeable rocks underlying and separating the sands from limestone or local water tables.

8. Avoid applications on coarse to fine sands and loamy sands when depth to bedrock is less than 10 feet and on sandy loams when less than 5 feet. If applications are necessary, space them out throughout the year (when workload and crop life stage permit) and reduce rates below estimated crop nitrogen needs (supplement with fertilizer).

9. Limit or avoid applications on fields or portions of fields where limestone is less than 2 feet deep (refers to limestone bedrock rather than to soil containing scattered pieces of limestone). Delay incorporation as long as possible if applications are necessary. Avoid injecting manure directly into limestone.

Special recommendations may be necessary when an entire farm is a critical area (for example, all fields contain numerous sinkholes). Such recommendations can only be made with on-farm inspections, but for example could include suggestions to apply manure on fields sloping to sinkholes if the applications occurred when chances of runoff were low; or to store and treat manure prior to application.

Milkhouse and milking parlor wastes. A considerable quantity of wastes can be generated from milkhouses or milking parlors. The quantity depends on the operation, but for example, a 100-unit cow operation with automatic washing equipment can use over 800 gallons of water per day for washing operations. Wastes can include feed, bedding, hoof dirt, medicines, residual cleaning chemicals, milk, and milk solids such as fat, albumin, and lactose.

Proper disposal of these wastes is essential and is discussed in Agricultural Engineering M-sheet 159. Portions of the following text are adapted from that sheet.

Milkhouse or milking parlor wastes should be discharged to a settling tank and from there be land-applied or stored in a lagoon and land-applied later (however the cautions discussed earlier regarding lagoon use should be noted). The settling tank must be frequently cleaned out to remove manure, feed, bedding, soil, and other solids.

Subsurface treatment of milkhouse or parlor wastes has generally proved unsuccessful. Milk solids do not settle out or decompose in a septic tank but rather flow to the drainfield trench or drywell and plug the system.

Large barns have rest rooms for human waste. These human wastes must be treated separately from parlor or milkhouse wastes by using the home sewage treatment systems discussed earlier in this publication.

Figure 7. Refuse-filled sinkhole in southeast Minnesota.
(Photo courtesy of the *Journal of Freshwater*, Navarre, Minnesota)

Land spreading of milkhouse or milking parlor wastes should be done in accordance with MPCA guidelines on septage disposal or the wastes should be treated as manure and disposed of as discussed previously.

Dead Animals. Leaving dead animals on the soil surface or disposing of them in the nearest ravine, gully, sinkhole, or quarry can be hazardous. The Minnesota Board of Animal Health requires that carcasses be burned, buried, or rendered. Rendering is preferable in karst areas.

HOUSEHOLD WASTES

The average household generates considerable quantities of waste in a year. Wastes include relatively harmless and solid materials, such as paper, wood, metal cans, and food debris; and more hazardous, generally liquid materials, such as solvents, adhesives, cleansers, lighter fluids, spent oil, paint thinners, and antifreeze.

Improper disposal of household wastes will pollute groundwater and is occurring in southeast Minnesota. Sinkholes, quarries, ravines, and dumps which cannot adequately contain wastes are being used as disposal sites (figure 7). This improper disposal need not occur because a number of good waste management practices exist.

Resource recovery. This is of major importance at the household level. Pollution is eliminated; landfills do not rapidly fill and nutrients, minerals, and other resources are conserved.

- *Composting,* discussed in Agricultural Extension Service Soils Fact Sheet 12, decomposes vegetable and other organic portions of garbage. Construction and use of compost heaps recovers nutrients, requires limited effort, and should be practiced.
- *Recycling* solvents, waste oils, glass, aluminum, and newspaper is equally important. A list of recycling facilities in southeast Minnesota is presented in this publication.

Waste reduction. Avoid disposable items when reusable ones are available. Prolong the life expectancy of materials.

Waste recovery and treatment plants. These plants replace or supplement landfills. Resources are recovered or treated rather than disposed of untreated. These facilities require commitment by local government and residents.

Certified sanitary landfills. Refuse which has not been recovered can be disposed of in these containment areas. Landfills are designed to hold solid and nonhazardous wastes. But normal household amounts (see glossary) of hazardous wastes are generally allowed in landfills. Only *certified* landfills have been found suitable for waste containment. The amount of wastes placed in them should be minimized by exercising options previously discussed.

Home disposal sites. Such sites are a final but least preferable waste management technique. Nonhazardous materials, which for some reason have not been recycled or recovered, can be disposed of on the homestead. The site must be kept sanitary, and filled, and covered. At least 5 feet of slowly permeable soil should separate the bottom of the site from water tables or limestone. Ravines, gullies, quarries, sinkholes, and similar features are *not* suitable. Hazardous materials such as empty pesticide containers should not be placed in homestead sites.

TILLAGE, EROSION, AND RUNOFF

Cropland and pastureland erosion rates are usually higher in the southeast than elsewhere in Minnesota. Runoff values are among the highest and the ability of runoff and soil particles to move off the field, is as great, if not greater than, anywhere else in the state.

High runoff and erosion rates are a problem in areas of sinkholes, disappearing, and losing streams. Contaminants contained in runoff move rapidly to these features and from there to groundwater. Erosion in areas where limestone bedrock is shallow is also critical because the protective soil covering the bedrock is lost.

The primary reason for excessive cropland soil loss is fall turnplow (moldboard) tillage followed by repeated secondary tillage. Approximately 70 percent of southeast Minnesota cropland is farmed this way. Erosion on sloping pastureland is caused primarily by overgrazing, poor maintenance of vegetation, and occasionally by failing to exclude livestock from critical areas.

SWCD, USDA-SCS, and local Agricultural Extension Service personnel should be contacted for help in determining the need for and installing of erosion and runoff control practices.

Tillage. Conservation tillage is of prime importance in southeast Minnesota. Any tillage system which limits the amount of soil turned over (inverted) and leaves enough crop residues remaining after planting to cover 25 percent of the soil surface is defined as conservation tillage. The term "system" is stressed because the type of tillage can vary over time depending on past, current, and projected future crops. Specifically, different types of conservation tillage can be used or rotated, depending on the crop rotation.

Agricultural Extension Bulletin 479 deals with soil conditions and crop rotations best suited to the various types of conservation tillage. Till-planting on ridges is one conservation tillage system adaptable to a number of crops and soil conditions. No-till is adaptable to only select conditions. Additionally, no-till's effects on runoff and deep leaching of nitrates have not been clearly defined. The use of no-till must, therefore, be carefully evaluated.

Use of Bulletin 479 and consultation with local experts will aid in the selection of a conservation tillage system resulting in crop yields or net incomes comparable to those from moldboard plowing.

Other cropland erosion and runoff controls. These include contouring, strip-cropping, diversions, terraces, grassed waterways and rotations (row crop, small grain, and meadow). *Diversion and terrace construction and use should not leave limestone bedrock exposed and the amount of runoff trapped or diverted should not trigger sinkhole formation or allow direct entry of nitrate rich water into limestone.*

Waterways, diversions, and terraces should not drain into disappearing or losing streams or sinkholes.

Pastures. These should be kept properly stocked and well vegetated. Local USDA-SCS and SWCD staff should be contacted to determine if livestock exclusion from critical, erodible slopes will also be necessary.

PESTICIDES

Field applications and handling of these chemicals can contaminate karst aquifers. Extension Bulletin 428 discusses all aspects of pesticide use. Agricultural Chemicals Fact Sheet 17 discusses in detail pesticide container disposal.

Field applications. Practices which encourage runoff and erosion are primarily responsible for movement of applied chemicals toward sinkholes and disappearing streams. But sprayed liquids and applied dusts can drift under favorable conditions (for example, when temperatures are high or air is gusty and turbulent, such as between 2 and 4 p.m.). Applying in close proximity to karst features increases the likelihood of spray drift or chemical enriched soil and water entering these features.

A number of practices can reduce chances of pesticides entering groundwater.

- *Estimating chemical needs.* Proper identification of pests and an understanding of crop and pest life stages are important. Misnaming a pest and applying the wrong chemical or applying the right chemical before it is needed can result in poor control and a need for additional applications. The Agricultural Extension Service has several publications on pest identification. Pest scouting programs are also being developed which help in pest identification and selection of control practices.

- *Even applications.* Sprayer equipment should be well-maintained and cleaned to prevent leakage as well as uneven applications. Sprayers should be properly calibrated to insure application of the right amount of pesticide in the right area. Extension Bulletin 428 or Agricultural Chemicals Fact Sheet 5 describes calibration procedures. Procedures or tables may also have

Table 3. Relative mobility of pesticides in soils (adapted from Helling et al. 1971. Advan. in Agon. 23: 147-240)

Mobility Class*				
5	4	3	2	1
Dalapon**	Picloram (Tordon 22K)	Propachlor	Bensulide (Betasan)	Chloroxuron
(Dowpon, Basfapon)	MCPA	(Bexton, Ramrod)	Prometryne (Prefas)	(Norex, Tenoran)
Dicamba (Banex, Banvel)	Amitrole (Weedazol)	Prometone (Pramitol)	Diuron (Karmex, Dynex)	DCPA (Dacthal, Fatal)
Chloramben	2,4-D	Naptalam (Alanap)	Linuron (Lorox, Afalon)	*Lindane*
(Amiben, Vegeben)		2,4,5-T	EPTC (Eptam, Ordram)	*Phorate* (Thimet, Rampart)
		Propham (Chem-Hoe, IFC)	Vernolate (Vernam)	*Parathion*
		Diphenamid (Dynid, Enide)	Chlorpropham (Furloe, CIPC)	*Disulfoton* (Dimaz)
		Atrazine (AAtrex)	*Azinphosmethyl* (Carfene)	Diquat (Ortho-Diquat)
		Simazine (Princep, Aquazine)	*Diazinon* (Basudin, Diazitol)	*Zineb*
		Alachlor (Lasso)		*Chloroneb*
		Ametryne (Evic)		(Demosan, Tersan-SP)
				Trifluralin (Treflan)
				Benefin (Balan, Balfin)
				Toxaphene
				(Motox, Toxakil)

*Class 5 compounds (very mobile) to Class 1 compounds (immobile) are in the scheme of Helling and Turner (1968). Within each class, pesticides are ranked in estimated decreasing order of mobility.

**Names of herbicides are set in roman type; insecticides, fungicides, and acaricides are in *italics*.

been included with the equipment or may be available from a pesticide dealer.

- *Use of mobile pesticides.* This should be minimized in areas of shallow soils over bedrock. Table 3 gives the relative downward mobility of some pesticides.
- *Rotate pesticides.* This reduces pests' ability to develop resistance to pesticides and reduces chances of chemical accumulation in the environment.
- *Minimize spray drift.* Extension Bulletin 428 and Folder 548 discuss procedures for minimizing spray drift.
- *Buffer strips.* Avoid applying chemicals in close proximity to sensitive areas (for example, sinkholes). A 50 foot no application area or a width consistent with vegetated buffer zones discussed earlier can serve as guidelines until research indicates differently.

Handling. The greatest misuse of pesticides occurs in the handling processes.

- *"Empty" pesticide* containers are seldom empty. Some undiluted chemical remains, Disposing of unrinsed "empty" containers or partially full or full containers in sinkholes, ravines, disappearing streams, and quarries, places chemicals in close proximity to pathways leading to groundwater. Disposal of empty containers in sinkholes and other karst features does occur in southeast Minnesota. Emptying the contents of full or partially full containers into these features or into roadside ditches is even more hazardous.

"Empty" containers should not be used to store food, feed, or water. Glass, metal, or plastic containers should be triple rinsed and this rinse water added to the makeup water of the applicator (when water is the carrier). The triple-rinsed containers as well as paper bag containers should then be disposed of in certified sanitary landfills. Metal containers can also be sent to drum reconditioners for recycling. Crush or puncture triple-rinsed metal containers before sending to a landfill.

Some landfill operators have been unwilling to accept containers fearing that the containers have not been triple rinsed. But the Minnesota Department of Agriculture is currently developing a container disposal certification program. Farmers will be encouraged to certify that they have triple-rinsed containers; reconditioners and landfill operators may then more willingly accept containers. Southeast farmers should join this program when it gets started.

Partially full or full containers which for some reason cannot be used, should if possible, be returned to the seller or manufacturer. Alternatively, a materials exchange site could be established. Consequently, farmers needing a chemical that others have in surplus can contact one another. If this is not possible, store the chemicals in a safe area and contact local officials, MPCA personnel, or the Minnesota Department of Agriculture for instructions. The stored containers should be periodically checked for leaks. Caches of arsenic based and other highly toxic pesticides should be called to MPCA officials' attention.

- *Formulation, tankfilling, and equipment washing activities,* if performed near disappearing streams, sinkholes and open-topped or improperly grouted wells can be hazardous because spilled chemicals, tank overflow, or wash water have only a short distance to travel to groundwater. These activities should be located at least 200 feet from wells, sinkholes, drainageways, ponds, and streams, and should not be sited on coarse soils overlying shallow bedrock. Never leave a sprayer unattended while the tank is being filled.
- *Lack of anti-siphoning devices* on tank filling equipment can result in dilute pesticide formulation moving down yard hydrant pipes into the soil and fractured limestone bedrock and then to groundwater (if the hydrant is shut off and the filling hose remains in the tank). Backflow in filler hoses can also occur when water pumps are used which have no devices preventing backflow (for example, pumping from a stream). Tank fillers should be equipped with anti-siphoning devices.
- *Pesticide storage* should be in original containers with labels intact. Never store pesticides with livestock feed, minerals, or other feed supplements. Pesticide storage areas should be separate and isolated from other facilities, as well as lockable. The area should be high and dry.
- *Disposal of excess chemicals in the sprayer* can be hazardous if the chemicals are indiscriminately dumped in one location—particularly in drainageways leading to sinkholes or disappearing streams or on shallow coarse soils. Carefully computing the amount of chemical formulation necessary to treat the target area and preparing no excess eliminates this problem. Excess chemicals, if remaining, should not be released in one spot. Waste pesticide solutions should preferably be land-applied at the same rate as for the target area and away from karst features.

Additionally, pesticide users may wish to consult university soil scientists to see if a portion of the farm could be used for excess applicator chemicals disposal. The area should not drain to sinkholes, well tops, or surface waters. Soil depth over limestone should be great and percolation rates should be moderately low. Cultivated fallow of the dedicated area may be necessary.

FERTILIZER USE

Excessive nitrogen fertilizer application. Applying more nitrogen fertilizer than crops can use during a year can result in excess nitrogen moving downward in the soil. Groundwater contamination can occur if the soils are sandy and the water table or limestone bedrock is near the soil surface (for example, 3-5 feet). Extension Bulletin 416 recommends fertilizer rates for various crops and yield goals. The nitrogen supplying power of soil organic matter and preceding leguminous crops is considered in the recommendations. Applying at recommended rates reduces chances of groundwater contamination—unless the expected crop yield is greatly overestimated.

Timing and manner of application. Nitrogen applied to soils at low crop demand periods (for example, late fall, winter, and spring) has the potential to leach downward if nitrogen is in the soluble nitrate form and water is moving downward in the soil profile (ammonium nitrate contains half nitrate and most other forms of fertilizer nitrogen eventually are converted to nitrate).

Applying nitrogen fertilizer to frozen ground, and at times of high runoff can result in nitrogen moving to sinkholes and streams when the site of application is near these features.

Usually, nitrogen fertilizer should not be applied on frozen ground or during the fall on coarse-textured soils (sands to loamy sands). Fall nitrogen fertilization should also be minimized on other soil types if possible. If not possible, select a nitrogen form that is not highly mobile. Incorporate nitrogen fertilizer, when possible, on high runoff fields draining to sinkholes and disappearing streams.

STORAGE FACILITIES

Leaking or ruptured storage tanks containing fuel oil, animal or human wastes, silage or chemicals result in contaminants moving toward groundwater. Underground tanks in areas of shallow soil over limestone bedrock result in only a few feet of soil separating potential leaks from channels leading to groundwater. Lack of periodic tank inspection unnecessarily increases risks.

Above ground storage facilities should be used in shallow soil areas. Periodic maintenance and inspection of both above and below ground tanks, including silos, is important. Leaks should be identified and controlled.

Figure 8. Sinkhole-dotted field in southeast Minnesota.
(Reprinted with permission from the *Minneapolis Tribune*)

Summary

Groundwater in southeast Minnesota's karst area is extremely susceptible to pollution. Shallow groundwater contamination is occurring. Contamination of deep, high-quality waters can also occur. Shallow aquifers will continue to be contaminated and deep aquifers will likely become contaminated if measures are not taken to reduce pollution.

The nature of karst areas permits many activities to contribute to groundwater pollution as well as allowing one individual to affect the quality of many individuals' well water. Consequently, all southeast Minnesota residents must consider the sensitive nature of karst areas when performing everyday activities and take measures when necessary to avoid groundwater contamination.

Practices listed in this publication can reduce pollution potential. Some require little effort to perform; others require commitment of time and money. Local experts should be consulted, however, to determine the need for and selection of the appropriate practice(s) for specific circumstances.

Finally, southeast Minnesota residents may wish to consider the development of local groundwater protection programs. Such programs might help offset the cost to individual landowners for some of the more expensive practices and insure that all individuals take measures to protect groundwater. Options for local government involvement include participation in feedlot pollution control programs; regulations governing home sewage treatment systems; development and implementation of waste recovery, recycling, or disposal plans; expanded well water testing and abandoned well identification programs; and sinkhole protection guidelines.

Helpful Agencies

Sinkholes	Home sewage treatment	Wells	Livestock and feedlots	Erosion and runoff control	Pesticides	Fertilizers	Geology	Agency
								LOCAL
■			■	■				Soil and Water Conservation Districts
■	■	■	■	■	■	■	■	Agricultural Extension Service
	■	■	■					County Health and/or Zoning
■	■	■	■	■			■	USDA-Soil Conservation Service (SCS)
			■	■			■	USDA-Agricultural Stabilization and Conservation Service (ASCS)
								REGIONAL
	■	■						Minnesota Department of Health Southeast District 1220 4th Ave. Southwest Rochester, MN 55901 (507) 285-7289
	■		■		■			Minnesota Pollution Control Agency 1200 S. Broadway Rochester, MN 55901 (507) 285-7343
								STATE
			■		■	■		Minnesota Department of Agriculture Agronomy Services Division 90 West Plato Blvd. St. Paul, MN 55155 (612) 296-6121
		■						Minnesota Department of Health Division of Environmental Health 717 Delaware St. Southeast Minneapolis, MN 55440 (612) 296-5338
■		■					■	Minnesota Geological Survey 1633 Eustis St. St. Paul, MN 55108 (612) 373-3372
	■		■		■			Minnesota Pollution Control Agency 1935 West County Road B2 Roseville, MN 55113 (612) 296-7373

Other Educational Materials

PUBLICATIONS

University of Minnesota Agricultural Extension Service publications can be obtained from local county extension offices or the Bulletin Room, 3 Coffey Hall, 1420 Eckles Ave., University of Minnesota, St. Paul, MN 55108.

Composting

University of Minnesota Agricultural Extension Service

Building a Compost Heap. Soils Fact Sheet 12

Minnesota Pollution Control Agency

Composting for a Better Garden and a Better Environment

Erosion Control

University of Minnesota Agricultural Extension Service

Tillage—Its Role in Controlling Soil Erosion by Water. Folder 479

Estimating the Effects of Crop Residue Mulches on Soil Erosion by Water. Folder 477

Grassed Waterways—Construction and Maintenance. Folder 480

Modern Terraces for Soil Conservation. Folder 499

Feedlots and Manure

University of Minnesota Agricultural Extension Service

Livestock Waste Facilities Handbook. Midwest Plan Service-18

Using Manure as a Fertilizer. Folder 168

Tax Benefits for Animal Pollution Control. Agricultural Engineering Fact Sheet 20

Minnesota Environmental Quality Board (101 Capitol Square Building, St. Paul, MN 55101)

Environmental Issues Relating to Animal Feedlots

Fertilizer

University of Minnesota Agricultural Extension Service

Fertilizer Recommendation Tables for Guide to Computer Programmed Soil Test Recommendations in Minnesota. Bulletin 416

Home Sewage Treatment

University of Minnesota Agricultural Extension Service

Town and Country Sewage Treatment. NCR Bulletin 130

Shoreland Sewage Treatment. Bulletin 394

How to Run a Percolation Test. Folder 261

Treatment and Disposal of Milkhouse and Milking Parlor Wastes. M-159

Minnesota Pollution Control Agency (1935 West County Road B2, Roseville, MN 55113)

Land Application and Utilization of Septage—Recommended Guidelines

Landfills and Recycling

Minnesota Pollution Control Agency

Recycling Information

Some Things Don't Belong in Your Trash Can

Operating a Recycling Program: A Citizen's Guide

Pesticides

University of Minnesota Agricultural Extension Service

Pesticide Applicator's Manual. Bulletin 428

How to Calculate Herbicide Rates and Calibrate Herbicide Applicators. Agricultural Chemicals Fact Sheet 5

Herbicide Spray Drift. Folder 548

Pesticide Storage and Formulation Shed. Agricultural Chemicals Fact Sheet 4

Fire Hazards of Stored Pesticides on Farms. Agricultural Chemicals Fact Sheet 1

Pesticides and Pesticide Container Disposal. Agricultural Chemicals Fact Sheet 17

Wells

University of Minnesota Agricultural Extension Service

Private Water Systems Handbook. MWPS-14

Chlorination of Private Water Supplies. M-156

Iowa State University Cooperative Extension Service (Ames, Iowa 50011)

Good Wells for Safe Water

Office of the State Register, Department of Administration, Documents Section (117 University Ave., St. Paul, MN 55155)

Minnesota Code of Agency Rules. Department of Health Water Well Construction Code (7MCAR: 1.210-1.224)

FILMS

Secrets of Limestone Groundwater. 13 minutes. Indiana University (available from Minnesota Agricultural Extension Service, Communication Resources)

TAPE-SLIDE SETS

Inquire at Minnesota Agricultural Extension Service, Communication Resources, about *Groundwater Pollution in Southeast Minnesota's Karst Region*, a companion to this publication.

Recycling Facilities in Southeast Minnesota

(check business hours with each)

DAKOTA COUNTY

Metals

Coca-Cola
Town's Edge Shopping Center
Farmington 55024
(507) 388-2951
aluminum

Coca-Cola
Mun. Liquor-Holyoke Ave
Lakeville
(507) 388-2951
aluminum

Glass

Hampton B&B 4-H Club
c/o Vernon Hupf-260th St
Randolph 55065
(507) 263-2705

Alcorn Beverage Co.
7879 218th St W
Lakeville 55044
(612) 469-5555

Faith Lutheran Church
7095 Upper 163rd St
Rosemount 55068
(612) 432-4658

Donal Tutewoht
23142 Denmark Ave
Farmington 55024
(612) 463-7489

Tim Turek
14809 Chili Ave W
Rosemount 55068
(612) 423-2888

Full service

Stoffel Beverage Co.
1272 W 8th St
Hastings 55033
(612) 437-6466
glass, aluminum

John Ginther
1226 Eddy
Hastings 55033
(612) 437-3570
glass, aluminum

Trinity Lutheran Church
413 Main St
Farmington 55024
(612) 463-8922
paper, glass

DODGE COUNTY

Metals

Coca-Cola
American Legion
Dodge Center
(507) 388-2951
aluminum

Coca-Cola
Municipal Parking Lot
Kasson-Mantorville
(507) 388-2951
aluminum

Darrel Quesnel
RR 1, Box 264A
Dodge Center 55927
(507) 374-6660
paper, corrugated, cans scrap metal, glass

Lin's Used Iron
502 3rd St. SE
Dodge Center 55427
(507) 374-2439
scrap metals, aluminum cans (not steel cans)

GOODHUE COUNTY

Metals

Coca-Cola
Hub Red Owl
Zumbrota 55066
(507) 388-2951
aluminum

Reynolds Aluminum
Pamida Store-Hwy 61 & Tylan Rd
Red Wing 55066
(800) 288-2525
aluminum

Coca-Cola
Pamida-Hwy 61 & Tylan Rd
Red Wing
(507) 388-2951
aluminum

Coca-Cola
Cannon Mall
Cannon Falls 55009
(507) 388-2951
aluminum

Buf's Truck Parts
Hwy. 56
Cannon Falls 55009
(507) 263-2226
scrap metal, aluminum cans

Glass

George Lucius
1005 W Hauffman St
Cannon Falls 55009
(507) 263-2594

Erwin Buck
610 Lincoln Ave
Zumbrota 55992
(507) 732-5836

MOWER COUNTY

Paper

First Methodist Church
204 1st Ave N
Austin 55912
(507) 433-8839

Pacelli School
311 4th St NW
Austin 55912
(507) 437-3278

Metals

Coca-Cola
Oak Park Mall
Austin 55912
(507) 388-2951
aluminum

Reynolds Aluminum
K-Mart Parking Lot
Austin 55912
(800) 228-2525
aluminum

Chas. Dubinsky & Co.
10th Dr. & 8th Ave. SE
P.O. Box 29
Austin 55912
(507) 433-3496
all metals

Gopher Distributing Co
Hwy 218 N
Austin 55912
(507) 437-3278
aluminum

Crowley Beverage Co.
617 NE 11th St
Austin 55912
(507) 433-8295
aluminum

Full Service

Delmar Ellis
Rt. 5
Austin 55912
(507) 437-1893
cans, glass, paper

OLMSTED COUNTY

Metal

Gopher Distributing Co
1640 SE 3rd Ave
Rochester 55901
(507) 288-4211
aluminum

Reynolds Aluminum
Apache Mall-Hwy 52 & 14
Rochester 55901
(800) 228-2525
aluminum

Rochester Iron & Metal
1950 3rd Ave. SE
Rochester 55901
(507) 288-3228
sheet iron, beverage cans, scrap metals (not steel cans or wire)

Coca-Cola
Apache Mall-Hwy 52 & 14
Rochester 55901
(507) 388-2951
aluminum

Coca-Cola
Boyum Foods
Stewartville 55976
(507) 388-2951
aluminum

Chaddock Truck Parts
832 14th St. NW
Rochester 55901
(507) 288-3346
scrap tin

Sexton Auto Parts & Salvage
Route 2 Box 139
Rochester 55901
(507) 282-3777
scrap metal, aluminum and steel cans

Paper

S.E. Minnesota Recycling
4802 8th St. SW
Rochester 55901
(507) 289-7510
newspaper

Glass

Rodney Watson
809 1st St SE
Rochester 55901
(507) 282-7710

Full Service

Hemker Recycling
1214 1st St NE
Rochester 55901
(507) 282-4729
glass, paper, aluminum

RICE COUNTY

Metal

Reynolds Aluminum
Faribault Plaza-Hwy 65 & Division
Faribault 55021
(800) 228-2525
aluminum

Coca-Cola
Faribault 55021
(507) 388-2951
aluminum

Harley's Auto
510 NW 20th St
Faribault 55021
(507) 334-8290
metals: all kinds

Kelley's Auto Parts
Faribault 55021
(507) 334-7035
scrap metals, batteries, aluminum cans

Viking Auto Salvage
N. Hwy. 3
Northfield 55057
(507) 645-5819
(612) 332-0660
scrap metals, aluminum and steel cans

Glass

Sunrisers 4-H Club
Rt 2
Northfield 55057
(507) 645-8185

Full Service

Consolidated Catholic Schools
Home and Schools Assoc.
Faribault 55057
glass, aluminum, newspaper, flattened cardboard

STEELE COUNTY

Metal

Coca-Cola
Prairie House Parking Lot
Blooming Prairie 55917
(507) 388-2951
aluminum

Coca-Cola
Cedar Mall
Owatonna 55060
(507) 388-2951
aluminum

Reynolds Aluminum
Pamida Store
Owatonna 55060
(800) 228-2525
aluminum

Glass

H & S Distributing Co
670 24th Ave NW
Owatonna 55060
(507) 451-4169

Owatonna Redemption Center
1031 S Oak
Owatonna 55060
(507) 451-1320

Full Service

Owatonna Reclamation Center
453 Clearview Place
Owatonna 55060
(507) 451-8846
glass, newspaper, aluminum, tin

Cumberland Hide & Fur, Wool & Metal Co.
Box 408 Route 3
Owatonna 55060
(507) 451-7607
all nonferrous metals, aluminum cans

Owatonna Scrap Iron & Metal
P.O. Box 72
Owatonna 55060
(507) 451-1470
all metals

Poly Plastic
18th St.
Owatonna 55060
(507) 451-8650
plastics, cars, newspaper, cardboard, office paper

WABASHA COUNTY

Metal

Coca-Cola
Super Valu
Lake City 55041
(507) 388-2951
aluminum

Coca-Cola
Lannings Red Owl
Plainview 55964
(507) 388-2951
aluminum

Lake City Auto Parts
Lake City 55041
(612) 345-4224
scrap metals (no cans)

WINONA COUNTY

Metal

William Miller Scrap Iron & Metal
222 W. 2nd St.
Winona 55987
(507) 452-2067
metals

S. Weisman & Sons, Inc.
450 W. 3rd St.
Winona 55987
(507) 452-5847
aluminum

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Township Testing Program Update-May 2022

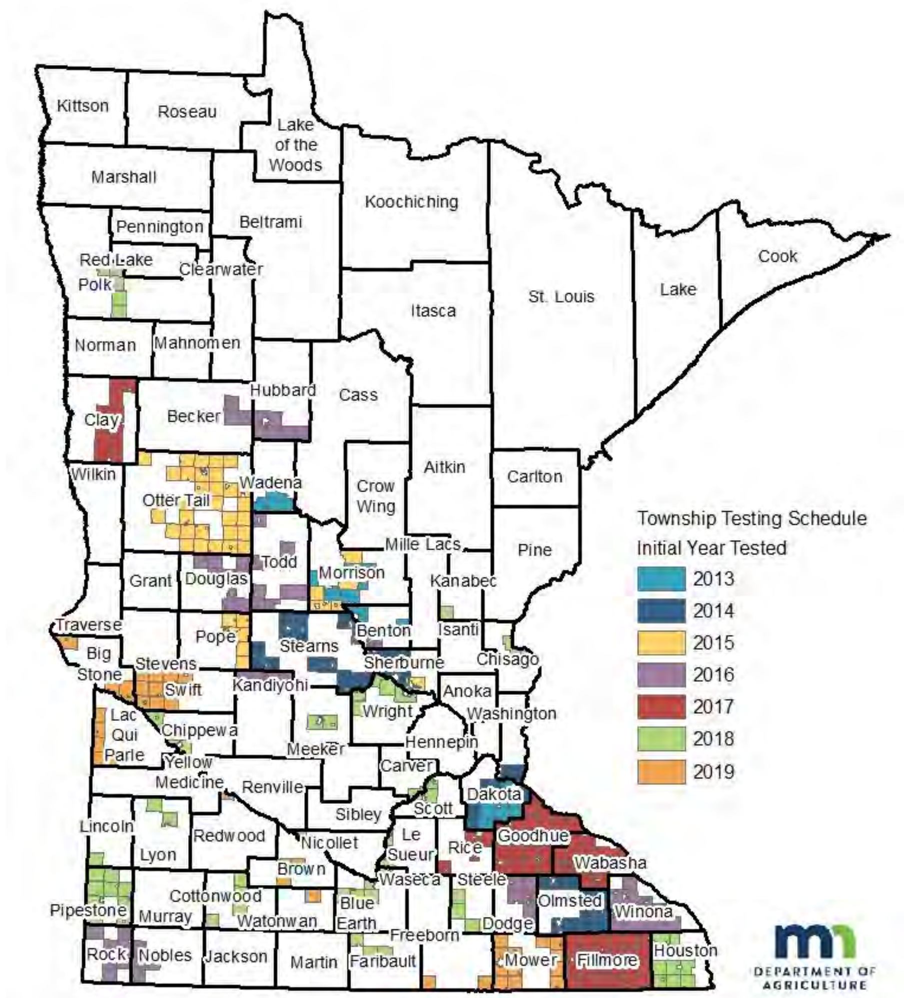

Figure 1. Township Testing Schedule

In a seven-year statewide effort, the Minnesota Department of Agriculture (MDA) offered nitrate-nitrogen (Nitrate-N) tests to private well owners. This extensive sampling effort was conducted as a result of a major revision of the Nitrogen Fertilizer Management Plan (NFMP). The NFMP called for an assessment of nitrate conditions at the township scale. In response, a statewide Township Testing Program (TTP) was established to assess the nitrate-nitrogen (Nitrate-N) concentrations in private wells.

Townships that are vulnerable to groundwater contamination and have significant row crop production were selected for nitrate testing. Some factors that make groundwater vulnerable are soil type and geology, which control how quickly nitrate can travel from the root zone to groundwater.

More than 90,000 private well owners were offered nitrate testing in 344 townships in years 2013 to 2019 for initial testing (Figure 1). Additional testing follow up continued through 2020.

The TTP was a substantial multi-year sampling effort to evaluate water quality in drinking water wells in areas vulnerable to ground water contamination from agricultural sources across the entire state and was a significant step towards addressing nitrate in groundwater in Minnesota. The data gathered is used to inform well owners about the water they are drinking and can be used to prioritize future work to address nitrate concerns, as described in the NFMP. Find more information about the NFMP at www.mda.state.mn.us/nfmp.

Initial Results

The MDA works with local partners such as counties and soil and water conservation districts (SWCDs) to coordinate private well nitrate testing using Clean Water Funds. In the initial sampling, all township homeowners using private wells were sent a nitrate test kit and the homeowner collected the sample.

From 2013-2019, 344 vulnerable townships from 50 counties participated in the initial TTP sampling. In the 344 townships tested, 143 (41%) had 10% or more of the wells over the Health Risk Limit (HRL) of 10 mg/L for Nitrate-N (Figure 2 & 3).

Through the TTP 32,217 private wells were tested for nitrate. Of the wells tested, 2,925 (9.1%) exceeded the HRL for Nitrate-N (Table 2). The minimum nitrate result was less than the detection limit and the maximum result was 159 mg/L Nitrate-N (Table 2). These initial results reflect nitrate concentrations in private well drinking water regardless of nitrogen sources, or well construction.

Final Results

If nitrate was detected in the initial sample, the homeowner was offered a follow-up nitrate test, pesticide test, and well site assessment. Trained MDA staff visited willing homeowners to collect the follow-up nitrate and pesticide water samples and conduct well site assessments, between 2014 and 2020. Once completed, the MDA analyzed the results and prepared a final report for each county. Final results were determined using two rounds of sampling and a process to remove wells with construction concerns, insufficient construction information, and those near potential non-fertilizer sources of nitrate. Final results represent wells that are potentially impacted by a fertilizer source.

For the final dataset, it was determined that 44 (13%) townships had 10% or more of the wells over the HRL for Nitrate-N, with the majority of these townships occurring in southeast Minnesota. For the final results, townships with less than 20 well were categorized separately because MDA considers less than 20 wells inadequate to characterize a township for the purposes of the NFMP (Figure 2 & Figure 4).

In the final dataset of 28,932 wells, 1,359 (4.7%) exceeded the HRL for Nitrate-N (Table 2). The minimum nitrate result was less than the detection limit and the maximum result was 69.8 mg/L Nitrate-N (Table 2). Detailed sampling results for each county are available at: https://www.mda.state.mn.us/township-testing-program. A detailed final report on statewide and regional data comparisons will be available in 2023.

Pesticide results were analyzed separately through the Private Well Pesticide Sampling Project, more information is available at: www.mda.state.mn.us/pwps

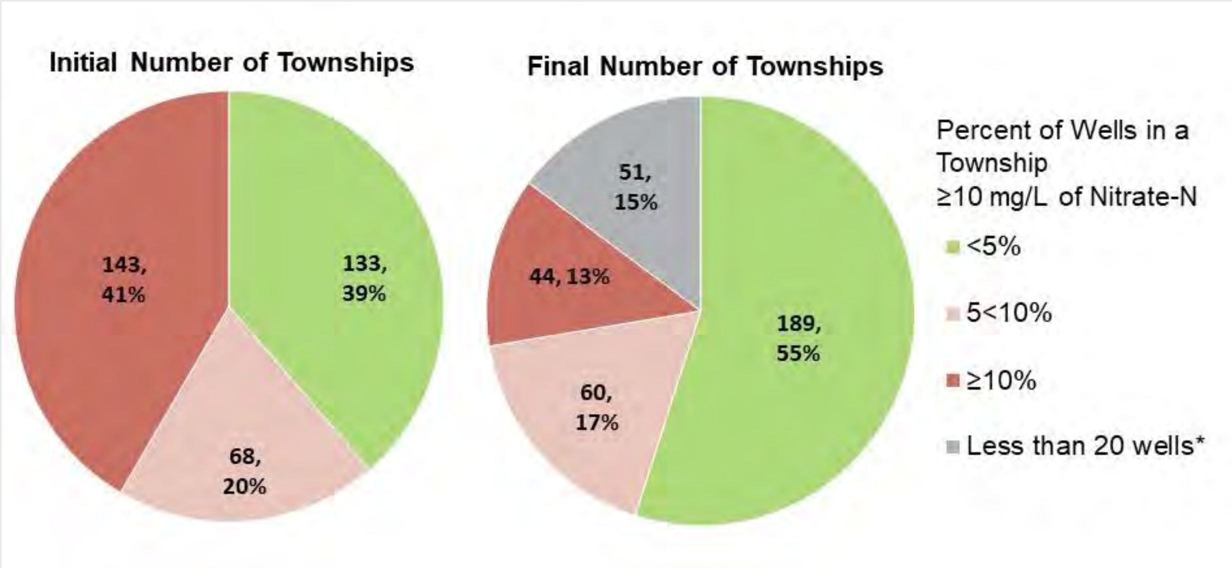

Figure 2. Initial and final number of townships and percent of townships in each nitrate category.
* Townships with less than 20 well were categorized separately because MDA considers less than 20 wells inadequate to characterize a township for the purposes of the NFMP

Table 1. Initial and final number of wells and percent of wells in each nitrate concentration range.

Township	Total Wells	Number of Wells <3*	Number of Wells 3<10*	Number of Wells ≥10*	Percent of Wells <3*	Percent of Wells 3<10*	Percent of Wells ≥10*
Initial	32,217	24,791	4,501	2,925	77.0%	14.0%	9.1%
Final	28,932	24,512	3,061	1,359	84.7%	10.6%	4.7%

* Nitrate-N mg/L or parts per million (ppm)

Table 2. Township testing program summary statistics for initial and final well dataset

Township	Total Wells	Min Value*	Max Value*	Mean Value*	50th Percentile[2]* (Median)	75th Percentile[2]*	90th Percentile[2]*	95th Percentile[2]*	99th Percentile[2]*
Initial	32,217	<DL[1]	159	3.5	1.7	4.5	9.4	13.9	22.2
Final	28,932	<DL[1]	69.8	1.8	0.6	2.1	5.1	8.1	14.6

[1]<DL means that this value is less than detection limit of the lab, which is typically between 0.03 and 0.25 mg/L nitrate-N.

[2]The 50th percentile (75th, 90th, 95th, and 99th, respectively) is the value below which 50 percent (75%, 90%, 95% and 99%) of the observed values fall

* Nitrate-N mg/L or parts per million (ppm)

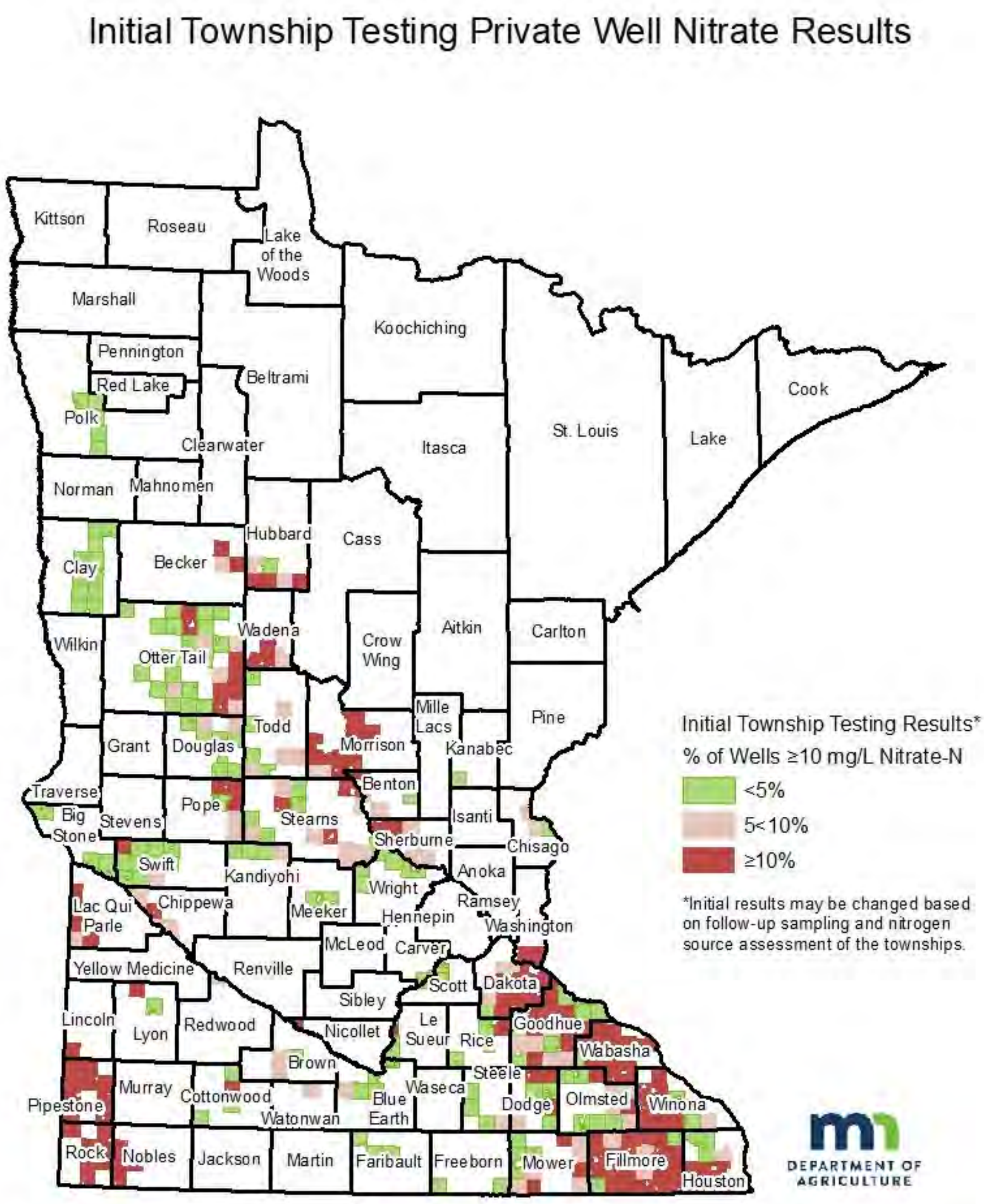

Figure 3. Initial Township Results Updated May 2022

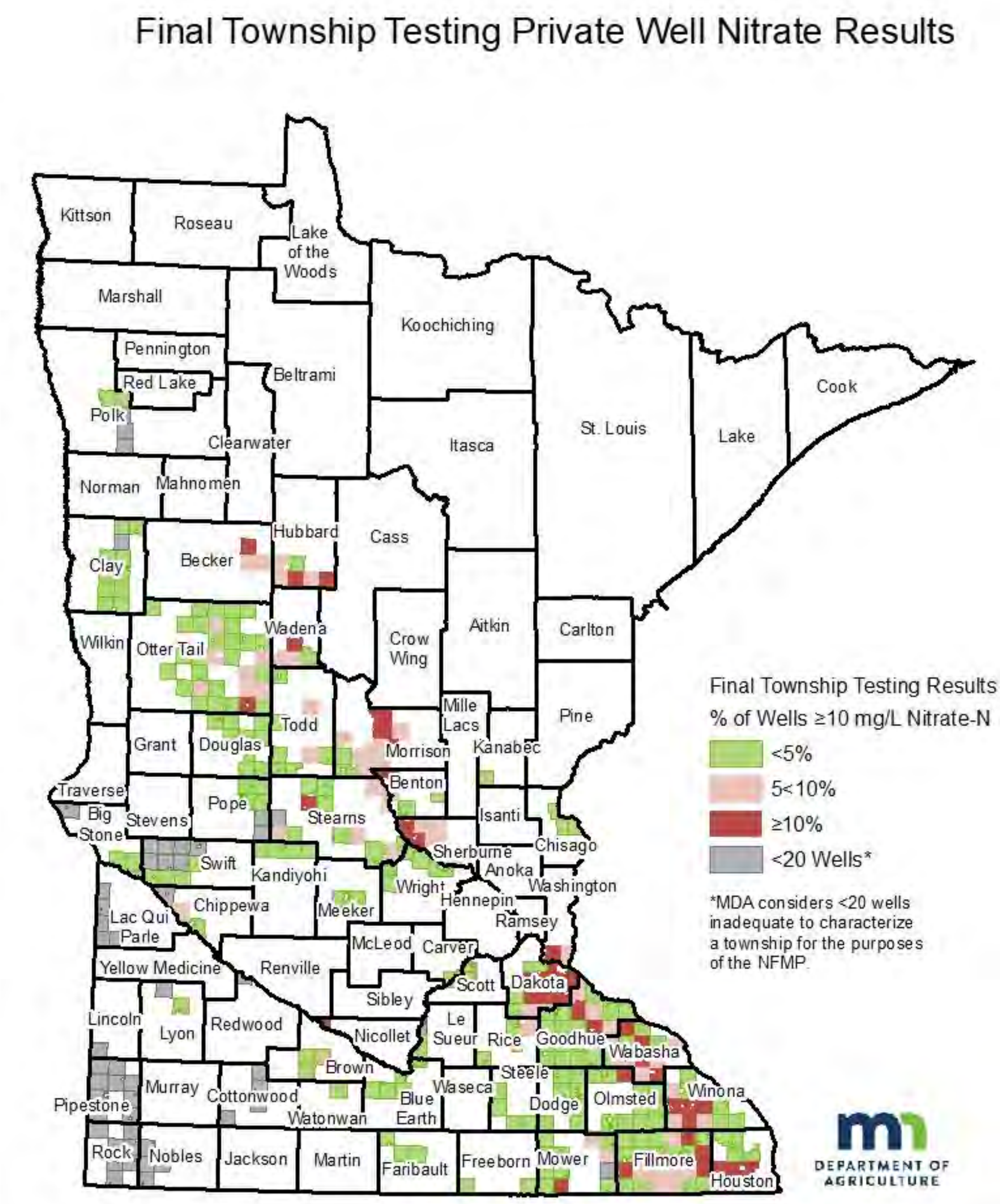

Figure 4. Final Township Results Updated May 2022

REGION 5 ADMINISTRATOR
CHICAGO, IL 60604

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Dear Dr. Cunningham, Mr. Peterson, and Ms. Kessler:

On April 24th, 2023, Petitioners[1] requested that the U. S. Environmental Protection Agency exercise its emergency powers under Section 1431 of the Safe Drinking Water Act (SDWA) to address groundwater nitrate contamination that presents a risk to the health of the residents in eight counties of the Southeast Karst Region[2] (Karst Region) of Minnesota. Section 1431 authorizes EPA to act upon receipt of information that a contaminant is present in or is likely to enter a public water system (PWS) or an underground source of drinking water (USDW), which may present an imminent and substantial endangerment to the health of persons, and that appropriate state and local authorities have not

[1] Petitioners: Minnesota Center for Environmental Advocacy, Environmental Working Group, Minnesota Well Owners Organization, Center for Food Safety, Clean Up the River Environment, Food & Water Watch, Friends of the Mississippi River, Izaak Walton League Minnesota Division, Land Stewardship Project, Minnesota Trout Unlimited, and Mitchell Hamline Public Health Law Center.
[2] Minnesota's Karst Region referenced in the petition consists of eight counties: Dodge, Fillmore, Goodhue, Houston, Mower, Olmsted, Wabasha, and Winona county.

acted to protect the health of such persons. Approximately 390,682[3] people reside in the Karst Region; about 300,000 people are served by 93 PWSs and approximately 93,805[4] people rely on private wells as their primary source of drinking water. Based on the information currently available from past nitrate monitoring, it had been estimated that 9,218[5] residents in the Karst Region were or still are at risk of consuming water at or above the maximum contaminant level (MCL) for nitrate, with Minnesota Department of Agriculture reporting that 12.1% of the private wells tested (equating to 1,058 wells) exceeded the MCL of 10mg/L[6]. Several of the PWSs in the Karst Region have also been impacted by MCL exceedances resulting in additional treatment and/or having to drill deeper wells.

We appreciate the time that you and your staff have taken to meet with my staff on numerous occasions to share each agency's efforts to protect Minnesota's drinking water, including the information you shared in and after our meeting on August 28, 2023 (See Enclosure). While we appreciate the collective commitment to address nitrate contamination through state-administered programs, based on our discussions and current available drinking water data, there is an evident need for further actions to safeguard public health.

EPA's immediate priority is to protect human health by ensuring that residents impacted by nitrate contamination are: (1) identified; (2) provided notice in all applicable languages regarding their potential exposure to elevated nitrate concentrations and information regarding the associated health risks; and (3) provided the opportunity to obtain alternate drinking water until nitrate contamination in groundwater falls below the MCL for nitrate of 10 mg/L.

EPA expects state agencies to take timely actions to address the nitrate contamination, especially with respect to providing public notice and alternate water. To address these priorities, EPA requests that the Minnesota agencies develop a coordinated and comprehensive work plan to identify, contact, conduct drinking water testing and offer alternate water to all impacted persons in the Karst Region, as soon as possible, and to sustain these efforts for as long as nitrate concentrations in the groundwater of the Karst Region remain at or above the MCL. An adequate work plan to address immediate health concerns should include the following:

1. **Coordination** – The state should create a communication plan that identifies how information and responsibilities will be shared among the state agencies, local governments

[3] Calculated using the 2022 data, for each county, reported on the Minnesota State Demographic Center "PopFinder For Minnesota, Counties, & Regions". https://mn.gov/admin/demography/data-by-topic/population-data/our-estimates/pop-finder1.jsp
[4] Calculated using Minnesota Department of Health "Community Water Systems: MNPH Data Access" to determine population serviced by CWS's, then subtracted by the population in the region. https://mndatamaps.web.health.state.mn.us/interactive/cwss.html last updated 03/07/2023.

[5] Calculated using the Township Testing Program "Final Report" by adding up the estimated population at risk, reported in the "Estimates of Population at Risk" section of each report, for each county. Data used ranges from 2014 – 2019. https://www.mda.state.mn.us/township-testing-schedule-reports
[6] From the Township Testing Program county reports for this region.

(county, city, township), and any private businesses or local utilities that have volunteered or been required to act, so that each entity's efforts serve a singular and coordinated response.

2. **Identification of Impacted Residences** – The state should identify each residence that obtains drinking water from a private well within the Karst Region. This includes wells that were constructed prior to the adoption of Minnesota's Well Code.

3. **Education and Outreach** – The state should provide notice to newly and previously impacted residents and continue to provide notice as long as contamination persists at or above the MCL for nitrate. If notice has not been provided to those that were previously identified as having private drinking water wells at or above the MCL for nitrate, we expect the state to provide notice *immediately* to such residents.

Similarly, if notice has not been provided to customers served by regulated PWSs that had nitrate levels at or above the MCL, we expect the state or owner/operators to provide notice *immediately*. Public education and outreach should be conducted in a form and manner reasonably calculated to reach all impacted residents in all applicable languages.

The state should prioritize its education and outreach toward the most vulnerable populations for associated health risks (e.g., homes with infants, pregnant women), including efforts to work with health care facilities and daycares serving such populations.

In addition to public health information, clear instruction for private drinking water well users to request drinking water testing should be included in appropriate languages. Minnesota should measure its progress in contacting all private well users identified as part of outreach efforts. For those private well users that do not respond to public notices, Minnesota should attempt personal communications, such as visits to individual residences (e.g., Minnesota Water Stewards).

4. **Drinking Water Testing** – Responsible agencies should create and implement a plan to provide analysis of drinking water samples obtained from any private well users in the Karst Region that request testing. For any residents identified as having private drinking water wells at or above the MCL for nitrate, we expect the state to provide timely notice to such impacted residents.

5. **Provision of Alternate Water** – Alternate drinking water should be offered as soon as practicable to each residence where water tests show an exceedance of the MCL for nitrate in the private well. The state should prioritize provision of alternate water to particularly vulnerable populations (e.g., homes with infants, pregnant women). As part of your response to EPA, please provide a detailed plan for distribution (e.g., water made available to residents at centralized locations) and a timeline for provision of such water.

Alternate water should be provided as needed for drinking, cooking, and maintaining oral hygiene. This shall be at no cost to the resident and in a manner that minimizes the burden on the impacted resident to obtain safe drinking water, such as water distribution locations and/or delivery services, reverse osmosis treatment units, or connection to a public water system.

6. **Public Records** – Maintain and regularly publish records such that Minnesota residents and the general public can better understand the scope and severity of nitrate contamination in the Karst Region and measure Minnesota's progress in implementing its response plan including provision of alternate water, and to establish an effective way to communicate updates to the general public.

7. **Communication with EPA** – EPA requests that the Minnesota agencies provide progress reports quarterly to EPA that (a) describe actions taken during the previous quarter to address the immediate health impacts of nitrate contamination; (b) identify major accomplishments and issues that arose; (c) describe actions and timelines planned for the next quarter; and (d) describe any problems or delays encountered and the solutions implemented to address them.

While this letter is largely focused on addressing immediate health concerns regarding nitrate contamination in drinking water in the Karst Region, Minnesota must also develop and implement a long-term solution to achieve reductions in nitrate concentrations in drinking water supplies.

Developing a complete understanding of potential sources of nitrate contamination is an important immediate step for the state. A risk analysis of current and future nitrate contamination of the impacted groundwater will be critical for determining long-term solutions, and such analysis should incorporate the latest science and technologies.

Minnesota has tools to effect reductions in nitrate concentrations through the National Pollutant Discharge Elimination System (NPDES) and State Disposal System permit programs, including development and implementation of more protective NPDES/SDS CAFO permits.

In addition, Minnesota should consider adopting monitoring requirements in NPDES/SDS permits related to (1) subsurface discharges from manure, litter, and process wastewater storage, as well as (2) discharges from land application, similar to those proposed by EPA as modifications to the EPA-issued CAFO general permit for Idaho: https://www.epa.gov/npdes-permits/npdes-general-permit-concentrated-animal-feeding-operations-cafos-idaho. We also encourage Minnesota to consider modifications to the state's Technical Standards for Nutrient Management with regard to land application of manure, litter or process wastewater, and any Minnesota guidelines for land application of commercial fertilizer, specific to Karst areas.

EPA expects Minnesota to hold sources of nitrate accountable using all available tools to reduce the amount of nitrate they release to ground water. While the Agency appreciates the state agencies' engagement and past efforts in addressing groundwater contamination in the Karst Region, EPA will continue to closely monitor this situation and consider exercising our independent emergency and enforcement authorities.

Given the urgency inherent in any situation involving drinking water contamination with known potential health risks, we respectfully request confirmation of your agencies' plan to provide "Education and Outreach" and "Provision of Alternate Water" as soon as possible. EPA expects a reply with respect to the elements noted above within 30 days, which must include the anticipated timeframe for submission of the agencies' work plan.

Sincerely,

DEBRA SHORE

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Date: 2023.11.03 08:31:31 -05'00'

Debra Shore
Regional Administrator
& Great Lakes National Program Manager

Enclosure: Summary of Minnesota Efforts to Address Nitrate Contamination

EPA recognizes the Minnesota's past and current efforts to address nitrate contamination:
The Clean Water council (consisting of MDA, MPCA, and MDH representatives) was able to advise the Legislature to appropriate $100,000 of the state's Clean Water Fund to the "Tap In" initiative, which was carried out at the county level, including counties in the Karst Region. This initiative in 2021 assisted low-income private well owners with nitrate contamination that exceeds the MCL. The initial grant covered 186 tests, 7 reverse osmosis filters, 6 new wells, and one well repair.

MDA and MDH created a private well network for residents in which to participate in the Central Sands and Southeast Karst Region. The purpose of the Southeast Minnesota Volunteer Nitrate Monitoring Network was to monitor long term trends of nitrate concentrations in private drinking water wells throughout Southeastern Minnesota. Samples were collected from 2008 – 2012.

MDA and MDH provide technical assistance to CWSs when the nitrate level is detected above 3 mg/L. MDA had established Nitrate Testing Clinics, which has provided 50,000 well owners with testing services and educational outreach since 1993, and local partners with equipment to carry out nitrate analysis.

MDA provided free nitrate sampling to private well owners in vulnerable Townships throughout the state from 2013 to 2019 via the Township Testing Program. Of the 344 townships determined to be vulnerable statewide, 133 are in the Karst Region.

MDA was the initial partner in the *We are Water MN*, providing technical assistance, staff time, and financial investments.

MDA continues to develop and publish videos, infographics, and additional resources targeted for residents of the Karst Region.

MDA developed the Groundwater Protection Rule to support the 2015 Nitrogen Fertilizer Management Plan, which went into effect on June 28, 2019.

MDH established and enforces laws and rules for proper construction and sealing of wells and borings and provides guidance to private well owners. MDH assists and regulates public water systems by approving system construction and treatment plans in response to nitrate issues, as well as requiring PWSs to protect water sources from contamination and providing technical assistance and grants to do so. Since 1993, MDH has successfully returned 8 CWSs and 38 NCWSs back to compliance with SDWA's regulatory limits for nitrates.

MPCA created the state's Nutrient Reduction Strategy in 2014 to guide the state in reducing excess nutrients in water to meet state and downstream water quality goals.

MPCA had released the Groundwater Protection Recommendation Report in 2016 which states recommendations for preventing nitrate contamination in groundwater.

MPCA uses NPDES permits to (1) prevent manure, litter, and process wastewater discharge to surface water from Large CAFO production areas and (2) minimize nutrient movement to surface water from manure, litter, and process wastewater application to land under the control of Large CAFOs. State Disposal System-based conditions in these permits, and in SDS-only permits for Large CAFOs, are for the purpose of protecting ground water. In a July 22, 2021 letter from MPCA to EPA, MPCA underscored that it set conditions in its 2021 statewide NPDES/SDS general permit for Large CAFOs for the specific purpose of addressing existing elevated levels of nitrates in ground water (Peter Tester letter to Cheryl Newton, page one). For decades, Minnesota has operated a supplementary state law regulatory program for feedlots as small as 50 animal units (10 in shoreland).

In addition, we thank Minnesota staff for taking time to participate in recent calls and sharing information on your work to address nitrate contamination including calls with MDH on May 8, May 18, and June 20; MDA on May 18, MPCA on August 22, and a joint call with all three agencies on August 28.

Winona County: Final Overview of Nitrate Levels in Private Wells (2016-2017)

The Minnesota Department of Agriculture (MDA) determines current nitrate-nitrogen concentrations in private wells, on a township scale, through the Township Testing Program. The MDA has identified townships throughout the state that are vulnerable to groundwater contamination and have significant row crop production. The MDA plans to offer nitrate testing to more than 70,000 private well owners in over 300 townships by 2019.

Winona County Final Highlights

- Number of townships with 10% of wells over the HRL : **4**
- **209** (22%) wells removed from initial data set.

Each selected township is offered testing in two steps, the "initial" sampling and the "follow-up" sampling. In the initial sampling, all township homeowners using private wells are sent a nitrate test kit. If nitrate is detected in their initial sample, the homeowner is offered a follow-up nitrate test, pesticide test and well site visit. Trained MDA staff visit willing homeowners to resample the well and then conduct a site assessment. The assessment helps to identify possible non-fertilizer sources of nitrate and to see the condition of the well. A well with construction problems may be more susceptible to contamination.

The MDA and Winona County Environmental Services worked together to select townships and implement the nitrate testing project. The following townships were selected: **Elba, Fremont, Hart, Hillsdale, Mt. Vernon, Norton, Pleasant Hill, St. Charles, Saratoga, Utica, Warren, Wilson, and Wiscoy.** The initial sampling in Winona County started in 2016 and follow-up sampling ended in 2017.

Results

Two datasets, "Initial" and "Final", are used to evaluate nitrate in private wells. The initial dataset represents private well drinking water regardless of the potential source of nitrate. The final dataset was formed through an assessment process to evaluate wells. In the assessment, wells that had nitrate-nitrogen results over 5 mg/L were removed from the initial dataset to form the final dataset if a potential non-fertilizer source or well problem was identified, there was insufficient information on the construction or condition of the well, or for other reasons which are outlined in the full report (see Appendix E for details). The final dataset represents wells with nitrate attributed to the use of fertilizer. The initial dataset for Winona County contains 940 wells; the final dataset contains 731 wells. A total of 209 wells (22%) were removed.

The results from the initial and final well datasets are summarized in the following table and figures. In the initial dataset nine townships had more than 10% of the wells over the Health Risk Limit of 10 mg/L of nitrate-nitrogen (see map). In the final dataset four of the townships had more than 10%. The final percent of wells over the Health Risk Limit in each township ranged from 0% to 42.9%. The Winona County Final Report is available on the MDA website in 2019: www.mda.state.mn.us/townshiptesting.

Next steps

The MDA uses the TTP data and assessment process and prioritization guidelines in the Minnesota Nitrogen Fertilizer Management Plan (NFMP) to determine next steps. It is MDA's intent to implement the voluntary aspects of the NFMP in townships with elevated nitrate with the highest priority placed on areas with high sampling results. Find more information about the NFMP on the MDA website at www.mda.state.mn.us/nfmp.

Funding for this project is provided by the Clean Water, Land and Legacy Amendment

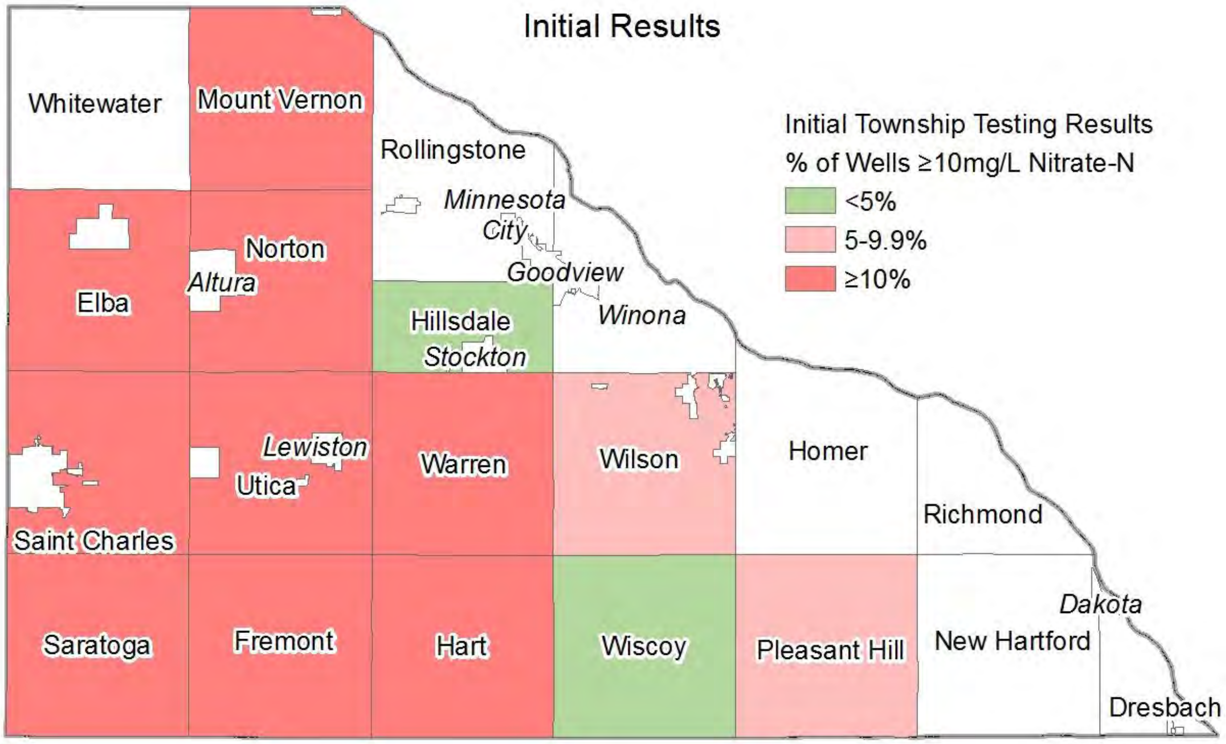
Updated September 2019

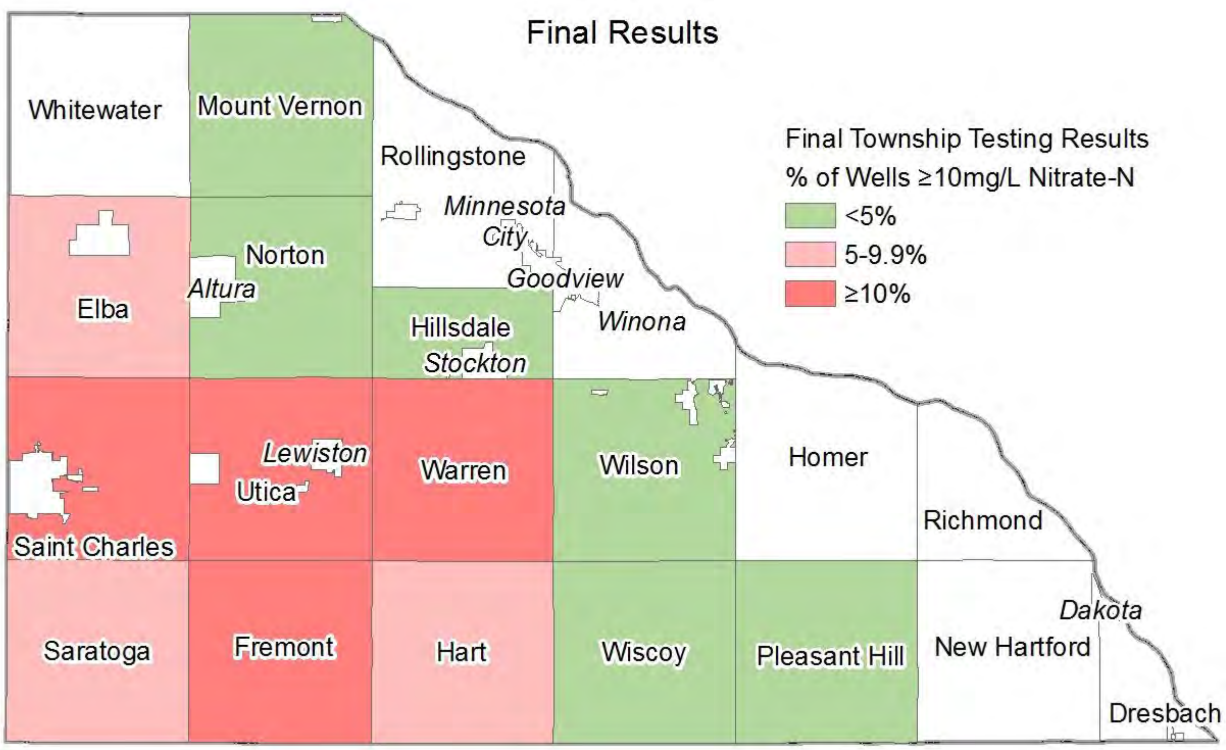
Table: Winona County Private Well Nitrate Results.

Township	Initial Well Dataset		Final Well Dataset	
	Total Wells*	Percent of Wells ≥10 mg/L Nitrate-Nitrogen	Total Wells	Percent of Wells ≥10 mg/L Nitrate-Nitrogen
Elba	62	16.1%	52	5.8%
Fremont	42	54.8%	28	42.9%
Hart	48	18.8%	31	6.5%
Hillsdale	52	1.9%	44	0.0%
Mt. Vernon	33	15.2%	24	0.0%
Norton	80	11.3%	62	4.8%
Pleasant Hill	58	8.6%	50	4.0%
St. Charles	85	34.1%	62	14.5%
Saratoga	56	19.6%	40	5.0%
Utica	86	46.5%	51	19.6%
Warren	92	28.3%	62	11.3%
Wilson	196	6.1%	179	1.7%
Wiscoy	50	0.0%	46	0.0%
Total	940	19.1%	731	7.1%

* All well types included.

Figure: Winona County Final Well Dataset Map. Clean Water Organizations Comments Exhibit 19





September 3, 2024
Clean Water Organizations Comments Exhibit 20

Southeast Minnesota Volunteer Nitrate Monitoring Network

mda.state.mn.us/southeast-minnesota-volunteer-nitrate-monitoring-network

Drinking water high in nitrate can cause serious health effects in infants. The state's Health Risk Limit (HRL) for nitrate-nitrogen is 10 mg/L. Karst geology makes the region's groundwater especially vulnerable to nitrate contamination. Because of this risk it is important to monitor for high nitrate concentrations in private wells.

In 2006, nine southeast Minnesota counties coordinated planning to develop a Volunteer Nitrate Monitoring Network (VNMN) to monitor long term trends of nitrate concentrations in private drinking water wells throughout southeastern Minnesota. From 2006 until 2012 the Project team included nine southeastern Minnesota counties and multiple state agencies funded by the EPA 319 Program and the MPCA Clean Water Partnership (CWP) Program. The first two years of the project were primarily the planning stage, the first round of samples were collected in 2008. In 2013, the program was changed to incorporate more analytes in selected wells, but was no longer sampling the entire network for nitrate. In 2014, the MDA coordinated with the County Water Planners and Southeast Minnesota Water Resources Board (SEMNWRB) to continue sampling all of the wells in the network on an annual basis to determine long term trends and keep the original network intact where possible.

Homeowners are the cornerstone of this network, this work could not be done without them. Network participants are sent a nitrate test kit directly to their home on an annual basis by the lab. The homeowner simply fills up the bottle and sends it directly back to the lab for analysis. The lab then sends homeowners their results.

In 2022, 376 private drinking water wells were sampled for nitrate. Results from 2022 are similar to previous years:

- 69.4% of nitrate results were < 3 mg/L
- 22.3% of nitrate results were 3<10 mg/L
- 8.2% of nitrate results were ≥10 mg/L

Southeast Volunteer Nitrate Monitoring Network Summaries

The nitrate testing results from this network of private wells is used in combination with other networks to determine the trend of nitrate levels in regional groundwater over time. The nitrate results and trend reports are available in the Minnesota Water Research Digital Library. Links to the most recent reports are listed below.

Yearly Results

- Southeast Minnesota Volunteer Nitrate Monitoring Network 2022 Results
- Southeast Minnesota Volunteer Nitrate Monitoring Network 2021 Results
- Southeast Minnesota Volunteer Nitrate Monitoring Network 2020 Results

Older reports are available in the Minnesota Water Research Digital Library . Search for reports using the following titles: Southeast Minnesota Volunteer Nitrate Monitoring Network, or Southeast Minnesota Domestic Well Network.

Trend Reports

- Nitrate Results and Trends in Private Well Monitoring Networks (2008-2018)
- Nitrate Trends in Private Well Networks (2017)

An overview:

- Nine counties in the Southeast region participate in the county wide private well network
- In 2020, 381 private drinking water wells were sampled for nitrate , 91% have water that is below the HRL
- Nitrate analysis of approximately 300-600 wells have been completed annually
- This project will help answer the question: Are nitrate concentrations in private drinking water wells increasing, decreasing or staying the same?

Why is this program focused on nitrates?

Nitrate is a water soluble molecule that is made up of nitrogen and oxygen. It is naturally occurring in the environment; however at elevated levels it can have negative effects on human health. According to a 2007 Minnesota Pollution Control report, nitrate is one of most common contaminants in Minnesota's groundwater, and in some areas of the state a significant number of wells have high nitrate levels (Minnesota's Ground Water Condition: A Statewide View, MPCA 2007). The U.S. Environmental Protection Agency (EPA) has established a drinking water Maximum Contaminant Level (MCL) of 10 mg/L for nitrate-nitrogen (EPA, 2009). Although nitrate occurs naturally, it can also originate from man-made sources such as fertilizer, animal manure and human waste.

Regions of Minnesota most vulnerable to nitrate contamination are central and southeastern Minnesota. Central Minnesota counties are vulnerable because of widespread sandy soil and regions of southeast Minnesota are vulnerable because of shallow bedrock, sinkholes and underground caves (referred to as karst geology), which lead to exchanges between surface and ground water resources.

County Partners:

Dodge, Fillmore, Goodhue, Houston, Mower, Olmsted, Rice, Wabasha, and Winona. The Olmstead Soil and Water Conservation District is the local partner contact.

Learn More

Nitrates in Minnesota drainage water

extension.umn.edu/agricultural-drainage/nitrates-minnesota-drainage-water

While artificial drainage offers tremendous benefits for crop production, it can also potentially transport nitrates from the soil to surface water. Here, we share strategies to help you avoid these nitrate losses, which can help protect the environment and reduce fertilizer costs.

Understanding nitrate loss

|

Nitrogen and its role

Nitrogen (N) is the atmosphere's single largest component and an important building block for all living organisms. It's found in many different forms in the soil depending on the nitrogen cycle.

It's taken up by crops in greater quantities than any other added nutrient. Grass crops, such as corn and wheat, require the addition of N-based fertilizers to maximize productivity. Legume crops, such as soybeans and alfalfa, don't require additional N inputs because they have the ability to fix N from the atmosphere in their root systems.

Overall, N used by crops for plant growth comes from fertilizer, soil organic matter, atmospheric deposition, animal manure and fixation (for legumes only).

Nitrate losses

Losses of nitrate, a mobile form of N, to water systems have been a concern for many years because of human health issues. When mammals—especially human infants under six months old—ingest nitrates, it interferes with the blood's ability to carry oxygen.

Standards

Thus, a standard of 10 parts per million (ppm) of nitrate-N has been established for drinking water by the Environmental Protection Agency. For decades, the primary focus has been on groundwater because of its connection with drinking water. Less attention has been given to nitrate levels in surface water, due to decreased dependence on surface water for drinking.

September 3, 2024
Clean Water Organizations Comments Exhibit 21

In addition, phosphorus is typically the limiting nutrient in Minnesota surface waters, rather than excess nitrate, that leads to increased plant and algae growth and significant surface water quality problems.

For decades, there hasn't been an established contaminant standard for nitrate-N in class 2 (aquatic life and recreation) waters in Minnesota. However, standards are currently under development and will be phased in over the next few years.

Scrutiny of agricultural drainage

Hypoxia in the Gulf of Mexico has led to increased scrutiny on nitrate contributions to surface waters from agricultural systems. Scrutiny has primarily focused on subsurface agricultural drainage, or tile drainage.

Tile drainage is a highly visible water pathway that transports nitrate from the landscape to surface waters. Other pathways of water movement from the landscape, such as leaching, shallow groundwater flow and surface runoff, are less visible and more difficult to sample and quantify.

Reducing nitrate in Minnesota surface waters

Figure 1: Artificial drainage isn't the only pathway of nitrate to surface waters, but it's the most easily seen and measured, and therefore under more scrutiny than other transport mechanisms.

The increased attention on the loss of nitrate via agricultural drainage has led many to call for significant changes to both N fertilizer management and agricultural drainage systems (Figure 1).

To make improvements, it’s essential to fully understand nitrate fluxes from agricultural systems in Minnesota, and how N management can affect losses. Plans to reduce nitrate in surface waters will need to account for inputs, set reduction goals and develop management strategies on both a watershed and an individual farm level.

Several conservation technologies have been developed, which reduce nitrate from surface waters after it’s already present. On this webpage, we look at the impact of managing N fertilizer inputs before it’s lost to surface water.

Corn is the most important crop in Minnesota in terms of total acreage and economic value. In addition, it’s the single largest user of N fertilizer on the state’s landscape. Most corn in Minnesota is either continuous (corn following corn), or in a rotation following soybeans.

Investigations on nitrate loss from Minnesota cropping systems have looked at all aspects of a crop rotation, but focused on corn for the aforementioned reasons.

Minnesota data

]Figure 2: The Southern Research and Outreach Center in Waseca established plots to collect drainage water in 1975. In 2009, the center automated data collection.

Research data on nitrate loss from cropping systems through drainage systems isn’t as common as you might think. In the early 1970s, the University of Minnesota Research and Outreach Centers (ROCs) in Waseca and Lamberton established plots for measuring drainage water quantity and quality (Figure 2).

Since then, they’ve examined many nitrogen management practices. These include N rate, application timing, source and the use of nitrification inhibitors. In addition, they’ve looked at various crops grown in rotation, tillage practices and mineralization of N from soil organic matter.

The drainage plots at the ROCs measure the total discharge of drainage water and the water’s nitrate concentration. Researchers use these numbers to calculate the total edge-of-field outflow of N via the drainage system.

Methods for presenting nitrate loss

Nitrate loss from tile drainage water varies greatly from year to year, primarily based on the total outflow of water from the tiles. In addition, research has shown that soil nitrate storage increased in the soil profile following dry years, but was then subject to loss during wet years.

This is why total nitrate-N loss is usually presented as either an average across years or a total amount over several years. Another method is to calculate nitrate concentration as a flow-weighted (FW) mean, which accounts for variability of total water flow from individual plots.

Annual nitrate loss

A literature review of a large number of worldwide drainage studies shows annual nitrate-N loss via tile lines varies from 0 to 124 pounds per acre. Plots kept devoid of vegetation (fallow) in Waseca measured an average annual loss of nearly 20 pounds of nitrate-N per acre from bare ground.

The source of this nitrate loss was N mineralized from organic matter. Corn grown without adding N fertilizer annually lost around 10 pounds of nitrate-N per acre. Loss rates for soybeans that received no N fertilizer were nearly identical (Table 1).

Generally, annual losses with row crops, where corn received near-optimum rates of N, ranged from 15 pounds of nitrate-N per acre (Table 1) on the low end in Waseca to 40 pounds per acre on the high end in Lamberton (Table 2) during four wet years. A separate project using larger plots at the Southern Research and Outreach Center (SROC) in Waseca located about a mile away confirmed annual losses ranging from approximately 10 to 18 pounds per acre.

The method shown to drastically reduce nitrate loss

In more than 40 years of drainage research at the ROCs, using perennial vegetation (as either native prairie plants or alfalfa) was the only method shown to drastically reduce nitrate loss at the Lamberton site.

Over a four-year period, these plots had an annual average flow-weighted nitrate concentration ranging from near zero to a high of 4 parts per million (ppm). In addition, because the total drainage volume greatly reduced, nitrate-N loss rates averaged only 1 to 1.5 pounds per acre (Table 2).

Table 1: Four-year nitrate-N loss in drainage water in Waseca

Crop rotation	N rate	N application timing	Nitrate-N concentration (four-year average)	Nitrate-N total (four-year average)
Corn-soybean-corn	0 lbs. per acre	--	6.1 ppm	37.7 lbs. per acre
"	60+40 lbs. per acre	Split	7.8 ppm	44.8 lbs. per acre
"	120 lbs. per acre	Preplant	8.2 ppm	52.1 lbs. per acre
Soybean-corn-corn	0 lbs. per acre	--	4.6 ppm	34.0 lbs. per acre
"	60+80 lbs. per acre	Split	7.9 ppm	64.2 lbs. per acre
"	160 lbs. per acre	Preplant	8.8 ppm	62.8 lbs. per acre
Corn-corn-soybean	0 lbs. per acre	--	5.5 ppm	30.5 lbs. per acre
"	0 lbs. per acre	--	8.4 ppm	40.9 lbs. per acre
"	0 lbs. per acre	--	8.7 ppm	38.3 lbs. per acre

Cropping system	Total discharge (four-year)	Nitrate-N: Concentration (four-year)	Nitrate-N: Total (four-year)
Continuous corn	30.4 inches	28 ppm	194 lbs. per acre

Cropping system	Total discharge (four-year)	Nitrate-N: Concentration (four-year)	Nitrate-N: Total (four-year)
Corn-soybean	35.5 inches	23 ppm	182 lbs. per acre
Soybean-corn	35.4 inches	22 ppm	180 lbs. per acre
Alfalfa	16.4 inches	1.6 ppm	6 lbs. per acre
Conservation Reserve Program (CRP)	25.2 inches	0.7 ppm	4 lbs. per acre

Influencing factors

The well-documented increase in the amount of artificial drainage in significant portions of Minnesota can be attributed to the practice's overall profitability, as well as the increased efficiency of farmers' time.

This has been accompanied by scrutiny about potential negative impacts, including nitrate loss. Minimizing nitrate loss via artificial drainage is in everyone's best interests, as it makes sense from both an environmental and economic standpoint.

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Figure 3: Corn grain yield and residual soil nitrate-N response as affected by fertilizer N rate on a Webster clay loam soil near Waseca, averaged from 2001 to 2003 (Source: Vetsch & Randall).

Crop response to fertilizer N rate generally follows a curve, where yield is maximized at some point and additional N inputs don't increase crop yield. The point where additional N inputs no longer produce an economic return is called the Economic Optimum N Rate (EONR).

Recommendations are based on EONRs from a large number of sites and years. Further examining the response curve relationship (Figure 3) shows how applying additional fertilizer N at or above the EONR results in little or no additional yield.

This is accompanied by greater accumulation of residual soil nitrate after harvest, which is susceptible to environmental loss. This relationship follows a similar curve but is inverse to the yield response to N. It shows the importance of N rate, as excessive N inputs are highly likely to be lost to the environment.

Fall fertilizer applications

Applying N fertilizer in the fall is a common practice in much of Minnesota. However, current BMPs don't recommend fall application in the southeastern part of the state, where there's very little artificial drainage.

Using urea as a fall fertilizer source is only recommended in the western part of the state, where annual precipitation averages less than 26 inches. A nitrification inhibitor is recommended with fall application of anhydrous ammonia (AA) in south-central Minnesota, where annual precipitation is around 35 inches.

A recent trend toward more continuous corn has resulted in less fall application of N. Most farmers find applying AA in the fall to be difficult due to the presence of corn residue from the previous year, especially with conservation tillage. A 2011 survey showed approximately 40 percent of N fertilizer was applied in the fall in southwestern, west-central and south-central Minnesota.

Research: Fall applications of AA with a nitrification inhibitor

Research has shown, on average, that fall applications of AA with a nitrification inhibitor (where recommended) have similar nitrate-N losses as spring applications. This, of course, varies from year to year based on climatic conditions. Mild falls and wet springs tend to increase nitrate loss.

Research showed that spring applications had greater corn yields than fall applications of AA with an inhibitor (Table 3). Increased yield (although not always statistically significant) is a likely indicator of decreased N loss into the environment.

Table 3: How applying N affects nitrate-N concentrations, losses and yield

N application: Rate	N application: Time	N application: N-Serve	Flow-weighted NO3-N concentration	Nitrate-N lost: Corn	Nitrate-N lost: Soybean	Nitrate-N lost: Total
80 lb/a	Fall	Yes	11.5 milligrams per liter (mg/L)	115 lb/a	90 lb/a	205 lb/a
120 lb/a	Fall	Yes	13.2 mg/L	121 lb/a	99 lb/a	220 lb/a
160 lb/a	Fall	Yes	18.1 mg/L	142 lb/a	139 lb/a	281 lb/a
120 lb/a	Spring	No	13.7 mg/L	121 lb/a	98 lb/a	219 lb/a

Best management practices: N fertilizers

The University of Minnesota established best management practices (BMPs) for applying N fertilizer in the early 1990s, which were updated in 2008.

These detailed guidelines are designed to help producers efficiently use N fertilizer to maximize profit, while minimizing N loss to the environment:

- How to apply nitrogen in Minnesota
- Southwestern and west-central Minnesota
- South-central Minnesota
- Northwestern Minnesota

- Southeastern Minnesota
- Irrigated potatoes
- Coarse-textured soils

Apply nitrogen at the right time

Figure 5: Ultimately, you may need technology and methods to reduce nitrate in surface waters. While you can fine-tune rates and timing, this is limited by time, climatic and crop growth constraints.

The N cycle dictates that conversion of the various forms of organic N must occur before nitrate becomes present in the soil. This conversion, caused by the actions of microorganisms, depends on temperature and time.

Nitrate's subsequent movement depends on the presence of water that exceeds field capacity. A growing crop's water demand lessens the likelihood of a drainage event. Optimum application timing also corresponds with the plant's need for N.

Guidelines

Applying N fertilizer would logically and ideally be as close as possible to when a plant needs the nutrient, to minimize the chance for loss into the environment. Best management practices dictate the minimum requirements to prevent excessive N loss (Figure 5).

You can lessen the chance of a significant leaching event by further delaying application to better correspond with planting or by split-applying so some of the application occurs to a growing crop.

However, take caution when late sidedress (in-season) applications are surface-applied and not incorporated. If meaningful rainfall doesn't occur for 10 to 20 days, you could lose this N to the atmosphere. In addition, it could become positionally unavailable to roots. In either case, yields will suffer due to lack of available N.

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Over-applying N fertilizers is another factor within the farmer's control. Generally, nitrogen loss through tile drainage increases as the N rate increases, especially at N rates greater than the economic optimum.

As illustrated in Figure 3, changing the N rate from 120 pounds per acre to 150 pounds per acre in corn following soybeans only increased yield by 4 bushels per acre. However, it increased the amount of residual N left in the soil profile by 40 percent, subjecting it to leaching.

Avoid applying nitrogen at rates higher than the EONR. It represents both an economic risk associated with higher-than-necessary fertilizer costs and a local environmental risk associated with potential losses. As the departure from EONR grows, so does the risk of nitrate loss to the environment.

Crop-specific fertilizer recommendations

A note on manure

Research conducted at the SROC found no differences in nitrate-N loss via agricultural drainage between manure and commercial fertilizer, provided recommended rates and application methods were used.

More on manure management

Nitrate reduction targets

The EPA has set a target for a long-term, 45 percent reduction of nitrates in the Mississippi River. Logically, following BMPs with respect to rate, source, timing and use of nitrification inhibitors is an important first step in reaching this goal.

Current rates of BMP adoption aren’t well-documented. Plus, model projections suggest further BMP adoption can only achieve modest improvements. Delaying applications until later in the season may achieve some reduction, but needs to be evaluated and account for the farmer’s ability to accomplish the application at the desired timing.

The recommendations we’ve shared here correspond with the national campaign for fertilizer applications to follow the 4Rs: The right fertilizer source, at the right rate, in the right place, at the right time.

The most effective strategy

In the end, our current cropping systems leak N and only perennial vegetation has been shown to effectively scour N from the soil profile. Note that while the environmental benefits of this practice are clear, an economic system to support these crops doesn’t exist. Therefore, the cost is high.

In the meantime, focus on making both economically and environmentally sound management decisions. These practices are easily within your control. Also, stay informed on new developments or practices that might achieve further reductions.

Water Testing for Nitrates handout (PDF)

Brad Carlson, Extension educator; Jeff Vetsch, researcher, Southern Research and Outreach Center and Gyles Randall, emeritus soil scientist, Southern Research and Outreach Center

Bierman, P., Rosen, C.J., Venterea, R., & Lamb, J.A. (2011). Survey of nitrogen fertilizer use on corn in Minnesota.

Carlson, B.M., & Ganske, L. (2012). A Minnesota farmer's guide to federal and state clean water law (University of Minnesota Extension publication #08680).

Fabrizzi, K., & Mulla, D. (2013). In Wall, Reducing cropland nitrogen losses to surface waters.

Kaiser, D.E., Lamb, J.A., & Eliason, R. (2011). Fertilizer Guidelines for Agronomic Crops in Minnesota (University of Minnesota Extension publication #06240-S).

Kaiser, D.E., Fernández, F., & Coulter, J.A. (2018). Fertilizing corn in Minnesota.

Fernández, F.G., & Kaiser, D.E. (2018). Understanding nitrogen in soils.

Lamb, J.A., Randall, G.W., Rehm, G., & Rosen, C.J. (2008). Best management practices for nitrogen use in Minnesota.

University of Minnesota Extension Manure management website

Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. (2008). Gulf Hypoxia Action Plan 2008 (Environmental Protection Agency publication #842K09001).

Randall, G.W., & Vetsch, J.A. (2011). Minimizing nitrate loss to drainage by optimizing N rate and timing for a C-C-S rotation.

Randall, G.W, Rehm, G., & Lamb, J.A. (2007). Best management practices for nitrogen use in southeastern Minnesota.

Randall, G.W., & Vetsch, J.A. (2005). Nitrate losses in subsurface drainage from a corn-soybean rotation as affected by fall and spring application of nitrogen and nitrapyrin. Journal of Environmental Quality, 34, 590-597.

Randall, G.W., & Vetsch, J.A. (2005). Corn production on a subsurface-drained mollisol as affected by fall versus spring application of nitrogen and nitrapyrin. Agronomy Journal, 97, 472-478.

Randall, G.W. (March 21-24, 2004). Subsurface drain flow characteristics during a 15-year period in Minnesota. In R. Cooke (Ed.), Proceedings of the Eighth International Symposium: Drainage VIII (pp. 17-24, ASAE publication #701P0304).

Randall, G.W., & Goss, M.J. (2008). Nitrate losses to surface water through subsurface, tile drainage. In J.L. Hatfield and R.F. Follett (Eds.), Nitrogen in the environment: Sources, problems, and management (pp. 145-175). Elsevier Sciences B.V.

Randall, G.W., & Mulla, D.J. (2001). Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices. Journal of Environmental Quality, 30, 337-344.

Randall, G.W., Iragavarapu, T.K., & Schmitt, M.A. (2000). Nutrient losses in subsurface drainage water from dairy manure and urea applied for corn. Journal of Environmental Quality, 29, 1244-1252.

Randall, G.W., Huggins, D.R., Russelle, M.P., Fuchs, D.J., Nelson, W.W., & Anderson, J.L. (1997). Nitrate losses through subsurface tile drainage in Conservation Reserve Program, alfalfa, and row crop systems. Journal of Environmental Quality, 26, 1240-1247.

Randall, G.W., personal communication, 2013.

Sands, G.R. (2018). Impact of agricultural drainage in Minnesota.

Sands, G.R., Song, I., Busman, L.M., & Hansen, B.J. (2008). The effects of subsurface drainage depth and intensity on nitrate loads in the northern Cornbelt. Transactions of the American Society of Agricultural and Biological Engineers, 51(3), 937-946.

U.S. Department of Agriculture National Agricultural Statistics Service Field Office (2013). 2012 Minnesota Agricultural Statistics.

Nitrogen | Minnesota Pollution Control Agency

pca.state.mn.us/pollutants-and-contaminants/nitrogen

Nitrogen, like phosphorus, is a nutrient that pollutes in state waters, and its concentration in many rivers has been increasing from historic natural levels over time due to human influences.

Statewide, data on nitrate concentrations in rivers over the past 20 years show a mixed bag:

- Many monitoring sites with variable levels and no trend
- Many sites where levels have increased
- Some sites where levels have decreased

Sources

More than 70% of the nitrate in Minnesota waters is coming from cropland, the rest from regulated sources such as wastewater treatment plants, septic and urban runoff, forests, and the atmosphere. Nitrate leaching into groundwater below cropped fields and moving underground until it reaches streams contributes an estimated 30% of nitrate to surface waters. Groundwater nitrate can take from hours to decades to reach surface waters.

Cropland sources account for an estimated 89% to 95% of the nitrate load in the Minnesota, Missouri, and Cedar Rivers, and Lower Mississippi River basins.

Tile drainage pathway

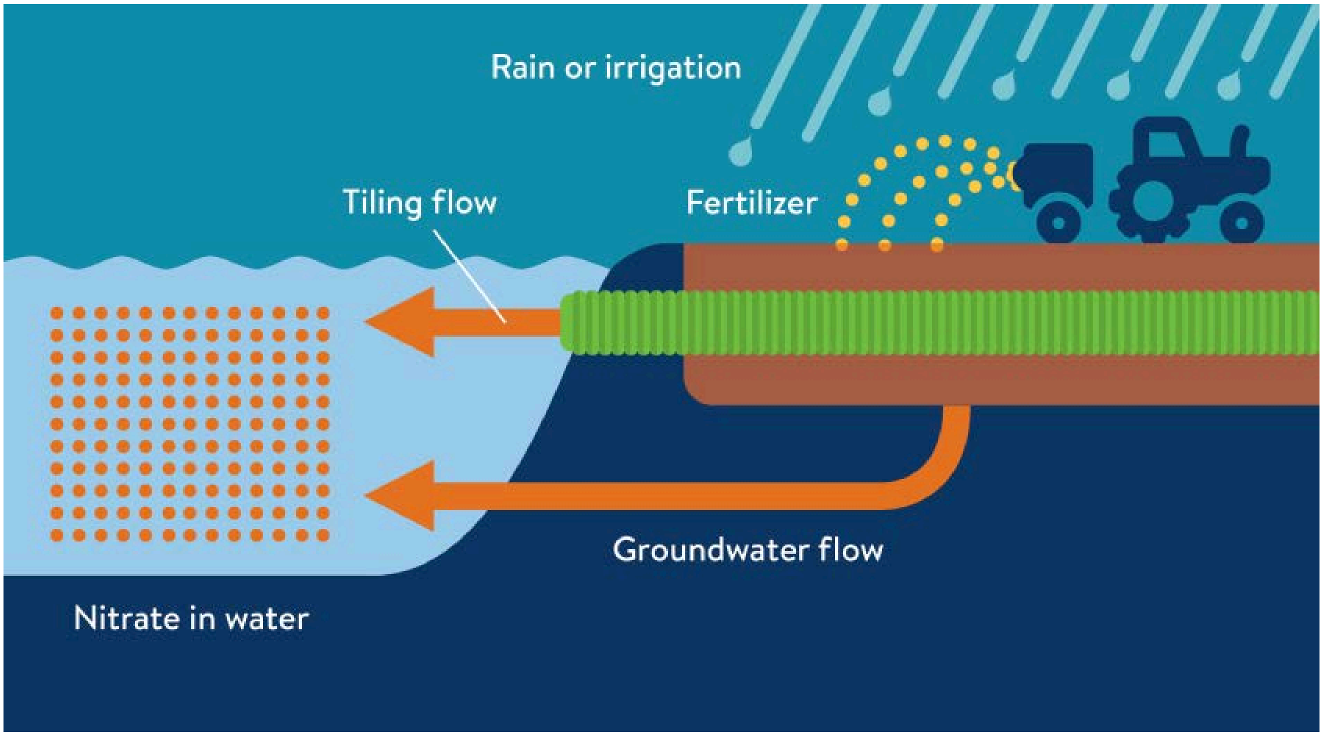
In tiled cropland, most of the rainwater that ends up in surface water (ditches, streams) flows through tile drainage. This water can be high in nitrate, but it is also potentially easier to control.

Groundwater pathway

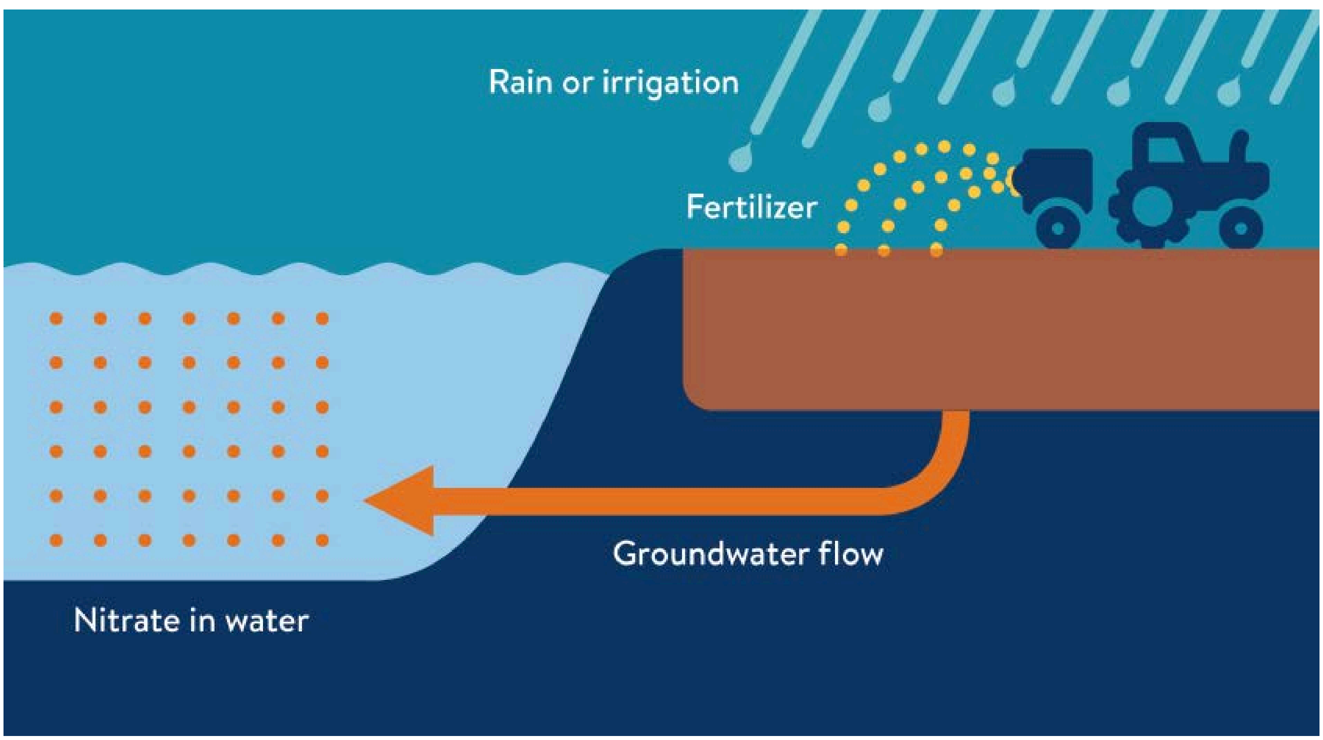
In cropland without tile drainage, most rainwater flows through the ground to get to surface waters. As it travels through the earth, some of the nitrate is removed, resulting in less nitrate reaching our streams and rivers. However, there are fewer options of controlling this kind of nitrate pollution once it moves below the crop roots.

September 3, 2024
Clean Water Organizations Comments Exhibit 22

Crop drainage with tiling



Crop drainage without tiling



Human health and environmental concerns

Nitrate (a form of nitrogen) in lakes, rivers, and streams is toxic to fish and other aquatic life; in drinking water, it's potentially harmful to humans. Proposed reductions in nitrogen will benefit both Minnesota waters and water downstream from us.

Ammonia is a form of nitrogen that is directly toxic to aquatic life. It comes from wastewater treatment plants and animal waste or air pollution and runoff from agricultural land. Water with high concentrations of ammonia allow the chemical to build up in the tissues and blood of fish, and can kill them.

Nitrate in the Mississippi River

On average, 158 million pounds of nitrate leaves Minnesota per year in the Mississippi River — 75% comes from Minnesota watersheds.

Nitrate leaving Minnesota via the Mississippi River contributes to the oxygen-depleted dead zone in the Gulf of Mexico. The dead zone cannot support aquatic life, affecting commercial and recreational fishing and the overall health of the Gulf. Nitrate concentrations have steadily increased in the Mississippi River since the mid-1970s.

Monitoring, reporting, and regulations

The MPCA's research shows elevated nitrate levels in water, particularly in the southern third of Minnesota.

Reducing nitrate

Tactics for reducing cropland nitrate that reaches surface waters fall into three categories:

- **Manage in-field nutrients** – Optimize fertilizer rates, apply fertilizer closer to timing of crop use
- **Manage and treat tile drainage water** – Plan tile spacing and depth, control drainage, construct and restore wetlands for treatment purposes, use bioreactors
- **Diversify vegetation/landscape** – Plant cover crops, plant more perennials on marginal cropland

Nitrate fertilizer efficiency is improving and further refinements in fertilizer rates and application timing could reduce nitrate loads by roughly 13% statewide. But additional and more costly practices will also be needed to make further reductions and meet downstream needs. Statewide reductions of more than 30% are not realistic with current practices.

Bigger reductions would require limiting nitrate leaching across large parts of southern Minnesota, particularly on tile-drained fields and row crops over thin or sandy soils. Only collective incremental changes by many over broad acreages will result in significant nitrogen reductions to downstream waters.

The Department of Agriculture's Nitrogen Fertilizer Management Plan is the state's blueprint for prevention or minimization of the impacts of nitrogen fertilizer on groundwater.

Contact

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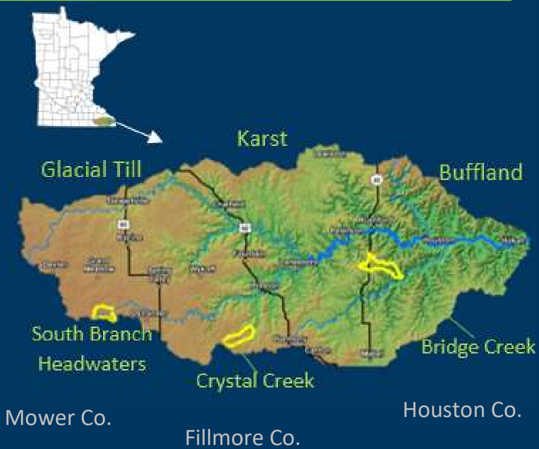
Nitrogen Rates



Field to Stream Partnership

The Root River Field to Stream Partnership (RRFSP) is a multi-organizational effort to evaluate agricultural practices and water quality at multiple scales and landscape settings. The strategic selection of these study watersheds allows the findings to be applied to similar areas across southeastern Minnesota.

On-Farm Nitrogen Rate and Timing

The relationship between corn yield, nitrogen rate and timing was studied over a seven-year period in southeast MN. Results across four different counties from 2015-2021 (24 site years) are summarized.

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August 2022

What's the Best Nitrogen Rate?

- A total of twenty-four nitrogen (N) rate and timing experiments were conducted on corn fields in southeast Minnesota from 2015 through 2021.
- Ten treatments were replicated four times in a randomized, complete-block design. Seven of the ten treatments were N rates applied at planting and three treatments were split applied.

Plot harvest near Grand Meadow in Mower County

- On-farm studies were conducted near the city of Grand Meadow in Mower County, Harmony in Fillmore County, Utica in Winona County and Elgin in Wabasha County. Most plots were located and repeated on the same farm.
- The **M**aximum **R**eturn **T**o **N**itrogen (MRTN) is the nitrogen (N) rate that maximizes return on investment. The MRTN is a data driven, economically and environmentally sound method for making N rate decisions and is a recommended best management practice (BMP) when fertilizing corn in Minnesota.
- The University of Minnesota updated the corn nitrogen fertilizer guidelines in 2022 and are summarized in Table 1. Using the most common N price to corn price ratio of 0.10, the acceptable range of nitrogen to apply is **130-150 lb N/ac** when corn follows soybeans and **160-190 lb N/ac** when corn follows corn. Total nitrogen applied should include credits from other fertilizers containing nitrogen such as MAP, DAP, AMS, starter and nitrogen credits from alfalfa and manure.
- The Corn Nitrogen Rate Calculator can be used identify the most profitable N rates using different nitrogen and corn prices. http://cnrc.agron.iastate.edu/

University Nitrogen Rate Guidelines for Corn

Previous Crop	N Price/Corn Price Ratio	MRTN	Acceptable Range
		-----------lb N/acre----------	
Corn (71 sites)	0.075	190	170-205
	0.100	**175**	**160-190**
	0.125	165	150-175
	0.150	155	145-165
Soybeans (165 sites)	0.075	150	135-165
	0.100	**140**	**130-150**
	0.125	135	125-145
	0.150	130	120-140

Table 1. Nitrogen fertilizer rate recommendations for non-irrigated corn in Minnesota. The most common nitrogen price to corn price ratio, 0.10, is highlighted. A $0.50 nitrogen price and $5.00/bu corn price equates to a 0.10 ratio. Source Aug 2022: *https://extension.umn.edu/crop-specific-needs/fertilizing-corn-minnesota.*

Results

Corn following Soybean

- A total of 13 corn fields were studied over a seven-year period. Most fields were located on well drained silt loam soils in Fillmore, Winona and Wabasha counties. Two sites were located in Mower County on poorly drained soils that contained subsurface drainage tile and high organic matter.
- Figure 1 shows the best rate of nitrogen (N) to apply on sites with well drained soils was 129 lb N/ac with an exceptional corn yield of 249 bu/ac.
- Figure 2 shows the response at a poorly drained site located south of Grand Meadow (GM south). This farm typically responded to more preplant nitrogen and required over 70 lb N/ac more preplant N when compared to well drained sites. The best preplant nitrogen rate at GM south was 202 lb N/ac with a yield 229 bu/ac.
- The zero-rate check produced over 150 bu/ac corn yield in plots with well drained soils while the poorly drained GM south site typically produced 40 bu/ac less yield. This could indicate that less N was supplied by the soil through mineralization.
- Even with drain tile, a natural dense layer of glacial till located at depths below one foot at the GM south site creates anaerobic conditions which likely results in more frequent N loss through de-nitrification and less soil N contributions from mineralization. This dense subsoil could also be affecting corn rooting depth.

Corn following Corn

- A total of 11 different fields were studied. Fields were located in Fillmore, Winona and Wabasha Counties on well drained silt loam soils.
- Across all plots and years, the best preplant rate to apply was 175 lb N/ac with a yield of 223 bu/ac (Figure 3).

Split Applied Nitrogen

- When N was split applied, corn yields were significantly higher at 5 of the 24 sites (21%) when compared to fields that received all N at preplant.
- At the poorly drained Grand Meadow South site, split N application rates were occasionally more profitable and required less N.
- Starting in 2022, enhancements to this study will provide new and better insights to MRTN values for split applied N applications.

Residual Soil Nitrate (RSN)

- Figure 4 shows the relationship between RSN and nitrogen rates above or below the MRTN. RSN samples were collected to a depth of four feet after harvest. Elevated RSN can increase the risk for nitrate movement to groundwater and surface water.
- RSN rarely exceeded 60 lb N/ac when rates were applied near the MRTN (within +- 25 lb N/ac). When N rates were applied above the MRTN (right side of the vertical line), the amount of RSN increased rapidly.

Summary

- When averaged across similar sites, the MRTN was consistent with University N rate guidelines for sites with well drained soils, but typically underestimated preplant N needs for a poorly drained site in Mower County. Continuation of this study will provide valuable information for growers and crop advisors that is current and specific to southeast Minnesota.

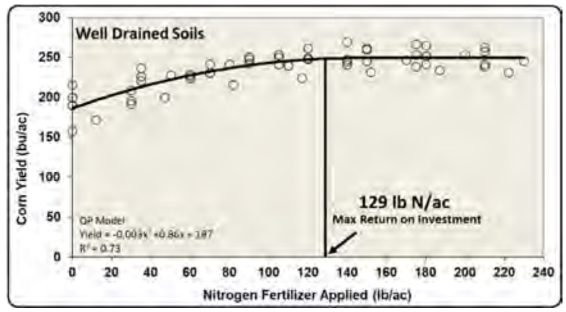

Figure 1. Corn following soybean yield as affected by nitrogen rate on well drained soils from 2015-2021 (8 site years).

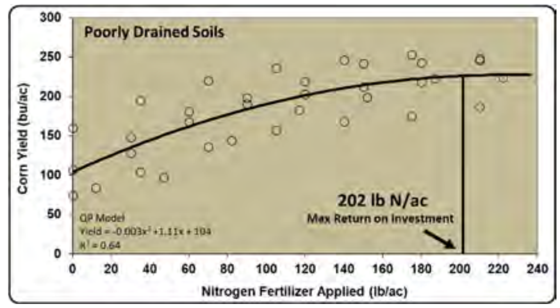

Figure 2. Corn following soybean yield as affected by nitrogen rate on a poorly drained site south of Grand Meadow (GM south) from 2017-2021 (5 site years).

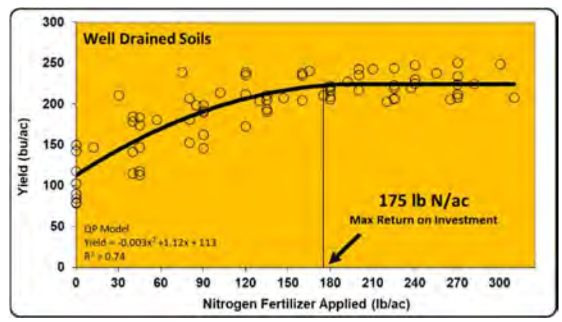

Figure 3. Corn following corn yield as affected by nitrogen rate on well drained soils from 2015-2021 (11 site years).

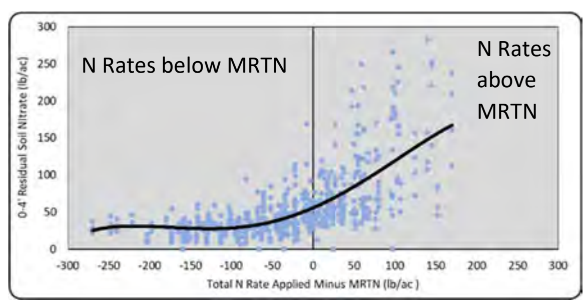

Figure 4. Relationship between residual soil nitrate and N rates above or below the MRTN from 2015-2021 (24 site years).

Root River Field to Stream Partnership

Minnesota Department of Agriculture
Minnesota Agricultural Water Resource Center
The Nature Conservancy

Mower SWCD
Fillmore SWCD
Root River SWCD

September 24, 2024
Clean Water Organizations Exhibit 23

FIELD RUNOFF

Root River Field to Stream Partnership

PRIMARY PROJECT GOAL

Determine the range of sediment and nutrient losses associated with runoff from representative farming systems and small watersheds in southeastern Minnesota.

Status:

Data collected from four fields, collected over seven years (2010–2018).

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DEPARTMENT OF AGRICULTURE

April 2019

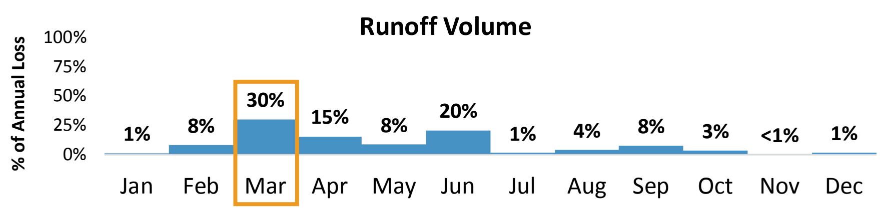
WHERE DOES THE WATER GO?

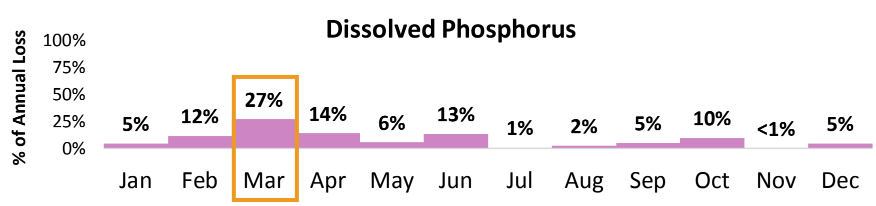
On average, 36 inches of precipitation was received annually. During the study, 7% of this total was measured as field surface runoff with a range of less than 1% in a dry year and up to 24% during a very wet year. How we manage this runoff can make a big difference for clean water.

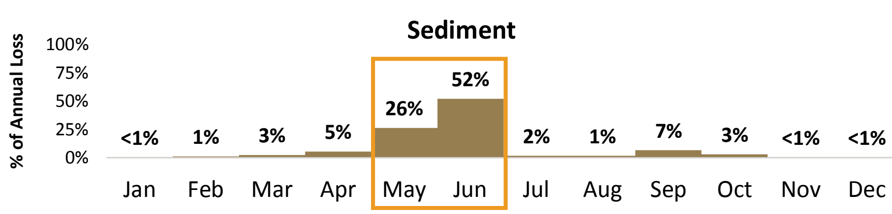
- **On average, 40% of the total runoff volume occurred when the soil was frozen.**
- Over 50% of the annual nutrient and sediment losses typically occurred during 1-2 rain events each year.

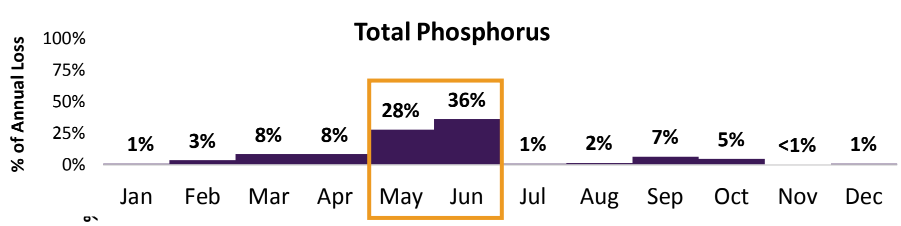
High Risk Periods

Sediment and nutrient losses peak at varying times of the year. Understanding these risk periods is key to reducing loss.









- Dissolved phosphorus losses were highest in March and often occur when the ground is frozen. **Incorporation of fertilizer and proper management of soil test phosphorus levels will help reduce these losses.**
- Nearly 80% of the sediment loss occurred during May and June. Total phosphorus loss is closely linked to soil loss. **Good soil conservation practices will help reduce these losses.**

September 24, 2024
Clean Water Organizations Exhibit 23

Precipitation & Runoff

- Precipitation averaged 4% above normal during the study period with a mix of dry, normal and wet conditions.
- Field runoff averaged 2.7 inches (7% of annual precip.) with 40% occurring during frozen soil conditions.
- Field surface runoff has been observed in every month averaging 20 runoff events each year. Runoff does not occur every time it rains.

Field Sediment Loss

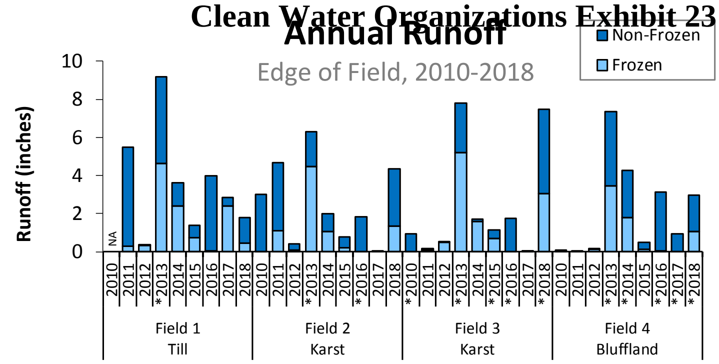
- **Average sediment loss:** 1,461 lb/ac. (0.7 tons/ac.) Range: <1 to 8,969 lb/ac.
- **Sustainable soil loss:** < 1,000 lb/ac./year If erosion is visible, losses likely exceed this.
- 78% of annual loss occurred during select storms in May & June. During this critical time, fields were prepared for planting, but not at full canopy.

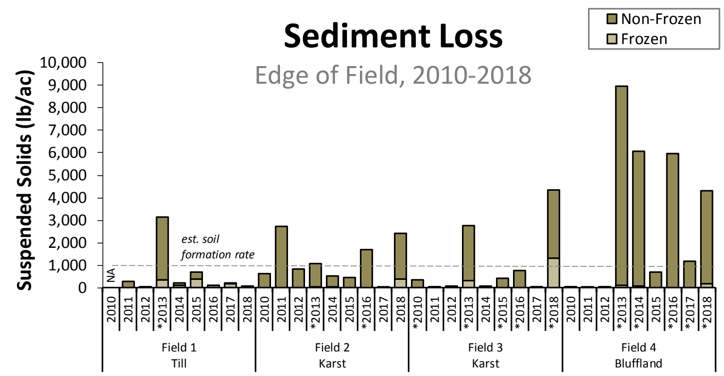
Field Phosphorus Loss

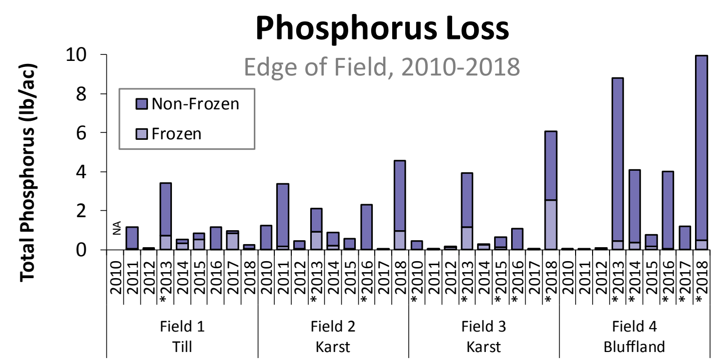
- **Average total phosphorus (P) loss:** 1.9 lb/ac. Range: <0.1 to 10.0 lb/ac.
- **Dissolve P (not attached to sediment):** Accounts for 16% of total P loss (44% of this loss occurs when the ground is frozen).
- **Particulate P (attached to sediment):** 64% of loss occurred in May & June.
- For every 1,000 lb/ac. of sediment loss about 1.0 lb/ac. of P is lost. Goal is to keep this loss to less than 1.0 lb/ac./yr.

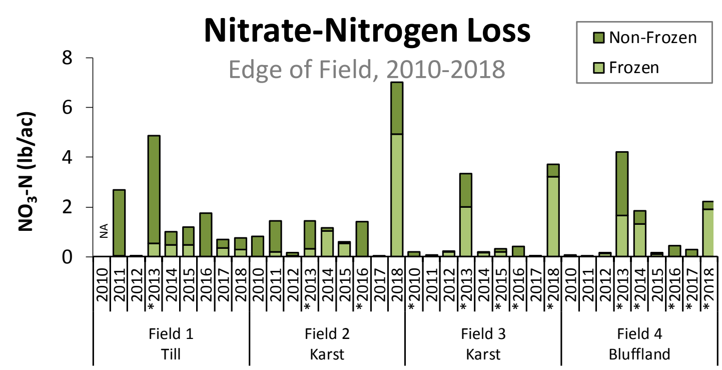
Field Nitrogen Loss

- **Average Total Nitrogen (TN) loss:** 9.8 lb/ac. (includes organic form of N) if substantial soil loss occurs, TN in surface runoff can exceed 37 lb/ac.
  - **Nitrate-N form:** 17% of TN Range: <0.1 to 4.9 lb/ac. **Surface** average runoff loss: 1.6 lb/ac. **Sub-surface** average tile loss: 41 lb/ac., max 63 lb/ac.
- **Surface Runoff:** Total nitrogen transported in surface runoff can be controlled through soil conservation.
- **Sub-Surface Leaching:** Most nitrogen is lost this way and is detected as nitrate-nitrogen in tile drainage, springs, streams, rivers, and groundwater.









*Loss was underestimated during overtop events

Reducing nitrate leaching losses will be challenging, but it is a very important task. Fine-tuning nitrogen rates, split applying nitrogen, crediting legumes and manure, growing perennials, and using cover crops are important practices.

Guidelines for manure application rates

extension.umn.edu/manure-management/manure-application-rates

Quick facts

- Manure nutrient management planning is important for maximizing crop productivity while protecting water quality
- Guidelines for manure application rates vary depending on crop and cropping history
- Manure application rates should consider all nutrient sources that will be or have been applied to a field. For example, if commercial, inorganic fertilizers will be applied or if manure was applied in the previous two years, take credit!

Credit: MPCA

September 3, 2024
Clean Water Organizations Comments Exhibit 24

Animal manure is a good source of nutrients for crops, including nitrogen (N), phosphorus (P), and potassium (K). The proportion of the nutrients in manure are typically not the same as needed by the crops, however. Manure application based on one nutrient may over- or under apply other required crop nutrients. Nitrogen is required in the largest quantities by non-legume crops. Applying manure to meet crop N needs will likely overapply P, and possibly K, for a crop such as corn. On the other hand, using manure to meet P needs of the crop will likely result in a lower application rate and will underapply N and possibly K. Commercial fertilizers will then be needed to balance out N and K needs. Consider the pros and cons of these two options when choosing a manure application rate.

Nutrients in manure are not 100% available in the first year. First-year plant-available N (PAN) will depend on animal species and how the manure is applied. Plant-available P (PAP) is assumed to be 80% of the total P applied in the first year. You can learn more about calculating PAN and PAP, including first-year PAN and PAP, from our “calculating manure application rates” recommendations. The guidelines for manure application rates below are based on PAN or PAP, not total N and P.

Nitrogen guidelines for manure

The rates below are the maximum amounts of N that should be applied when manure is used, whether it is all manure or a combination of manure and inorganic commercial fertilizers. Lower rates may be considered based on the productivity of the soils in your fields, economics, or environmental concerns. In all cases, all sources of N should be taken into consideration when estimating how much N to apply, including:

- N from irrigation water.
- Credits from manure, or other organic N sources, that was applied in the past 2 years.
- Credits from legumes like edible beans, red clover, etc.

Why is that? Research across the US Midwest has shown that applications of N above the economically optimum N rate (EONR) for a crop significantly increase the potential for N losses. For example, once N leaches past the plant root zone into the ground water, it becomes a concern for drinking water and will eventually end up in lakes, rivers, and streams. On the other hand, excess N that is not taken up by crops can also be lost as a gas through denitrification. When manure N becomes plant available, it behaves exactly the same in the environment as N from commercial inorganic fertilizer, so it is important that all forms of N applied to the soil are taken into consideration. Don’t waste your manure!

|

The maximum rate of plant-available nitrogen (PAN) that should be applied with manure to non-irrigated corn, depending on the crops prior to corn, can be found in the table below.

Nitrogen recommendation for non-irrigated corn

Crop prior to corn	Crop 2 years prior to corn	Maximum lbs of PAN to apply
Corn (or other non-legume crop)	Not applicable	195
Corn	Alfalfa (1-year-old stand)	120
Corn	Alfalfa (>2-year-old stand)	80
Soybean	Not applicable	150
Alfalfa (1-year-old stand)	Not applicable	80
Alfalfa (>2-year-old stand)	Not applicable	40

Corn grown under irrigation is a special case because it is usually done on coarse-textured (or sandy) soils. Under these conditions, there is a higher risk of N loss due to the high leaching potential of these types of soils. With manure, there are other nutrients to consider that could potentially also be lost through leaching. Because of this, we suggest applying a lower rate of manure (as an example, see the section on “Phosphorus Guidelines for Manure” below), then supplement with commercial N fertilizers to meet total N needs. See the table below for the total N rate guidelines.

A good rule of thumb is to apply a lower rate of manure (195 lbs of plant-available N [PAN] or lower), then add the remaining N as commercial fertilizer.

Nitrogen recommendation for irrigated corn

Crop prior to corn	Pounds of nitrogen to apply
Soybean	205
Other crops	235

The maximum amount of PAN you should apply to non-legume crops with manure should follow University of Minnesota guidelines for nitrogen fertilizers.

Details for each crop:

- Barley
- Buckwheat
- Canola
- Grasses
- Grass-legume mixtures
- Oat
- Potato (irrigated)
- Rye
- Sugarbeet
- Sunflower
- Wheat

If manure is applied to a legume crop, you can apply as much PAN as the crop will likely take up in the harvested portion. You can find out how much N will be taken up per harvestable unit in the table below Multiply this number by the amount of yield you expect from that field to get your application rate.

Amount of nitrogen removed per unit of harvested yield

Crop	Yield unit	Crop N removal (lbs per yield unit)
Alfalfa	Tons (air dry)	50.4
Red clover	Tons (air dry)	45.1

Crop	Yield unit	Crop N removal (lbs per yield unit)
Soybean	Bushels	3.5

Example: Assume that field conditions have been poor, so you need to apply manure in the fall to a field where soybean will be planted the following spring because it is the only dry field you have. You expect the soybean yield to be about 60 bushels per acre (this is the typical yield you get from this field). If you multiply 60 bushels per acre by 3.5 lbs of N per bushel, you will find that the crop will take up 210 lbs of N. This means you can apply a manure rate of 210 lbs of PAN per acre.

Phosphorus guidelines for manure

In cases where manure is readily available frequently, using a P-based manure application rate may make the most long-term, economic sense because the crops will use nutrients more efficiently. For manure, it is recommended to apply as much plant-available phosphorus (PAP) as the crop will use.

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Where do our guidelines come from?

Inorganic commercial fertilizers are often used to figure out crop nutrient needs in experiments across Minnesota. These fertilizers are designed to release 100% of the N and P in the first year, so it makes it easier to determine how much to apply to get the optimized yields. As an example, N guidelines for corn are based on 170+ experiments across the state, most of which occurred in the past five years. As new experiments are completed, the data on optimal N needed are added to the overall database, and N guidelines are adjusted accordingly.

With manure, we can calculate the estimated plant-available nutrients that will be available in a given year. Once nutrients from manure are plant-available, they behave in the environment exactly the same as a nutrient from a commercial fertilizer. Thus, our guidelines for manure application are based on optimal nutrient rates needed, which is known from fertilizer experiments, and how much plant-available nutrient will be available in the first year after application.

Guidelines for Manure Application Rates (printable PDF, 2022)

Melissa Wilson, Extension manure management specialist
Reviewed in 2022

Fertilizing corn in Minnesota

extension.umn.edu/crop-specific-needs/fertilizing-corn-minnesota

Nitrogen guidelines

Minnesota corn growers receive a substantial return for money invested in nitrogen (N) fertilizers. For many situations, the most profitable yield cannot be achieved unless N fertilizers are used.

There are many management decisions involved in the use of N fertilizers. The most important decision is the selection of an N rate that will produce maximum profit while limiting the potential for environmental degradation. The choice of an appropriate rate of fertilizer N is not easy because of the transient nature of N in soils.

|

The consideration of soil productivity, price/value ratio and previous crop are used to arrive at the fertilizer N guidelines for corn. This represents a significant change compared to previous approaches. This process has been in place since 2005 and is the product of a multi-state effort to use a similar philosophy/approach for determining N rate guidelines for corn.

Because of technology improvements in corn production practices such as weed and pest control, expected yield is not as important of a factor in determining N rate as it has been in the past.

Soil productivity has become a better indicator of N need. A majority of Minnesota soils are highly productive and have generally produced maximum economic corn yield with similar N rates over the last 15 years.

Some soils have a reduced yield potential due to erosion, reduced water holding capacity caused by lower organic matter content, sandy soil texture, poor drainage and restricted root growth. The fluctuation in fertilizer price affects the economic optimum N rate. To account for this change, the ratio of the price of N per pound to the value of a bushel of corn has been added to the N rate decision.

An example calculation of the price/value ratio is if N fertilizer costs $0.40 per lb N (or $656 per ton of anhydrous ammonia), and corn is valued at $4.00 per bushel, the ratio would be 0.40/4.00 = 0.10.

The maximum return to N value (MRTN) shown in Table 1 is the N rate that maximizes profit to the producer based on the large number of Minnesota experiments supporting these guidelines. Once the soil productivity and price/value ratio have been determined, a producer's attitude towards risk must be factored into the process.

A producer who is risk-averse and cannot tolerate risk associated with less-than-maximum yields in some years, even though economic return to N may not always be highly profitable, may want to use the N rates near the high end of the acceptable range shown in Table 1.

On the other hand, if corn is grown on medium or fine-textured soils considered to be of low or medium productivity and/or localized N response data support lower N rates, producers may choose N rates near the low end of the acceptable range in Table 1 if they are willing to accept the possibility of less-than-maximum yield in some years without sacrificing profit.

The acceptable range gives the producer flexibility in arriving at an acceptable and profitable N rate that is calculated as the rate +/- $1 from the MRTN rate.

Table 1: Guidelines for use of nitrogen fertilizer for corn grown following corn or soybean when supplemental irrigation is not used

Prior crop	N price/Crop value ratio	MRTN	Acceptable range
Corn	--	lbs N/acre	lbs N/acre

Prior crop	N price/Crop value ratio	MRTN	Acceptable range
	0.075	190	170-205
	0.100	175	160-190
	0.125	165	150-175
	0.150	155	145-165
Soybeans	--	lbs N/acre	lbs N/acre
	0.075	150	135-165
	0.100	140	130-150
	0.125	135	125-145
	0.150	130	120-140

The N rate guidelines in Table 1 are used if corn is grown in rotation with soybean or following corn when NOT irrigated. Corn grown on sandy soils deserves special consideration.

If irrigated, the guidelines listed in Table 2 are appropriate when corn is grown in rotation with corn. If corn is grown following soybean on irrigated sandy soils, a credit of 30 lbs of N per acre should be taken from the suggestions given in Table 2.

Table 2: Guidelines for use of N fertilizer for corn following corn when grown on irrigated sandy soils

N price/Crop value ratio	MRTN	Acceptable range
0.05	235 (lbs N/acre)	210-255 (lbs N/acre)
0.1	210	190-225
0.15	190	175-210

N price/Crop value ratio	MRTN	Acceptable range
0.2	180	165-190

For non-irrigated corn grown on soils with a loamy fine sand texture and less than 3% organic matter, use the guidelines provided in Table 3.

Soils considered medium productivity in the past were given special consideration. More recent data has not shown strong support for a separate suggested application rate of N for medium-productivity soils.

The rate of N can be adjusted based on the acceptable range if a soil is considered to be medium productivity and has shown to be more or less responsive to fertilizer N.

Table 3: Nitrogen guidelines for corn grown on non-irrigated loamy fine sands with less than 3% organic matter

N price/Crop value ratio	Corn/Corn	Soybean/Corn
0.05	100 (lbs N/acre)	70 (lbs N/acre)
0.1	90	60
0.15	80	50
0.2	70	40

Alfalfa, which includes pure stands of alfalfa and alfalfa-grass mixtures with at least 50% alfalfa in the stand, can eliminate or greatly reduce the need for N from fertilizer or manure during the two subsequent years if corn is grown.

Past guidelines assigned N credits to corn based on alfalfa stand density, but analyses of field trials from across Minnesota and the Midwest indicate that the frequency and level of yield response to N in first and second-year corn following alfalfa are more closely associated with soil texture, age of alfalfa at termination, alfalfa termination timing and weather conditions.

It is well established that first-year corn following alfalfa rarely responds to N except on sandy soils, on fine-textured soils when there are prolonged wet early-season conditions and on medium-textured soils when following very young alfalfa stands or in some cases when following spring-terminated alfalfa.

In past field trials from across Minnesota and the Midwest, yield of second-year corn following alfalfa did not respond to N in half of the fields studied.

Suggested rates of N for first and second-year corn following alfalfa are in Table 4. In some cases, the optimal rate of N can vary greatly due to weather-related variability in soil N mineralization. In such cases, limit the amount of N from fertilizer and manure applied before and near corn planting and apply additional N to corn during the growing season if necessary based on weather and crop conditions.

Table 4: Nitrogen suggestions for first and second-year corn following alfalfa^a

Soil texture[b]	Irrigated or non-irrigated	Alfalfa age[c]	Alfalfa termination time	First-year corn following alfalfa	Second-year corn following alfalfa
Coarse	Irrigated	1 year	Fall or spring	140-170 (lbs N/acre)	140-170? (lbs N/acre)
Coarse	Irrigated	2 or more years	Fall or spring	70-150	70-150
Coarse	Non-irrigated	1 year	Fall or spring	40-80[d]	80-120[d]
Coarse	Non-irrigated	2 or more years	Fall or spring	0-20	0-80
Medium	Both	1 year	Fall or spring	40-80[d]	80-120[d]
Medium	Both	2 or more years	Fall	0-20	0-80

Soil texture[b]	Irrigated or non-irrigated	Alfalfa age[c]	Alfalfa termination time	First-year corn following alfalfa	Second-year corn following alfalfa
Medium	Both	2 or more years	Spring	0-40	0-80
Fine	Both	1 year	Fall or spring	40-80[d]	80-120[d]
Fine	Both	2 or more years	Fall	0-20[d]	0-80[d]
Fine	Both	2 or more years	Spring	0-40[d]	0-80[d]

[a] Includes pure stands of alfalfa and alfalfa-grass mixtures with at least 50% alfalfa in the stand.
[b] Coarse = sands and sandy loams; medium = loams and silt loams; fine = clays, clay loams and silty clay loams.
[c] Alfalfa age at termination, including the establishment year if alfalfa was direct-seeded without a small grain companion crop.
[d] An additional 30 to 40 lbs N/acre can be applied to corn during the growing season if necessary based on the Corn calculator for supplemental nitrogen.

To arrive at a guideline following other crops, an adjustment (credit) is made to the corn following corn guidelines. The adjustments can be found in Table 5.

In Table 5, several crops are divided into Group 1 and Group 2. The crops for each group are listed in Table 6.

Table 5: Nitrogen credits for different previous crops for first-year corn

Previous crop	1st year N credit
Group 1 crops	75 (lbs N/acre)
Group 2 crops	0
Edible beans	20
Field peas	20

The N rates listed in Tables 1 and 2 define the total amount of fertilizer N that should be applied. All N applied should be accounted for in the calculation, including N in starter fertilizer, weed and feed program, DAP (di-ammonium phosphate) or MAP (mono-ammonium phosphate) applied late fall (after 4” average soil temperatures stabilize at 50^{o}F) on non-sandy soils or for all soil types in spring, and with sulfur.

It is generally accepted that legume crops provide N to the next crop in the rotation. Some forage legumes provide some N in the second year after the legume was grown.

Red clover is the only crop other than alfalfa that may provide a second-year N credit. If red clover was grown two years before the current crop, 35 lbs of N per acre should be subtracted from the N rate when corn follows the crops listed in Group 2, Table 5.

Table 6: Crops in Group 1 and Group 2

Crop	Group number
Alsike clover	1
Birdsfoot trefoil	1
Grass/legume hay	1
Grass/legume pasture	1
Fallow	1

Crop	Group number
Red clover	1
Barley	2
Buckwheat	2
Canola	2
Corn	2
Grass hay	2
Grass pasture	2
Oats	2
Potatoes	2
Rye	2
Sorghum-sudan	2
Sugar beet	2
Sunflower	2
Sweet corn	2
Vegetables	2
Wheat	2

The use of manure as a fertilizer source can raise questions about adequate nitrogen rates. The economics of manure application are not straightforward when on-farm sources are used in corn production.

Manure presents challenges as not all of the nutrients are 100% available to crops in the first year of application. Plant available N (PAN) is a term used when applying manure to identify the amount of N applied that is plant available in any given year and may be less than the total N applied.

Suggestions for N application when manure is the primary nutrient source are given in Table 7. If commercial fertilizer is used along with manure, the suggested rates in Table 7 should not be exceeded. Lower application rates similar to the 0.10 price ratio may be considered based on the productivity of the soils in your fields, economics or environmental concerns.

Table 7: Nitrogen suggestions for corn when manure is used as a fertilizer source

Crop grown prior to corn	Crop 2 years prior to corn	Field irrigated?	Suggested PAN to apply (lbs N/acre)
Corn		No	195
Corn		Yes	235
Corn	Alfalfa (1 year old stand)	No	
Corn	Alfalfa (2 or more year old stand)	No	80
Soybean		No	150
Soybean		Yes	205
Alfalfa (1 year old stand)		No	80
Alfalfa (2 or more year old stand)		No	40

The pre-plant soil nitrate test (PPNT) can be a useful tool for assessing situations where residual soil nitrate can be credited to the corn crop. The PPNT should not be used when commercial fertilizer or manure was applied in the previous fall or in the spring prior to the sample being taken.

September 3, 2024
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Western Minnesota

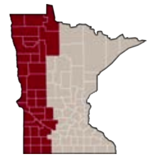

Fig. 1. The fall pre-plant nitrate test is appropriate for the maroon-shaded counties.

The use of the fall or spring PPNT is a key management tool for corn producers in western Minnesota. The suggestion that residual N in the fall can impact the need for nitrogen is contingent on the fact that the evapotranspiration of water historically has exceeded precipitation in this area of the state.

Use of the fall PPNT is appropriate in the maroon counties shown in Figure 1. The PPNT is particularly useful for conditions where elevated residual nitrate-N is suspected. Figure 2 is a decision tree that indicates situations where the nitrate-N soil test would be especially useful.

For the PPNT, soil should be collected from a depth of 6 to 24 inches in addition to the 0 to 6-inch sample that is used to test for pH, phosphorus and potassium.

Corn growers in western Minnesota also have the option of collecting soil from 0 to 24 inches and analyzing the sample for nitrate-nitrogen (NO_3-N). This 0 to 24-inch sample should not be analyzed for pH, phosphorus and potassium because the results cannot be used to predict lime needs or rates of phosphate and potash fertilizer needed.

When using the spring or fall PPNT, the amount of fertilizer N required is determined from the following equation:

NG = (Table 1 value for corn/corn) - (0.60 x STN(0-24in.))

- NG = Amount of fertilizer N needed (lbs N/acre)
- Table 1 value = the amount of fertilizer needed to be adjusted for soil potential, value ratio and risk
- STN(0-24 inch) = Amount of nitrate-N measured by using the fall PPNT (lbs N/acre)

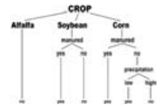

Figure 2: Flow chart decision-aid for determining probability of having significant residual nitrate-nitrogen in the soil following specific crop and situations where manure has been applied in a field within two to three cropping years prior to soil sample collection.

South-central, southeastern, east-central Minnesota

Research has led to the inclusion of a spring PPNT to adjust fertilizer N guidelines in south-central, southeastern and east-central Minnesota (gray counties in Figure 1). Soil nitrate-N, measured in the spring before planting from a two-foot sampling depth, is an option that can be used to estimate residual N.

In implementing this test, the user should first evaluate whether conditions exist for residual N to accumulate. Factors such as previous crop, soil texture, manure history and preceding rainfall can have a significant effect on the accumulation of residual N.

A crop rotation that has corn following corn generally provides the greatest potential for significant residual N accumulation. In contrast, when soybean is the previous crop, much less residual N has been measured. The PPNT should not be used following alfalfa.

The spring PPNT works best on medium and fine-textured soils derived from loess or glacial till. The use of the soil N test on coarse-textured soils derived from glacial outwash is generally not worthwhile because these soils consistently have low amounts of residual nitrate-nitrogen.

The amount of residual nitrate-nitrogen in the soil is also dependent on the rainfall received the previous year. In a year following a widespread drought (2012 for example) a majority of fields will have significant residual nitrate. However, following relatively wet years, little residual nitrate can be expected.

Nitrogen fertilizer guidelines for corn can be made with or without the soil N test. The University of Minnesota’s N guidelines (Table 1) are still the starting point. A five-step process is suggested when the soil nitrate-nitrogen test is considered.

1. Determine N rate guideline using Table 1 using soil productivity, price/value ratio, and previous crop for the specific field. The prescribed (rate assumes that best management practices (BMPs) will be followed for the specific conditions).

2. Determine whether conditions are such that residual nitrate-nitrogen may be appreciable. Figure 2, which includes factors such as previous crop, manure history and previous fall rainfall can provide insight as to the applicability of testing for nitrate-nitrogen. If conditions are such that the probability of residual nitrate is small and soil testing for nitrate is not recommended, use the N guideline derived in Step 1.
3. If conditions suggest that a soil nitrate test is warranted, collect a pre-plant, 0-2 ft. soil sample taking enough soil cores from a field so that the sample is representative of the entire field. The sample should be sent to a laboratory and analyzed for nitrate-nitrogen.
4. Determine residual N credit based on the measured soil nitrate-nitrogen concentrations. Use Table 8 to determine this credit.
5. Calculate the final N rate by subtracting the residual N credit (Step 4) from the previously determined N guideline (Step 1). The resulting fertilizer N rate can then be applied either pre-plant and/or as a side-dress application.

Table 8: Residual N credit values based on the concentration of nitrate-N measured before planting in the spring from the top two feet of soil

Soil nitrate-N	Residual N credit
0.0-6.0 (ppm)	0 (lbs N/acre)
6.1-9.0	35
9.1-12.0	65
12.1-15.0	95
15.1-18.0	125
Over 18.0	155

Because of the diversity of soils, climate and crops in Minnesota, there are no uniform statewide guidelines for the selection of a source of fertilizer N, placement of the N fertilizer and use of a nitrification inhibitor.

In order to accurately address this diversity, Minnesota has been divided into five regions and BMPs for N use in each region have been identified and described. The listing of these management practices for all regions is not appropriate for this publication, but they are available at the Minnesota Department of Agriculture.

Currently, the use of these BMPs is voluntary. Corn growers should implement BMPs to optimize N use efficiency, profit and protect against increased losses of nitrate-nitrogen to the environment.

Authors: Daniel Kaiser, Fabian Fernandez and Melissa Wilson, Extension nutrient management specialists; Jeffrey Coulter, Extension corn agronomist; and Keith Piotrowski, director of the soil testing laboratory

Taking soil samples for nitrogen analysis could pay big this year

blog-crop-news.extension.umn.edu/2022/03/taking-soil-samples-for-nitrogen.html

By: Brad Carlson, Extension educator

Record high nitrogen (N) fertilizer prices have received plenty of attention over the past several months. While farmers are scratching their heads trying to keep input costs down, an unusual opportunity is presenting itself this spring. Last year's exceptionally dry weather may have led to a nitrogen carryover credit that is not normally there.

The 2021 drought and soil nitrate levels

The nitrogen cycle naturally converts nitrogen in soil to the nitrate form. The nitrate ion is negatively charged, like soil particles, and therefore is not bound to the soil. Nitrate moves readily with water beyond the rooting zone or it can be lost via "denitrification" to the atmosphere if the soil stays saturated for long periods of time. Some nitrate naturally accumulates in the soil after the crop matures but before the soil cools down. This accumulated nitrate, together with any leftover fertilizer from the previous growing season, is usually lost during the spring before the next year's crop can take it up. It is for this reason that N fertilizer recommendations do not consider a nitrogen credit for this late season accumulated nitrate under normal circumstances. However, when there is enough of a water deficit in the soil profile that melting snow and spring rains do not

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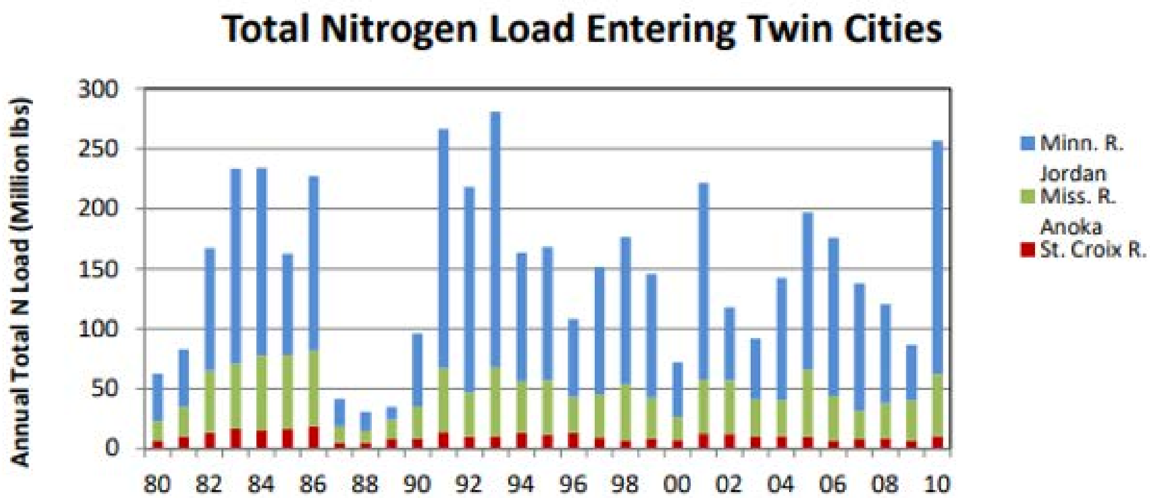
saturate the soil, there is less risk of N loss, meaning you may be able to take an N credit. Historical records show that nitrate concentration in surface water spikes in years following drought or excessively dry conditions. After the soil finally saturates, this accumulated soil nitrate flushes into surface water. It is anticipated that this may be the case at some point either this year or next year following the dry conditions experienced in much of the state in 2021.



This chart shows how nitrogen loads in Minnesota's rivers spiked following the drought of the late 1980s (MPCA 2013). A similar situation could unfold this year or next year if residual soil nitrate levels aren't accounted for and N fertilizer is overapplied.

Clues from last fall

Minnesota Valley Testing Labs conducted soil nitrate tests last fall and generously shared their data with us. Over 70% of the nearly 240 samples analyzed had an N credit of at least 35 pounds. Nearly 30% of the samples had N credits of 155 pounds or more. It should be noted that this is not a random sampling of sites, but rather an indication of soil N status where a carryover is suspected on the western side of the state where a fall test is considered acceptable. Data provided by Centrol Crop Consulting for tests run in advance of planting sugar beets shows a much smaller prevalence of N credits, with less than 5% showing a credit following soybeans, and about one third of samples indicating a credit following corn. However, there are some fundamental differences between the pre-sugar beet test and the pre-plant soil nitrate test (PPNT) for corn, and the samples were from the Red River Valley in northwest Minnesota where conditions were not nearly as dry as the southern third of the state.

Table 1. *Fall 2021 MVTL soil nitrate test results (239 samples, 0-24 inch samples)*

Nitrate level (ppm)	% of samples	N credit (lbs./ac)
0-6	28%	0
6-9	14%	35
9-12	10%	65
12-15	10%	95
15-18	5%	125
18+	28%	155

Pre-plant soil nitrate test (PPNT) tips

University of Minnesota research going back to the early 1990s resulted in the recommendation to use a soil nitrate test to measure this effect and credit residual N against the next year's fertilizer inputs under certain circumstances. The situations where a credit is likely are:

1. Fields that have a long-term manure history
2. Continuous corn following a drought

It should be noted that the test will work on any field where corn is going to be grown this year; it is just less likely that you will find an N credit in other circumstances, so it may not be worth the cost of testing. Also, it is important to note that portions of western and northern Minnesota received enough precipitation toward the end of, or after, the growing season to bring soils to field capacity, meaning you are less likely to find an N credit.

Minnesota's PPNT recommendations call for taking samples two feet deep to capture any nitrate that has already moved but still within the rooting zone. With the ability to variable-rate apply, it makes sense to break a field into management zones where there are likely to be differences (like one would do for any soil test). Be sure to take enough cores to ensure a good average, mix thoroughly, and dry quickly. Our research has shown that, in this case, more is always better. We suggest a minimum of 10 cores in a composite sample for the test results to be representative of the portion of the field or area of interest.

There are a few points to keep in mind if you are going to use the PPNT. Since nitrate is subject to leaching or denitrification loss, you want to take the sample as late as possible to ensure that what is measured is still there at the time the plants need it.

Furthermore, if it becomes extremely wet after a sample is collected, the credit may disappear, so be prepared to compensate for this with additional sidedress N.

If you're growing corn this year, remember to request and use the PPNT, which gives results in ppm nitrate, and not the pre-sugar beet test, which gives results in pounds per acre. The interpretation of the results for corn are only calibrated for the PPNT, not the pre-sugar beet test. For similar reasons, you should not use recommendations from out-of-state, as the interpretation of the test results may not be correlated to the test protocol used in Minnesota.

Another point to remember is that the test only finds nitrate, so it will not accurately measure any fertilizer already applied or available N from a manure application. And lastly, Do not confuse the PPNT with the pre-sidedress nitrate test (PSNT), which is taken in-season and is most useful for fields with substantial potential for mineralization because of previous manure applications or where alfalfa was terminated.

More detailed instructions for taking the PPNT, as well as a chart to interpret results, can be found in our corn fertilizer guidelines.

For the latest nutrient management information, subscribe to the Minnesota Crop News email newsletter, like UMN Extension Nutrient Management on Facebook, follow us on Twitter, and visit our website.

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Manure Overload | Environmental Working Group

ewg.org/research/manure-overload

Manure Plus Fertilizer Overwhelms Minnesota’s Land and Water

In almost all of Minnesota’s farm counties, the combination of manure plus commercial fertilizer is likely to load too much nitrogen or phosphorus or both onto crop fields, threatening drinking water and fouling the state’s iconic lakes and rivers, according to an Environmental Working Group investigation.

The problem arises from the extraordinary expansion and intensification of both livestock and crop production in the state. Since 1991, the number of large concentrated animal feeding operations, or CAFOs, in Minnesota has tripled. At the same time, fertilizer sales have increased by more than a third, fueled by the nearly 1.5 million additional acres devoted to corn.

Every year, feedlots of all sizes in the state produce nearly 50 million tons of manure – rich in nitrogen and phosphorus, the same chemicals in the more than three million tons of commercial fertilizer applied annually. Nitrogen and phosphorus are essential crop nutrients, but when they run off the fields, they can pollute drinking water sources and other bodies of water.

Using advanced geospatial techniques, EWG simulated and mapped every crop field across Minnesota likely to receive manure from nearby cattle, hog or poultry feedlots, to estimate the amount of manure applied in each county. We then added those amounts to the nitrogen and phosphorus in the fertilizer sold in the county.

The results are bad news for the state's water quality .

- In 69 of Minnesota's 72 agricultural counties, nitrogen from manure combined with nitrogen in fertilizer exceeded the recommendations of the Minnesota Pollution Control Agency, or MPCA, and the University of Minnesota. In 13 counties, nitrogen from the two sources surpassed the recommendations by more than half. (Table 1.) This excess nitrogen is the major cause of nitrate pollution in drinking water, which is linked to elevated rates of cancer.
- In nine counties, phosphorus pollution from manure is of high concern. These nine counties account for over half of the nearly 1.5 million acres where application of manure adds at least 10 pounds per acre more phosphorus than needed by crops. (Table 2.) Four of those counties are also among the 13 with the most excess nitrogen. Phosphorus pollution of lakes and rivers can trigger algae blooms, which are not only ugly but can also produce toxic bacteria harmful to human and animal health.

TABLE 1. MINNESOTA COUNTIES WITH HIGH RISK OF NITROGEN OVERLOAD

COUNTY	PERCENT N RECOMMENDATION MET BY MANURE APPLIED	PERCENT N RECOMMENDATION MET BY FERTILIZER SOLD	PERCENT N RECOMMENDATION MET BY MANURE AND FERTILIZER COMBINED	TONS OF N OVERLOAD
Martin	69%	107%	176%	14,368
Stearns	69%	91%	160%	12,564
Fillmore	33%	122%	154%	7,641
Goodhue	38%	122%	160%	7,180
Rock	73%	87%	159%	6,808
Morrison	84%	91%	175%	6,646
Nicollet	50%	107%	157%	6,111
Waseca	46%	107%	154%	5,376
Pipestone	68%	87%	155%	5,136
Winona	73%	122%	194%	4,977
Todd	67%	91%	157%	4,266
Wabasha	40%	122%	162%	4,150
Houston	57%	122%	179%	2,476

Source: EWG via Minnesota Pollution Control Agency, USDA-ARS Agricultural Conservation Planning Framework Database, Midwest Plan Service, University of Minnesota Extension and Minnesota Department of Agriculture.

TABLE 2: COUNTIES WITH HIGH RISK OF PHOSPHORUS OVERLOAD

COUNTY	AVERAGE P EXCESS ON MANURED ACRES *(LBS/ACRE)*	NUMBER OF MANURED ACRES	TONS OF EXCESS P ON MANURED ACRES
Morrison	25	132,566	1,730
Todd	19	114,027	1,092
Kandiyohi	19	128,282	1,331
Stearns	18	295,547	2,608
Winona	16	68,490	609
Otter Tail	14	126,474	873
Meeker	12	97,893	667
Fillmore	10	88,502	358
Goodhue	8	93,823	359

Source: EWG via Minnesota Pollution Control Agency, USDA-ARS Agricultural Conservation Planning Framework Database, Midwest Plan Service, University of Minnesota Extension, Minnesota Department of Agriculture and USDA National Agricultural Statistics Service.

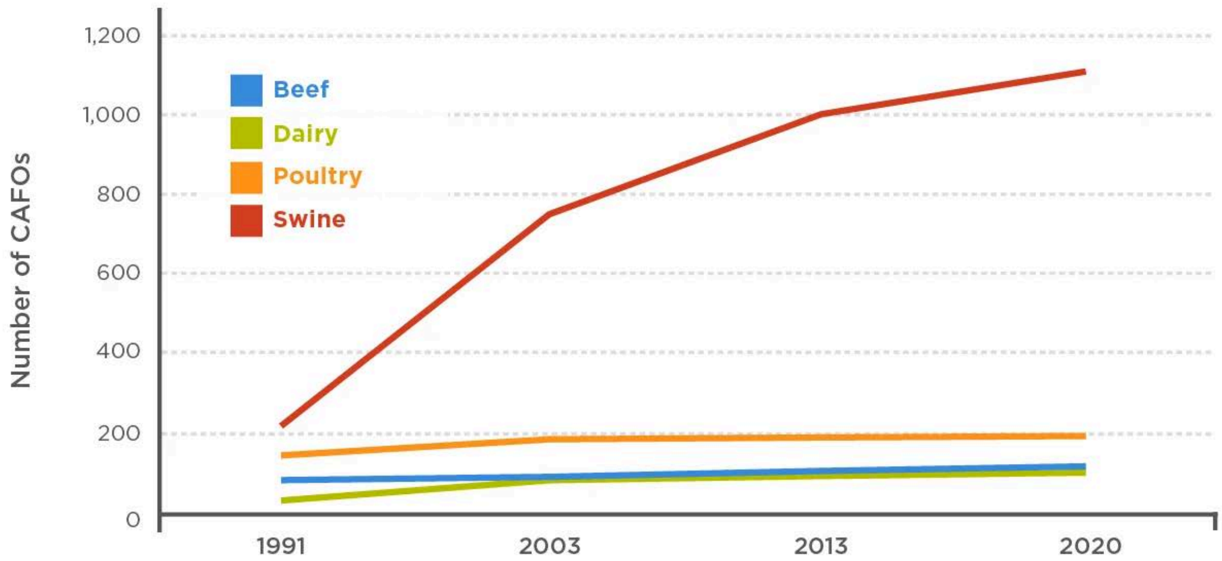
Water Pollution Is Increasing

The statewide overload of nitrogen and phosphorus is taking its toll.

- An earlier EWG investigation found that 63 percent of Minnesota public water utilities with elevated levels of nitrate saw worsening contamination between 1995 and 2018.
- In the Sauk River watershed , the MPCA has listed nine lakes and four stream reaches as "impaired" because of bacteria, excess nutrients – mainly phosphorus – and algae blooms.
- After assessing all of the state's major watersheds, the MPCA estimates that 56 percent of surface waters do not meet basic water quality standards, and that non-point source pollution, such as that from crop and livestock production, contributes to 85 percent of the state's water pollution.

Since 1991, the number of large CAFOs in Minnesota has swelled from 468 operations to 1,497. (Figure 1.) Of the new operations, 86 percent were for feeding hogs, although the number for all other animals also grew. These operations are also getting bigger: Eight of the 67 dairy CAFOs built since 1991 house more than 8,000 cows, compared to just one of that size in 1991.

FIGURE 1: NUMBER OF LARGE LIVESTOCK OPERATIONS IN MINNESOTA TRIPLED SINCE 1991

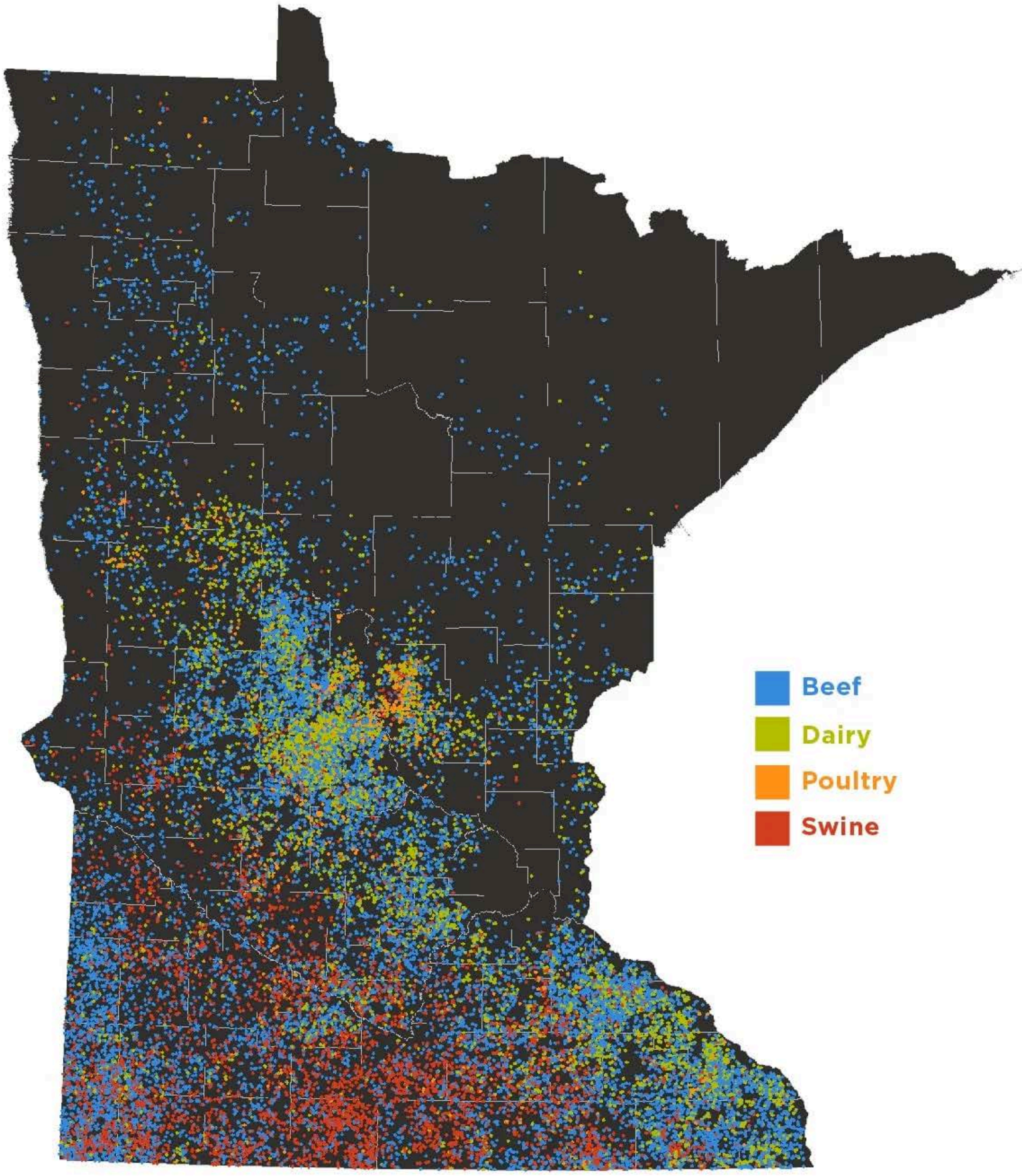


Source: EWG via Minnesota Pollution Control Agency Feedlot Database.

This extraordinary expansion raises concerns about the environmentally safe disposal of manure. Large CAFOS are just 4 percent of feeding operations in the state, but they produce nearly a third of the manure. Medium-size feedlots are 18 percent of all operations and contribute another 43 percent of the manure that goes on Minnesota fields every year.

Today Minnesota has 23,725 feedlots of all sizes. Packed into counties in southern and central Minnesota, these operations house up to 1.2 million dairy cows, 1.6 million beef cows, 10.9 million hogs, and 66 million turkeys and chickens. These feedlots produce an estimated 49 million tons of manure annually – the equivalent of the waste from 95 million people, 17 times the state's human population.

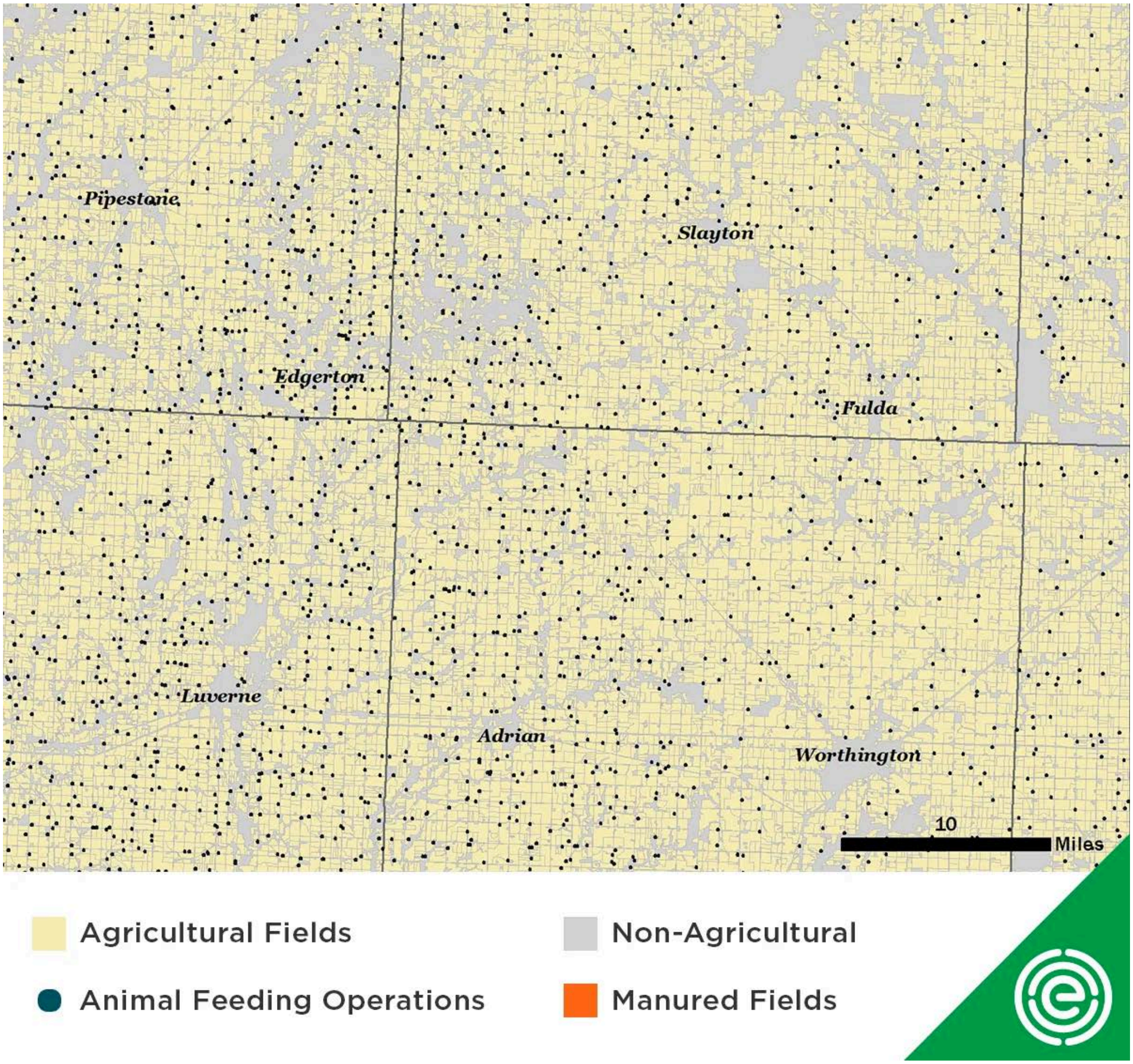
FIGURE 2: MINNESOTA IS HOME TO NEARLY 24,000 FEEDLOTS



Source: Minnesota Pollution Control Agency Feedlot Database.

EWG simulated which individual fields could safely accept manure, based on distance from the feedlot and the amount of nitrogen recommended for growing crops. Nitrogen rates were based on MPCA guidelines and University of Minnesota fertilizer recommendations.

Figure 3: How Manure Moves From Feedlots to Fields



Source: EWG via Minnesota Pollution Control Agency, USDA-ARS Agricultural Conservation Planning Framework Database, Midwest Plan Service, University of Minnesota Extension and Minnesota Department of Agriculture.

In areas with a dense concentration of livestock, nearly every single crop field is needed if all the manure produced by nearby feedlots is to be used safely, without overloading nitrogen. In a few isolated areas, there is simply too much manure to dispose of within a reasonable distance. EWG's simulation likely understates the risk of this overload, because we assumed every field within 5 miles of a cattle or hog feedlot and 25 miles of a poultry feedlot was available to take manure.

Moreover, research shows that much of the nitrogen considered lost to the atmosphere during manure storage and application ends up redeposited on the land nearby, adding to the potential overload.

The concentration of feedlots leaves little or no room to adapt to year-to-year changes in cropping patterns and fluctuating manure composition. It also increases the risk of overloading fields with phosphorus.

Manure Is Only Half the Story

You might expect fertilizer sales to be low in counties with dense concentrations of livestock, where manure alone can take care of the need for nitrogen fertilizer. Instead, we found little relationship between manure produced and fertilizer sold. Table 1 above lists 13 counties that are hot spots for nitrogen overload, where nitrogen from manure combined with nitrogen in fertilizer sold in the county exceeded crop recommendations by more than 50 percent.

Fertilizer sold in a county does not necessarily mean it was used there: A county might have half a neighboring county's crop acreage yet sell twice as much fertilizer. To account for this, we grouped fertilizer sales for counties within Minnesota's major crop regions, then allotted this regional sales data to counties based on fertilizer needs.

The interactive map below shows areas with an overload of nitrogen, as identified by our simulation.

Explore the Map

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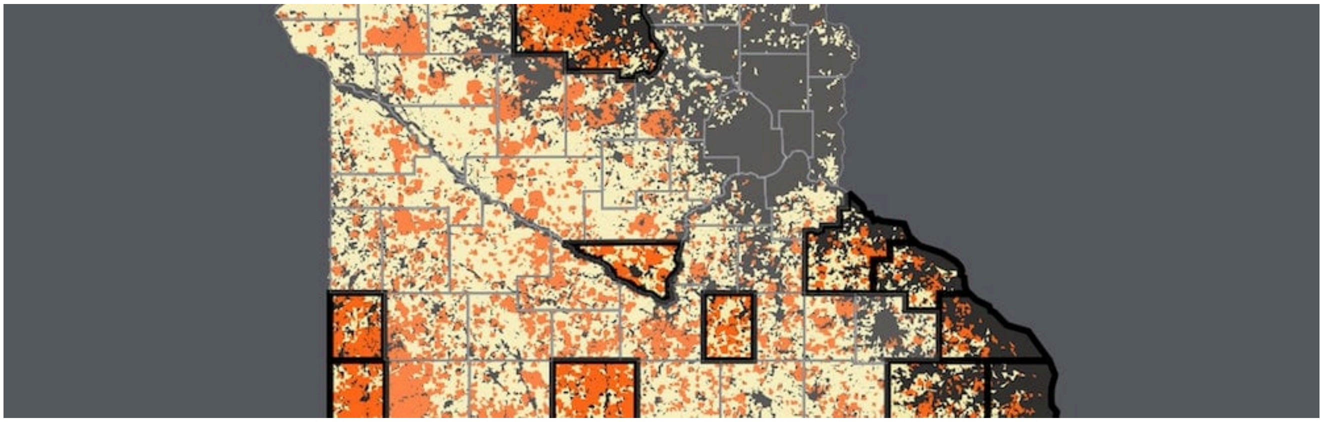

It’s not surprising that the counties we identified are dealing with nitrate overload issues. Southwest Minnesota has struggled with nitrate-contaminated water for decades. In 2014, the MPCA declared that most bodies of water in the area did not meet standards for supporting aquatic life and recreation, and the town of Adrian has been forced to shut down a water treatment plant after nitrate levels exceeded the U.S. Environmental Protection Agency’s legal limits. In Minnesota’s farthest southwest corner, Rock County’s water system’s average nitrate concentration increased by a staggering 890 percent from 1995 to 2018, according to EWG calculations.

Most of the CAFO growth in the state has been in Martin County, in south central Minnesota, home to 15 lakes on Minnesota’s 2020 list of nutrient-impaired water bodies. The list includes Budd Lake, which serves as the drinking water source for the town of Fairmont.

In townships in Morrison and Winona counties, the Minnesota Department of Agriculture found that more than 40 percent of private wells sampled had nitrate levels above the federal health limit of 10 micrograms per cubic liter. Many high-risk counties are located in vulnerable areas of the state, where karst bedrock or sandy soils make it easy for pollutants to reach groundwater.

The Phosphorus Problem

An inherent problem with manure is the imbalance between nitrogen and phosphorus relative to crop needs. When manure is applied to meet the nitrogen recommendation for crops, phosphorus is often overapplied. This nutrient imbalance is worse for poultry and cattle manure. The University of Minnesota Extension states that when turkey manure is applied to meet the nitrogen recommendation for corn, the crop gets more than five times the phosphorus needed.

Applying more phosphorus than the growing crop needs can lead to a buildup in the soil and greatly increases the risk of pollution. This risk is elevated in steep fields or those closer to lakes and streams. Long-term research in South Dakota showed that cattle manure applied to meet the nitrogen recommendation of crops dramatically increased soil phosphorus levels in less than 10 years. Eight pounds an acre of excess phosphorus can increase the level of phosphorus in the soil by 1 part per million, or ppm, which can quickly create problems for fields receiving manure year after year.

In Minnesota, soil phosphorus levels above 150 ppm (or 75 ppm near bodies of water) triggers action that requires farmers to lower the phosphorus levels from manure application. Other states, such as Indiana, have set this level even lower, suggesting that soil phosphorus is a concern once levels pass 50 ppm.

Our simulation found that on over 2.6 million Minnesota crop acres, or 57 percent of fields that received manure, more phosphorus was applied than removed. On nearly 1.5 million acres, this excess was more than 10 pounds per acre. On 590,000 acres, or 14 percent of manured fields, the excess was more than 25 pounds an acre.

Of the manured fields with a phosphorus excess greater than 10 pounds an acre, more than half fell in nine counties, as shown in Table 2, above. All nine counties are located in central and southeast Minnesota, and all have high densities of poultry and dairy operations.

In four counties – Morrison, Stearns, Todd and Winona – phosphorus from manure alone exceeds total crop requirements. Compounding the problem are the tons of additional phosphorus fertilizer sold in these same counties. Manure plus fertilizer phosphorus exceeds crop requirements in all but one of the counties in Table 2 (Otter Tail) and ranged from 90 percent to just over twice the phosphorus needed for the crop.

To limit phosphorus pollution from manure, farmers should apply manure to meet the phosphorus, not nitrogen, requirements of the crop. But because manure has much more nitrogen than phosphorus, far more acres are needed to apply manure at the proper rate for phosphorus. This can be twice as many acres needed for swine, compared to five times as many acres for turkey manure. In areas already saturated with manure, it is unlikely that this additional land is available.

Phosphorus pollution is the primary driver of algae growth in lakes. In the Sauk River watershed, in the heart of central Minnesota, the MPCA has set a Total Maximum Daily Load, or TMDL, to address bacteria, excess nutrients (mainly phosphorus) and nuisance algae blooms. Lake Osakis, a well-visited recreation area in the Sauk River watershed, was identified as a priority lake for water quality improvements.

Algae blooms are not only unsightly, they also have the potential to produce toxic cyanobacteria that are harmful to both human and animal health. Not far from Lake Osakis, in 2015 a child swimming in Lake Henry was hospitalized after exposure to blue-green algae. This followed the death of two dogs exposed to blue-green algae in nearby Red Rock Lake.

These examples are in central Minnesota, but algae blooms are common across all areas of the state with dense concentrations of cropland and livestock. The interactive map below shows the areas our simulation identified as having an overload of phosphorus.

Explore the Map

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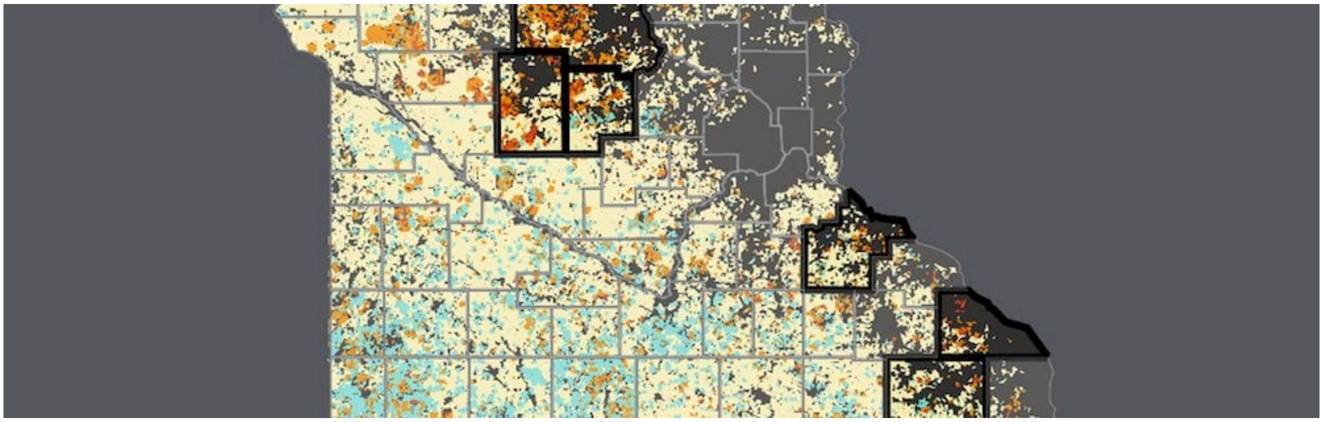

Manure Overload and Public Health

Contamination of water resources poses a real threat to Minnesota drinking water and public health. Growth and consolidation of animal agriculture intensifies this threat. Accurately crediting the amount of nitrogen and phosphorus in manure before any fertilizer is applied will improve soil health, protect drinking water and improve Minnesota's lakes, rivers and streams while saving farmers millions of dollars in reduced commercial fertilizer costs. The data strongly suggest, however, that isn't happening – especially in areas with dense concentrations of livestock.

A Minnesota Department of Agriculture survey revealed that almost three-fourths of farmers did not know how much nitrogen their manure contained, a basic requirement for good manure and fertilizer management. The same survey showed that almost two-thirds of farmers apply manure in the fall, a practice that increases the risk of nitrogen and phosphorus loss from manured fields, especially for liquid manure produced by hog and large dairy operations. Meanwhile, conservation practices that could reduce pollution from manure, such as cover crops, are drastically underused.

A comprehensive assessment of the capacity of Minnesota's landscape to handle its manure and fertilizer load is essential to ensure current and future residents have clean water. That assessment must drive decisions about where to site new or expanded feedlots and set standards for fertilizer and manure management, especially in areas with dense livestock.

For methods and detailed results, click here.

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Sources and Risk Factors for Nitrate and Microbial Contamination of Private Household Wells in the Fractured Dolomite Aquifer of Northeastern Wisconsin

ncbi.nlm.nih.gov/pmc/articles/PMC8221036

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Author information Article notes Copyright and License information PMC Disclaimer
See "Quantitative Microbial Risk Assessment for Contaminated Private Wells in the Fractured Dolomite Aquifer of Kewaunee County, Wisconsin" in volume 129, 067003.
See "Farm to Faucet? Agricultural Waste and Private Well Contamination in Kewaunee County, Wisconsin" in volume 129, 114001.
Go to:

Abstract

Background:

Groundwater quality in the Silurian dolomite aquifer in northeastern Wisconsin, USA, has become contentious as dairy farms and exurban development expand.

Objectives:

We investigated private household wells in the region, determining the extent, sources, and risk factors of nitrate and microbial contamination.

Methods:

Total coliforms, *Escherichia coli*, and nitrate were evaluated by synoptic sampling during groundwater recharge and no-recharge periods. Additional seasonal sampling measured genetic markers of human and bovine fecal-associated microbes and enteric zoonotic pathogens. We constructed multivariable regression models of detection probability (log-binomial) and concentration (gamma) for each contaminant to identify risk factors related to land use, precipitation, hydrogeology, and well construction.

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Results:

Total coliforms and nitrate were strongly associated with depth-to-bedrock at well sites and nearby agricultural land use, but not septic systems. Both human wastewater and cattle manure contributed to well contamination. Rotavirus group A, *Cryptosporidium*, and *Salmonella* were the most frequently detected pathogens. Wells positive for human fecal markers were associated with depth-to-groundwater and number of septic system drainfield within 229m. Manure-contaminated wells were associated with groundwater recharge and the area size of nearby agricultural land. Wells positive for any fecal-associated microbe, regardless of source, were associated with septic system density and manure storage proximity modified by bedrock depth. Well construction was generally not related to contamination, indicating land use, groundwater recharge, and bedrock depth were the most important risk factors.

Discussion:

These findings may inform policies to minimize contamination of the Silurian dolomite aquifer, a major water supply for the U.S. and Canadian Great Lakes region.

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Introduction

The paradox presented to the 13.1 million households in the United States that rely on private wells for supplying their drinking water (NGWA 2020) is that the household owns the well and the land on which the well is constructed, but it does not control the source, movement, and quality of the pumped groundwater. Anthropogenic disturbances on neighboring properties, such as changes in land cover, building development, agricultural practices, septic systems, and groundwater withdrawals, can alter the supply and quality of groundwater on which the household depends. Thus, as a shared natural resource, groundwater is susceptible to the “tragedy of open access” (Bromley and Cernea 1989), where without appropriate institutional safeguards the resource (i.e., groundwater) can become diminished and degraded.

This tension of having competing land uses affect the shared groundwater resource is particularly noteworthy in northeastern Wisconsin, where both dairy farms and exurban development have expanded atop the underlying Silurian dolomite aquifer. The aquifer is the water source for at least 85% of private wells in the region (K. Bradbury, Wisconsin State Geologist, personal communication). In the region’s four main agricultural counties, Brown, Calumet, Kewaunee, and Manitowoc, the number of milking dairy cows increased from 132,558 to 180,860 between 2002 and 2017, a 36% increase (USDA NASS 2002, 2017). This number of milking cows produces approximately 5.9×109 kg of excrement (manure and urine) per year (Nennich 2005), which in northeastern Wisconsin is all applied to the landscape (Erb et al. 2015). Population growth in the four-county region between 1950 and 2000 increased exurbanization by as much as 60% (Brown et al. 2005). Dairy farms and exurban homes are in greater proximity than years ago, each land use potentially contributing to the degradation of the common groundwater resource on which they depend.

Compounding the effects of more intensive land use on groundwater quality is the highly vulnerable nature of the Silurian dolomite aquifer, which is an important water supply for the region (Figure 1). The dolomite bedrock is densely fractured in both horizontal and vertical directions, and in many regions the surficial sediment overlying the bedrock is thin, i.e., 6m or less (Sherrill 1978). Groundwater recharge is extremely rapid because soil macropores and the extensive vertical fracture network allow rain and snowmelt water to infiltrate easily (Muldoon and Bradbury 2010). Infiltrating water carries contaminants originating at the land surface to the water table, after which groundwater flow in horizontal fractures can be rapid, providing little attenuation to contaminant transport (Bradbury and Muldoon 1992; Muldoon et al. 2001).

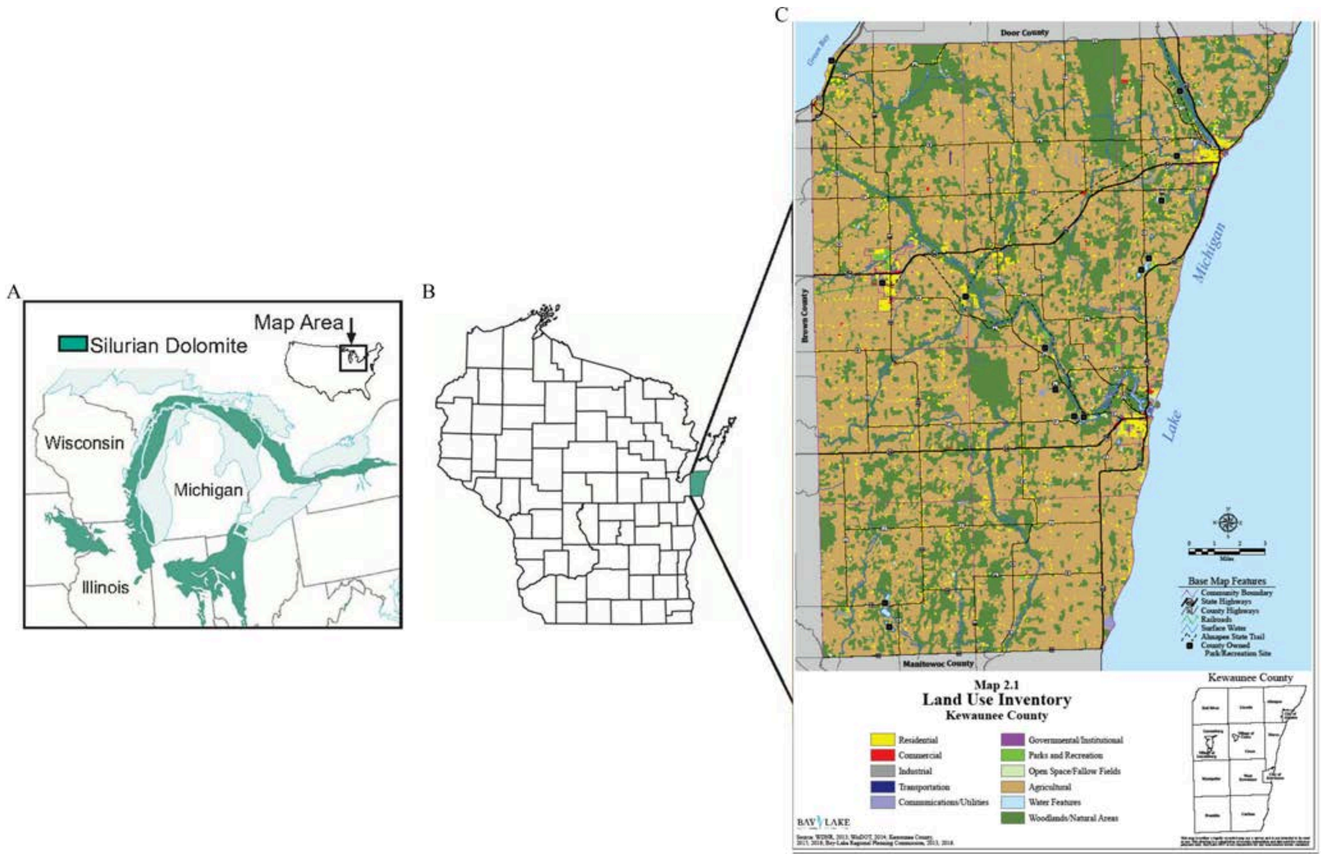

Figure 1.
Location of study site including (A) map of generalized Silurian dolomite subcrop shown as shaded area (modified from Shaver et al. 1978); (B) location of Kewaunee County, Wisconsin, United States; and (C) map of land use within the county. Land use map reprinted with permission from Bay Lake Regional Planning Commission, Green Bay, Wisconsin.

Contamination of private household wells open to the Silurian dolomite aquifer has been evaluated primarily by standard indicator bacteria for water sanitary quality (i.e., total coliform bacteria and *Escherichia coli*) and nitrate–nitrogen (NO_3^--N) In the five-county region where the aquifer is most vulnerable (Brown, Calumet, Door, Kewaunee, and Manitowoc Counties), 14% of 7,521 samples from private wells exceeded the U.S. Environmental Protection Agency (U.S. EPA) health advisory of 10mg/L for NO_3^--N for public water supplies (U.S. EPA 2020). Twenty-three percent of 6,739 samples tested positive for total coliforms, and 2% of 6,583 samples were positive for *E. coli* (Center for Watershed Science and Education Wisconsin 2018). Although these analyses may indicate the extent of contamination, they do not provide information on the source of contamination.

The most obvious contamination events happen when manure enters the aquifer and is pumped from a household well into indoor taps as odoriferous brown water (Figure 2). Manure-containing brown water incidents are more likely during groundwater recharge when snow is melting and after dairy manure is applied to agricultural fields (Erb et al. 2015). Erb et al. (2015) documented 25 brown water incidents between 2008 and 2014 in domestic wells located in Brown, Calumet, Kewaunee, and Manitowoc counties, and these incidents can present a health risk (Wisconsin Department of Health Services n.d.).

Figure 2.
"Brown water" event at a Kewanee County household with a private well. Note: Photo provided and permission granted by Chuck Wagner.

As the "tragedy of open access" of the groundwater resource in northeast Wisconsin was unfolding, public debate centered on two questions: *a*) what is the true extent of groundwater contamination? and *b*) what are the sources of contamination, septic systems or dairy manure? Through interactions

with stakeholders, we learned that historical total coliform and nitrate data were considered biased by some because it was believed samples were submitted only from problem wells that were not representative of groundwater conditions. As for the source question, opposing sides generally took positions without having data in hand, because the technology of microbial source tracking (MST) to identify fecal sources has rarely been applied to household wells. To help resolve these questions and bring information to bear on potential solutions, we proposed three study objectives: *a*) conduct random sampling of private wells, stratified by depth-to-bedrock, for indicator bacteria and nitrate; *b*) from the subset of wells in Objective 1 that were positive for total coliform bacteria or had NO_3^--N>10mg/L, conduct random sampling for enteric pathogens and MST markers indicating whether fecal contamination was from septic systems or dairy manure; and *c*) perform statistical analyses to identify land use, weather, hydrogeology, and well construction risk factors that were associated with private well contamination.
Go to:

Methods

Study Area

The study area was Kewaunee County located in northeast Wisconsin, USA (Figure 1). The county's population is 20,600, of which 11,300 (55%) live in 4,900 rural homes served by septic systems and private wells (Bay Lake Regional Planning Commission 2016). Land cover in the 2808-km2 county is predominantly agriculture (63%), natural areas (29%), and residential (3%) (Bay Lake Regional Planning Commission 2016). Dairy farming and associated crop production are the primary agricultural activities. Cattle and calves number approximately 107,000 on 306 farms (USDA NASS 2017). The climate is continental, modified by the proximity of Lake Michigan, with precipitation (rain and snow) of 78cm water per year (NOAA n.d.). Soils are medium- to fine-textured, underlain by Pleistocene glacial deposits; unconsolidated sediments vary in thickness from several centimeters to more than 30m over the bedrock (Erb et al. 2015). Karst features such as open fractures are present, albeit many are covered with soil (Erb et al. 2015).

Indicator Bacteria and Nitrate

Private household wells were selected by stratified random sampling for tests of total coliforms (hereafter coliforms), *E. coli*, and nitrate. Candidate wells were identified from a list of property parcels that *a*) were not served by municipal water systems and *b*) had improvement values greater than USD $30,000, which indicated that a residence (and therefore private well) was likely present (n=4,896). Parcels with mailing and property addresses that did not match were excluded to prevent confusion regarding sample location (n=948).

Water sampling was conducted during two synoptic events, 13–14 November 2015 and 29–30 July 2016. Strata were defined by depth-to-bedrock (i.e., the depth of unconsolidated sediment overlying bedrock at the well site) because earlier work suggested this parameter influenced groundwater contamination (Final Report of the Northeast Wisconsin Karst Task Force 2007). Using ArcMap software (version 10.3.1; ESRI), candidate wells were grouped into three strata based on an existing depth-to-bedrock map (Sherrill 1979): <1.5m (n=269), 1.5–6.1m (n=473), and >6.1m (n=3,206). (Depth-to-bedrock data were not available for individual wells at the time of well selection.) Letters inviting participation were mailed, and all willing well owners (approximately 50% of invitees) received a sampling kit. After accounting for unreturned kits, 323 and 401 private well samples were submitted for the fall and summer sampling events, respectively. Some wells (103) were sampled in both events (see Figure S1 for well recruitment, exclusion, and dropout). All study wells were completed in the Silurian dolomite or overlying sediment.

Samples were collected by well owners following written instructions to sterilize the sample tap with a flame for 15 s or by alcohol swab and run the water run for at least 5 min prior to filling two polypropylene bottles provided in the sampling kit. The 60-mL nitrate bottle contained 160μL of 96% sulfuric acid for preservation. Samples were collected on the scheduled dates and on the same day delivered to designated receiving locations in the county where they were transported that day on ice to the laboratory. Coliforms and *E. coli* were analyzed by Colilert Quanti-Trays (IDEXX) within 48 h of sample collection. Nitrate was measured on an AQ1 Discrete Analyzer (SEAL Analytical) by cadmium reduction and reaction with sulfanilamide in conjunction with N-(1-naphthylethylenediamine) dihydrochloride (Method 4500-NO^-_3F; American Public Health Association 1995).

Microbial Source Tracking and Pathogen Occurrence

Wells positive for coliforms or with NO^-_3-N>10mg/L were eligible for additional sampling to assess sources of fecal contamination and the occurrence of enteric pathogens. From this group, wells were selected for five sampling events: 18–22 April, 1–3 August, and 31 October–2 November in 2016 and 23–24 January and 27–29 March in 2017. For each event, selection was randomized and stratified by the three depth-to-bedrock categories. We sampled 22 to 30 wells during each event, resulting in 138 samples from 131 wells; seven wells were sampled in two events.

Sampling was conducted by trained staff using dead-end ultrafiltration (Smith and Hill 2009) with Hemodialyzer Rexeed-25s ultrafilters (Asahi Kasei Medical MT Corp.). Water taps were flame-sterilized before ultrafilter attachment; all ultrafilter tubing and fittings were new for each sample. Well water was collected prior to softening or other treatment systems. Mean sample volume was 839L (range: 522–1,517L, n=138). Ultrafilters were bagged, placed on ice, and back-flushed in the laboratory within 72 h.

Ultrafilters were back-flushed using a 500-mL solution containing 0.01% sodium polyphosphate (NaPP), 0.5% Tween 80, and 0.001% antifoam Y-30 (Smith and Hill 2009). Bacto beef extract (ThermoFisher Scientific Catalog No. 211520) was added to the back-flushed eluate at a 1% weight to volume ratio (typically 6.5g of beef extract into 650mL of eluate) to provide an organic matrix for sample archival at −80°C and to aid flocculation of the secondary concentration step by polyethylene glycol (PEG) flocculation (Lambertini et al. 2008). Briefly, samples were incubated overnight at 4°C following addition of 8% PEG 8,000 and 0.2M NaCl. Samples were centrifuged for 45 min at 4,700×g at 4°C, and the pellet was resuspended in TE buffer to a final concentrated sample volume (FCSV) of 3–26mL (4mL average). FCSVs were stored at −80°C until extraction of nucleic acids. Nucleic acids were extracted from 280μL of final concentrated sample volume with the

QIAamp DNA blood mini kit and buffer AVL using a QIAcube® (Qiagen). Final volume of the nucleic acid suspension was 140μL. Three extractions were performed per sample to produce sufficient template for all gene markers assayed.

Virus RNA was reverse-transcribed (RT) by adding 25.8μL nuclease-free water and 2.1μL random hexamers (ProMega) to 25.8μL of the extracted nucleic acids. This mixture was heated for 5 min at 95°C and then mixed with 96.3μL RT master mix consisting of the following components reported as final concentrations in the 150μL total reaction volume: 50 mM Tris-HCl (pH 8.3), 75 mM KCl, 3 mM $MgCl_2$, 0.6 mM dithiothreitol, 70μM of each deoxynucleoside triphosphate (ProMega), 1U/μL RNasin® (ProMega), 0.5U/μL SuperScript® III reverse transcriptase (Invitrogen Life Technologies). Reaction incubation was 42°C for 60 min followed by 5 min at 95°C and then held at 4°C until polymerase chain reaction (PCR) amplification.

Samples were analyzed by quantitative real-time polymerase chain reaction (qPCR) for 33 gene markers specific to 30 microbial taxa or groups (see Table S1). The microbes tested were all fecal-associated and, based on the biology of the microbe or validation studies reported in the scientific literature, placed in one of three host-specificity categories: human-specific, bovine- or ruminant-specific, and no host specificity. qPCR was performed with a LightCycler® 480 instrument (Roche Diagnostics) using the LightCycler 480 Probes Master kit for all markers except for human *Bacteroides*, which used TaqMan Environmental Master Mix 2.0® (Applied Biosystems). Six μL extracted DNA or cDNA from reverse transcription was added to 14μL of master mix, producing a 20-μL reaction volume. Primers and hydrolysis probes (Integrated DNA Technology), and their concentrations are reported in Table S1. For all markers except human *Bacteroides*, thermocycling began at 95°C for 5 min followed by 45 cycles of 10 s at 95°C and 1 min at 60°C with ramp rates of 4.4 and 2.2°C per second, respectively. Thermocycling for human *Bacteroides* began at 95°C for 10 min followed by 45 cycles of 30 s at 95°C, 2 min at 56°C, and 1 min at 72°C with ramp rates of 2.2, 1.1, and 2.2°C per second, respectively. Two qPCR technical replicates were performed per marker. If both replicates were negative the result is reported as 0. If only one was positive, that concentration is reported. If both replicates were positive, the average concentration is reported.

To ensure laboratory contamination was absent (i.e., no false positives), we performed negative controls (i.e., no-template controls) of every gene marker for the extraction, reverse transcription, and qPCR steps for every batch of these process steps, and we tested for every marker in every batch of ultrafilter backflush solution. All tests had to be negative [i.e., no cycle quantification (Cq) value] for sample data to be accepted.

Inhibition was evaluated following the approach of Gibson et al. (2012), using as controls Hepatitis G virus RNA oligonucleotide (IDT) and G-lambda DNA (New England Biolabs) for reverse transcription and qPCR inhibition, respectively. Samples with Cq values of controls that increased two or more were considered inhibited. Twelve of 138 samples were qPCR-inhibited, requiring dilution with AE buffer (Qiagen).

Extraction positive controls were bovine herpes virus vaccine for DNA and bovine respiratory syncytial virus vaccine for RNA (both vaccines from Zoetis Inc.), the latter serving also as the reverse transcription positive control. qPCR positive controls were gBlocks® or Ultramers® (IDT) of each marker, with sequences modified to distinguish from wild type while maintaining the same guanine and cytosine content.

Standard curves were generated by serially diluting the positive controls in AE buffer with 0.02% bovine serum albumin, creating a concentration range of 1 to 106 gene copies (gc)/reaction. Quantification cycle (Cq) values were calculated using the second derivative maximum method and regressed against the decimal logarithm of marker concentration using the nonlinear function provided by the LightCycler® 480 software. Standard curve parameters and 95% limits of detection are reported in Table S2 and Table S3, respectively.

Samples positive by qPCR for rotavirus group A were further analyzed following the methods of Iturriza-Gómara et al. (2004) and Madadgar et al. (2015) to determine human and bovine G and P genotypes using seminested PCR assays targeting the *VP7* and *VP4* structural viral protein genes. In brief, nucleic acid extraction and reverse transcription were performed as described above. The first PCR amplified the *VP7* or *VP4* gene using VP7-F/VP7-R or Con-3/Con-2 primers, respectively. The 20-μL reaction contained 6μL of cDNA from reverse transcription, 14μL of Roche LightCycler 480 master mix, and 200 nM of each primer. A separate seminested reaction was run for each human and bovine G- and P-type (19 type-specific reactions). For all seminested reactions, 2μL of amplicon from the first reaction were added to 18μL of master mix containing one of the initial primers and a type-specific primer at 200 nM each for a final reaction volume of 20μL. (See Table S4 for all primers and their concentrations and Table S5 for thermocycling conditions for each reaction.)

PCR products (20μL) were visualized by gel electrophoresis on 1.5% agarose gel (100 V for 90 min). A negative control and two positive controls [RotaTeq® vaccine-positive human fecal specimen and bovine CalfGuard® vaccine (Zoetis)] were included in each analysis batch along with the DNA ladder (ProMega). Gel bands matching specific genotypes were purified with illustra™ GFX PCR DNA and Gel Band Purification Kit (GE Healthcare), and identity was confirmed by sequencing. Direct sequencing of the amplicons was performed in both directions using the seminested reaction primers (see Table S4). We used the BigDye® Terminator V3.1 Cycle Sequencing Kit (Applied Biosystems) for the sequencing reaction, and the University of Wisconsin–Madison Biotechnology Center performed the reads on an ABI 3730xl DNA Analyzer. Consensus sequences were constructed with Lasergene (DNAStar) and submitted for identification using BLAST (National Center for Biotechnology Information, Bethesda, MD). Genotypes were used to classify all rotavirus group A detections as human or bovine for inclusion in human and bovine-specific outcome measures: G1P[8] and G10P[11] were considered human- and bovine-specific genotypes, respectively (Pitzer et al. 2011; Papp et al. 2013).

Samples positive for human-specific *Bacteroides* (HF183/BacR287; Green et al. 2014) or ruminant-specific *Bacteroides* (Rum-2-Bac; Mieszkin et al. 2010) were reanalyzed by PCR (676 bp amplicon) and sequencing, following the method of Bernhard and Field (2000), to confirm *Bacteroides* identity. *Bacteroides* DNA was extracted by the method described above and 6μL DNA extract was added to 14μL LightCycler 480 Probes Master including 500 nM of primers Bac32F and Bac708R (Bernhard and Field 2000). PCR commenced at 94°C for 5 min followed by 35 cycles consisting of 94°C for 30 s, 53°C for 1 min, and 72°C for 2 min, followed by a final 6-min extension at 72°C. PCR product (10μL) was visualized on 1.5% agarose gel. If the amplicon band was absent or faint, sensitivity was increased by reamplifying 1–6μL of amplicon under the same thermocycling conditions. Product purification from the gel, the sequencing reaction, and analyses were performed as described above for rotavirus A genotyping. Direct sequencing of the amplicons was performed in both directions using primers 32F and 708R.

Risk Factor Variables

Well construction variables were obtained from well driller reports filed at the Wisconsin Geological and Natural History Survey or Wisconsin Department of Natural Resources. Reports were available for 65% of sampled wells. As described above, initial well selection was stratified using existing depth-to-bedrock maps. However, for the statistical analyses, the exact depth-to-bedrock value for each well was obtained from its construction report. When a report was not available (n=116 and 135 for fall and summer sampling events, respectively), bedrock depth was estimated by interpolation from reports of nearby wells. Well elevation was obtained from the county digital elevation model.

Groundwater depth was measured continuously in U.S. Geological Survey monitoring well KW-183 (USGS 443535087345401 KW-25/24E/34-0183) and data are available in the USGS National Water Information System (USGS 2020). The well is located in Kewaunee County near an agricultural field. Relative to the ground surface, depth-to-bedrock is $2.1 \mathrm{m}$, borehole depth is $9.14 \mathrm{m}$, and casing depth is $3.05 \mathrm{m}$ (Muldoon and Bradbury 2010).

Groundwater recharge was estimated by the water table fluctuation method (Healy and Cook 2002), using graphical extrapolation of the antecedent recession curve and a specific yield of 0.04 based on previous assessments of recharge in the fractured rock in this area (Bradbury and Muldoon 1992). Cumulative recharge was obtained by summing individual recharge events for the 2-, 7-, 14-, and 21-d periods preceding sample collection.

Quantitative precipitation estimates (QPE) for each sampled well location (in 4-km grids) were provided by the North Central River Forecast Center of the U.S. National Weather Service. Because QPE values include snow, and frozen snow will not infiltrate soils, we excluded precipitation measurements for all well locations for days when snow without rain was recorded at the nearby National Weather Service station in Green Bay, Wisconsin. Cumulative precipitation was calculated by summing hourly QPE values over 2, 7, 14, and 21 d prior to sampling. Precipitation was not included in analyses of coliform and nitrate data because the synoptic design precluded variation in precipitation over the short time samples were collected.

Geographic Information System (GIS) data layers maintained by the Kewaunee County government reported locations of septic systems, agricultural fields, manure storages, and surface bedrock features. Agricultural field data included whether the field had a nutrient management plan (NMP) and therefore likely received manure applications.

Septic systems were divided into three categories for analysis: *a*) septic systems, included active systems of all types; *b*) drainfield, included inspected and uninspected systems that are designed to release effluent to the subsurface (i.e., excludes holding tanks); and *c*) not inspected, included only those systems that had not been inspected by county staff. Systems not in use were excluded from all three categories. The risk factor “distance to nearest septic system” excluded the system on the same property as the well, whereas counts of septic systems included the system on the same property.

Using ArcMap and Python script, fecal contamination sources and bedrock features were enumerated for each study well in two forms: *a*) distance from the well to the nearest contamination source or bedrock feature; and *b*) the count or areal size of the source or feature within three circular areas surrounding the well. The circular areas were defined by three radii from the well: 229, 457, and 914m (equal to 750, 1,500, and 3,000 ft, respectively), corresponding to 16, 66, and 262 ha (approximately 40, 160, and 640 acres). These area sizes were selected prior to data analysis based on an earlier study of septic system counts in similar-sized areas that were associated with childhood infectious diarrhea (Borchardt et al. 2003a).

Statistical Analyses

Stratified random sampling was employed to generate estimated contamination rates of coliforms, *E. coli*, and nitrate. Sampling strata were defined by depth-to-bedrock (<1.5, 1.5–6.1, and >6.1m). Smaller strata were oversampled relative to a simple random sample. This approach, in conjunction with the use of corresponding analytic weights and finite population correction factors in the analyses, resulted in more precise estimates for the smaller depth-to-bedrock strata without sacrificing the ability to estimate a countywide contamination rate. The analytic weight was defined as the product of the inverse of the sampling probability and the inverse of the response rate (i.e., the proportion of sampled well owners who agreed to participate in the study) within the appropriate depth-to-bedrock stratum. Rao-Scott likelihood ratio chi-square tests (Lohr 2010) were used to test associations between contamination rates and depth-to-bedrock as well as compare fall 2015 (groundwater recharge period) and summer 2016 (no recharge period) estimated contamination rates, both overall and within depth-to-bedrock strata. Statistical computations accounted for the complex sampling design.

Risk factors for well contamination were evaluated for independent variables relating to land use, precipitation, hydrogeology, bedrock, and well construction. Variables were tested for association with *a*) well contaminant detection and *b*) well contaminant concentration (among wells where contaminants were detected). Five contaminants (or contaminant groups) were tested for associations with risk factors: coliform bacteria, nitrate, human fecal markers, bovine fecal markers, and any fecal marker. Tests for coliform bacteria and nitrate associations were performed for each sampling period, groundwater recharge and no recharge.

For dichotomous (detect/nondetect) dependent variables, univariable screening for inclusion in the multivariable modeling process was performed using logistic regression. Each independent variable was represented as a linear (in the logit) term in the models. For independent variables with >10% zero values, a dichotomous (zero vs. greater than zero) term was included in the screening model in addition to the linear term. A plot of the estimated detection probability across the observed range of values for the independent variable being evaluated was also generated as part of the screening process. The same univariable screening process was performed for the well contaminant concentration dependent variables except that gamma regression with a natural log link function was used (Garson 2013), the model terms were linear in the log, and plots of estimated mean concentrations were generated.

For both univariable and multivariable analyses, outliers were excluded from the models for some of the concentration dependent variables. Specifically, 4 and 11 outliers were excluded from the analyses of coliform concentration for groundwater recharge and no recharge periods, respectively. And one, two, and four outliers were excluded for human, bovine, and any fecal marker concentration models, respectively. The criterion for excluding data points from the analyses was that their inclusion in the model caused the fitted curve to deviate meaningfully from the pattern exhibited by the remaining data. Concentration values for outliers were generally orders of magnitude larger than those in the remaining data points.

To be included in the multivariable model for a particular dependent variable, risk factors had to meet several criteria: *a*) strength of association (i.e., $p \leq 0.15$); *b*) plausibility, the association had to be biologically or physically possible; and *c*) internal consistency, where variables of the same measurement but at different levels (e.g., count of septic system drainfields within 229, 457, or 914m of a well) had similar directions of association (positive or negative) and strengths of association. When two variables of different measurements (e.g., well elevation and depth to bedrock) were correlated, the variable that most satisfied criteria 1, 2, and 3 was selected.

Additional screening was applied for inclusion in multivariable modeling when risk factors of the same measurement but at different levels were all associated with well contamination. Levels could differ in time (2, 7, 14, or 21 d) or area (within 229, 457, or 914m from a well). Under this situation, the risk factor with the greatest strength of association was selected. For example, 2-, 7-, and 14-d cumulative precipitation variables were all strongly associated with well contamination of human-specific markers. However, the 2-d cumulative precipitation variable had the largest regression coefficient and lowest *p*-value, so it was selected for inclusion.

Once the independent variables for a given multivariable model were identified, a screening process for interaction terms among these variables was undertaken. Only interactions deemed plausible and relevant were assessed. A screening model contained a term for the interaction and main effect terms for the individual risk factors comprising the interaction. As with the univariable screening of main effects, the independent variables comprising the interaction were represented as linear terms in the models; an interaction term was included in the multivariable model when its *p*-value was ≤ 0.15.

For multivariable analyses, the same procedure was used for both well contaminant detection and well contaminant concentration. Gamma regression was employed for all multivariable analyses of well contaminant concentration. Prior to performing multivariable regression analyses, each independent variable retained after the screening process was reassessed at the univariable level to establish whether a more complex representation than linear (e.g., quadratic or spline) would be appropriate in the multivariable model. To decide on an appropriate representation, a plot of the logit of the detection probability (log of the mean concentration) across the observed range of values for the independent variable was generated and examined, with the independent variable represented as a natural cubic spline (Hastie et al. 2001) in the corresponding logistic (or gamma) regression model. If a more complex representation was deemed appropriate, it was used in both main effect and interaction terms in the multivariable models.

All risk factors and interaction terms retained after the above screening processes were included in each final multivariable model. We did this in order that the independent effects of each risk factor could be evaluated in the presence of (i.e., adjusting for) the other model terms.

The final multivariable models were fit using log-binomial (or gamma) regression to facilitate interpretation of the results (McNutt et al. 2003). These models permit direct estimation of ratios of detection probabilities (or mean concentrations). This is in contrast to logistic regression models, which estimate ratios of odds rather than probabilities. When presence of the dependent variable is not rare (roughly <10%), which is typical in studies of well contaminant detection, the odds ratio does not closely approximate the corresponding ratio of detection probabilities and must be interpreted with caution.

For each multivariable model, procedures specific to generalized linear models were used to determine whether the information matrix was ill-conditioned (http://support.sas.com/kb/32/471.html). This approach entailed examining whether collinearity in the weighted risk factors was present, where the weights were determined by the model fitting algorithm.

Separate multivariable models for well construction risk factors were created because a number of wells were missing well construction reports. Had all risk factors been combined into a single model, only those wells without missing construction data would have been included, reducing statistical power to evaluate the other risk factors.

SAS version 9.4 was used to conduct all analyses (SAS Institute Inc.).
Go to:

Results and Discussion

Groundwater Levels during Sampling

Groundwater levels during the first study year followed the pattern typical for the upper Midwest with rising levels in the fall and spring and falling levels in the summer and winter (Figure 3). However, there was a prolonged recharge period from fall 2016 to spring 2017 (Figure 3). In January 2017, snowmelt raised groundwater levels during a long warm period (NOAA n.d.). Coliform and nitrate sampling corresponded with fall recharge (hereafter “recharge”) and with the summer decline when groundwater was at nearly its deepest level (hereafter “no recharge”). Sampling for microbial source tracking occurred during recharge (3 events) and no-recharge (2 events) periods.

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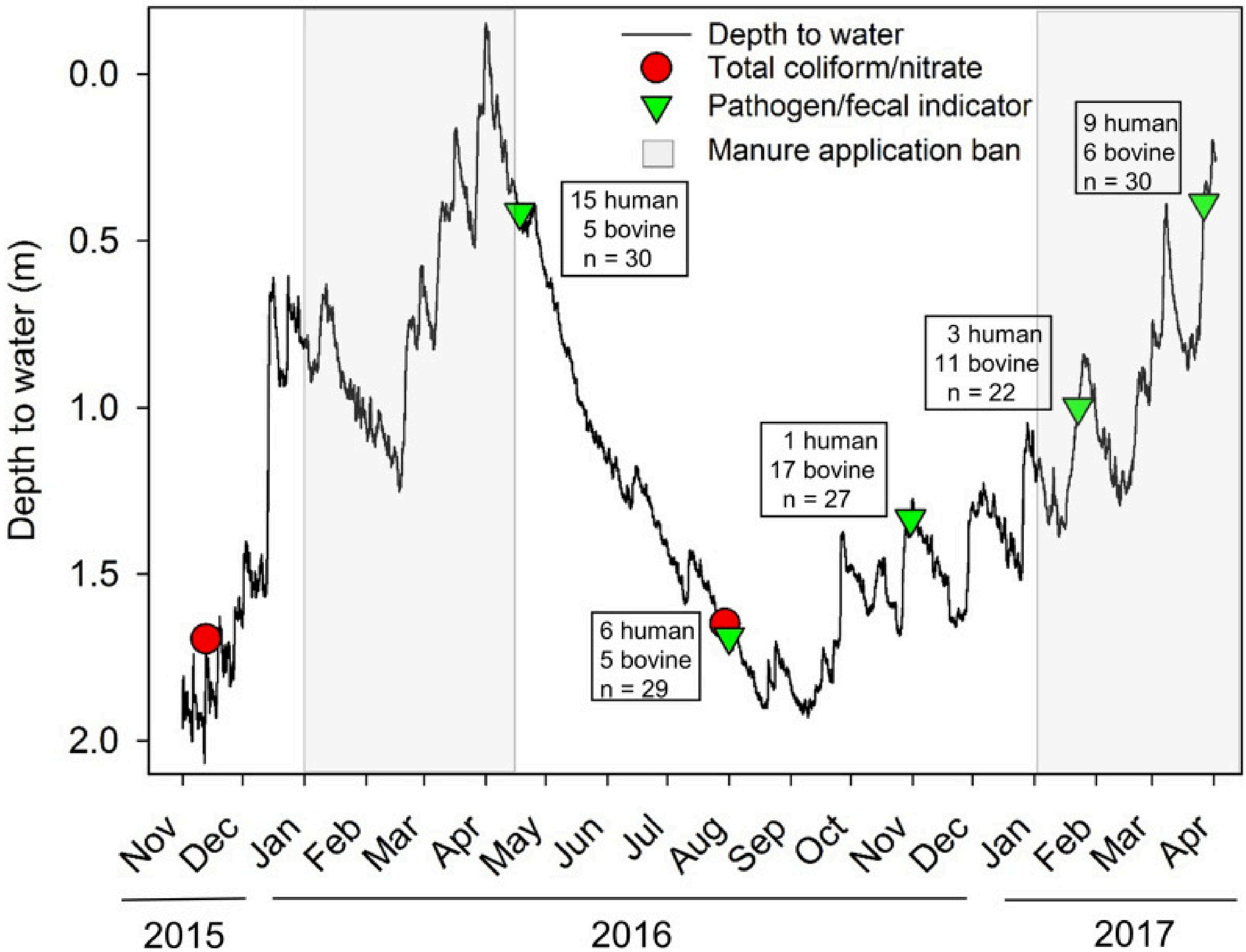

Figure 3.
Sampling periods in relation to groundwater level in Kewaunee County monitoring well KW-183 (USGS 443535087345401; USGS 2020). Sampling times indicated by red circles (total coliforms and nitrate) and green triangles (pathogens and fecal indicators). Boxes indicate the number of wells positive for human-specific or bovine-specific markers; $n = \text{total number of wells sampled}$. Gray shaded areas designate seasonal manure application ban for fields with bedrock depths $< 6.1\text{m}$.

Bacteria and Nitrate Contamination Rates

The countywide private well contamination rates for coliforms, *E. coli*, and NO_3^--N>10mg/L were similar to the average rates for the state of Wisconsin (Table 1). However, for wells in the two shallowest bedrock depth strata (<1.5m and 1.5–6.1m), contamination rates were generally greater than the statewide averages, and rates were consistently greater than rates for wells in the deepest stratum (>6.1m to bedrock). The greater the bedrock depth and transport distance through surficial sediments, the less likely these contaminants will reach bedrock fractures that allow rapid transport (Final Report of the Northeast Wisconsin Karst Task Force 2007; Rasmuson et al. 2020).

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Table 1

Estimated contamination rates (percent positive wells) for total coliform bacteria, *Escherichia coli*, or nitrate-N>10mg/L.

Sampling period or reference data	Region or depth-to-bedrock category[a]	Number of wells sampled	Percent positive wells (95% confidence interval)			
			Total coliforms	*E. coli*	Nitrate-N>10mg/L	Total coliforms or nitrate-N>10mg/L
Groundwater recharge	<1.5m to bedrock	26	46 (30, 63)	4 (0, 9)	7 (0, 15)	50 (34, 66)
	1.5–6.1m to bedrock	120	28 (18, 37)	1 (0, 2)	20 (7, 33)	42 (28, 55)
	>6.1m to bedrock	167	19 (11, 26)	0.3 (0, 0.6)	6 (1, 10)	23 (15, 31)
	Kewaunee County	313[b,c] 316[c,d]	21 (14, 27)	0.4 (0.1, 0.7)	7 (3, 11)	26 (19, 34)
No groundwater recharge	<1.5m to bedrock	24	23 (6, 39)	7 (0, 15)	10 (0, 20)	33 (12, 53)
	1.5–6.1m to bedrock	122	29 (16, 41)	1 (0, 3)	19 (9, 28)	40 (28, 53)
	>6.1m to bedrock	252	21 (15, 27)	26 (19, 32)		
	Kewaunee County	396[b,c] 400[c,d]	22 (17, 28)	1 (0.1, 2)	7 (4, 10)	28 (22, 33)
Reference data	Wisconsin[e]	534	23	3	7	—
	Wisconsin[f]	3,838	18	—	10	—

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Note: —, no data available. Estimates and corresponding 95% confidence intervals account for the stratified random sampling design employed in the study.

[a]The estimated number of wells in each bedrock depth category are 76, 575, and 4,156 wells at <1.5, 1.5–6.1, and >6.1m, respectively, totaling 4,807 wells in Kewaunee County. Our final estimates of the number of wells in each bedrock depth category are different than the initial estimates at the study beginning using the bedrock map created by Sherrill (1979).
[b]*n* for coliforms and *E. coli*.

[c]The *n*'s do not equal the number of samples analyzed (see Figure S1) because some wells had missing depth-to-bedrock values (six wells for the groundwater recharge period and one well for the no recharge period) for which analytic weights could not be generated.

[d]*n* for nitrate.

[e]Data for private wells; U.S. General Accounting Office 1997.
[f]Knobeloch et al. 2013.
Groundwater recharge and no-recharge periods did not have significantly different contamination rates, regardless of contaminant type or level of data aggregation (Table 1). There was one exception; coliform contamination during recharge was greater than the no-recharge period for wells with bedrock depths <1.5m (p=0.042).

Table 2 reports descriptive statistics for coliforms, *E. coli*, and nitrate-N concentrations of positive samples. In both recharge and no-recharge periods, 25% of wells positive for nitrate-N had concentrations greater than 9mg/L.

Table 2

Descriptive statistics of coliform bacteria, *Escherichia coli*, and nitrate concentrations.

Sampling period	Measurement	Number of positive samples	Number of non-detects	Concentration of positive samples[a]					
				Mean	Median	Minimum	25th percentile	75th percentile	Maximum[b]
Groundwater recharge	Coliforms	87	232	73.2	5.2	1.0	2.0	17.3	>2,419.6
	E. coli	5	314	5.0	2.0	1.0	2.0	4.1	16.1
	Nitrate-N	203	119	6.3	4.7	0.2	1.6	9.0	29.7
No groundwater recharge	Coliforms	87	310	116.8	6.2	1.0	2.0	55.4	>2,419.6
	E. coli	10	387	105.0	3.1	1.0	1.3	8.8	1011.2
	Nitrate-N	205	196	6.5	5.2	0.2	2.1	9.1	33.3

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Note: MPN, most probable number.

[a]Coliforms and *E. coli*, MPN/100mL; nitrate-N, mg/L.

[b]2,419.6 MPN/100mL was the upper limit of quantification.

Coliforms, although nonpathogenic, are the standard indicator of drinking-water sanitary quality in the United States. Studies of coliform-positive private wells have observed (DeFelice et al. 2016) and not observed (Strauss et al. 2001) associations with acute gastrointestinal illness. High nitrate in drinking water can cause methemoglobinemia, and in some studies it has been linked with colorectal cancer, thyroid disease, and central nervous system birth defects (Ward et al. 2018). The U.S. National Primary Drinking Water Standards apply only to public water systems, not private wells. Nonetheless, the U.S. drinking water Maximum Contaminant Level Goals (MCLG) for coliforms and nitrate-N provide public health benchmarks, which are zero and 10mg/L, respectively (U.S. EPA 2020). Multiplying the MCLG exceedance rates for coliforms or nitrate-N (Table 1) by the estimated number of wells in each bedrock depth category in Kewaunee County [76, 575, and 4,156 wells at <1.5, 1.5–6.1, and >6.1m, respectively (Borchardt et al. 2019)], we estimate approximately 1,300 wells (27%) during the study period did not meet U.S. EPA public health goals for safe drinking water.

Calculating well contamination rates by county, state, or other governmental units has the advantage of matching policy-making jurisdictions. However, aggregating data in this manner can overlook factors underlying contamination "hotspots," in this case, bedrock depth. For example, the statewide averages for coliform and nitrate MCLG exceedances in Wisconsin, irrespective of bedrock depth, are 18% and 10%, respectively (Knobeloch et al. 2013). Using the multivariable models for coliforms and nitrate for recharge and no-recharge periods, respectively (see below and Figures 4B and and4C),4C), the statewide percentages are equivalent to detection probabilities at bedrock depths of 10m (coliforms) and 14m (nitrate) in Kewaunee County. We estimate the number of wells with shallower bedrock depths, and therefore higher detection probabilities than the statewide averages, to be 1,562 (coliforms) and 2,464 (nitrate), which is 32% and 50% of the county's private wells. This assessment is consistent with the high rates of coliform and nitrate exceedances for carbonate aquifers (e.g., Silurian dolomite) and agricultural areas observed in private well data nationally (DiSimone 2009).

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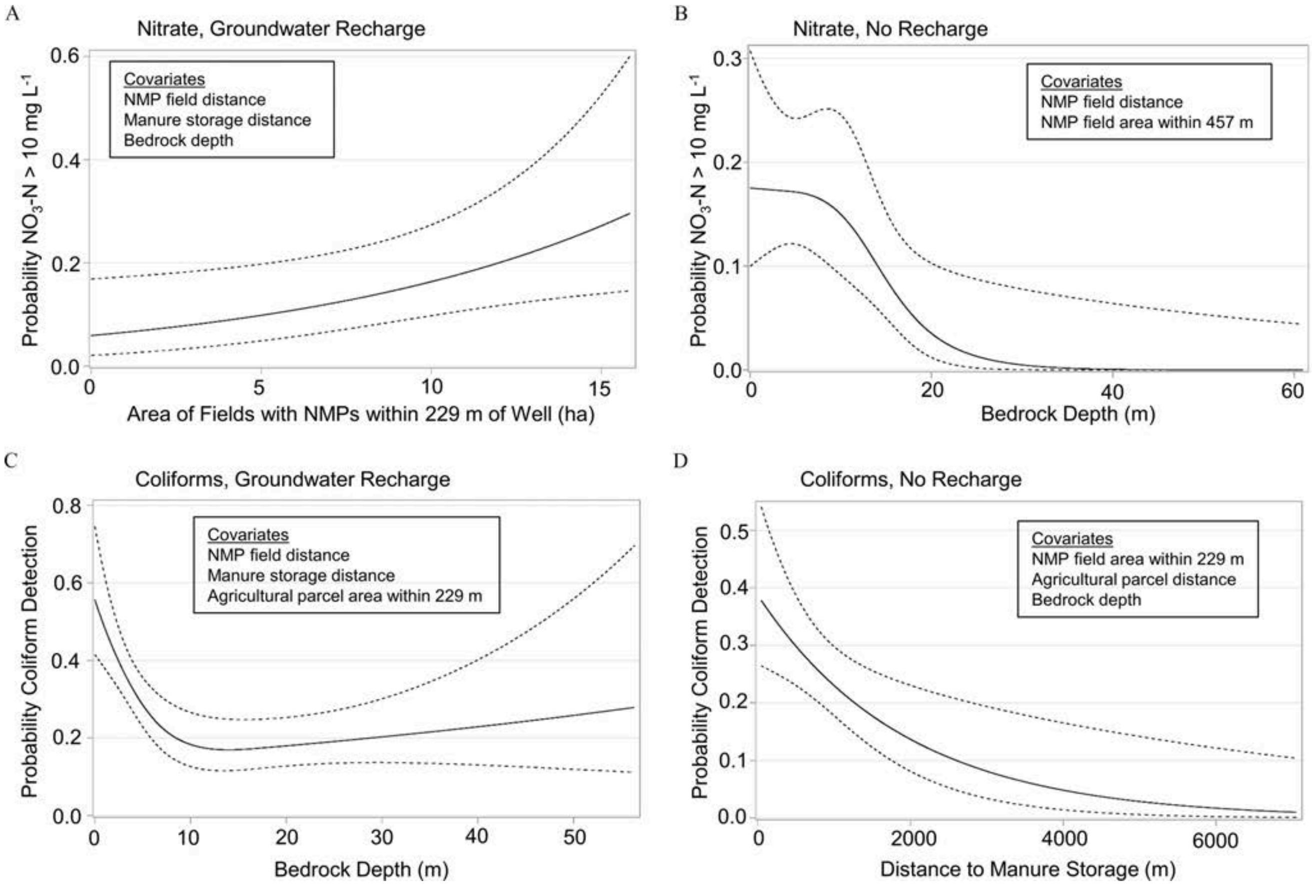

Figure 4.
Detection probabilities for NO3-N>10mg/L and coliform bacteria in private wells regressed (log-binomial) on key risk factors during groundwater recharge and no recharge periods. Coefficients and *p*-values are reported in Table 4. Black line: estimated probability of detection. Dashed lines: 95% pointwise confidence limits. Covariates in the multivariable models were fixed at their median values for the purpose of plotting. Fields with NMPs likely receive manure and inorganic fertilizer inputs. Note: NMP, nutrient management plan.

Microbial Source Tracking and Pathogen Occurrence

Of 138 samples from 131 wells, 82 samples (59%) from 79 wells (60%) were positive for markers of fecal-associated microbes (Table 3). Among the 79 wells with fecal contamination, 32 wells had markers for pathogens that could infect humans (human-specific and zoonotic pathogens without host specificity). Seventy wells were positive for two or more markers. Well water concentrations of fecal-associated markers were generally low; *Bacteroidales*-like CowM2 and *Bacteroidales*-like CowM3 had the highest median concentrations (Table 3).

Table 3

Gene markers of fecal-associated microbes detected in samples (n=138) from private household wells (n=131).

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Host specificity	Microbe[a]	Gene marker[b]	Number of positive wells[c]	Number of positive samples[c]	Concentration of positive samples (gene copies/L)	
					Median	Range
Human-specific	*Bacteroidales*-like Hum M2	Glycosyl hydrolase family 92	7	8	4	<1–1,050
	Human *Bacteroides*	16s rRNA (HF183/BacR287)	27	28	<1	<1–34
	Cryptosporidium hominis	18s rRNA	1	1	<1	<1
	Adenovirus A	hexon	1	1	1	1
	Rotavirus group A, G1 P[8]	*NSP3*	7	7	<1	<1–3
	Rotavirus group A, G1 P[8]	*VP1*	3	3	1	<1–22
	Any human marker	—	33	34	<1	<1–1,050
Bovine- or ruminant-specific	*Bacteroidales*-like Cow M2	DHIG domain protein	2	2	472	29–915
	Bacteroidales-like Cow M3	HD super family hydrolase	4	4	174	3–49,818
	Ruminant *Bacteroides*	16s rRNA (Rum-2-Bac)	36	36	1	<1–42,398
	Bovine polyomavirus	*VP1*	8	8	4	<1–451
	Bovine enterovirus	5’ non-coding region	1	1	2	2
	Rotavirus group A, G10 P[11]	*NSP3*	12	12	12	2–4,481
	Rotavirus group A, G10 P[11]	*VP1*	5	5	23	<1–732
	Any bovine or ruminant marker	—	44	44	3	<1–49,818

Host specificity	Microbe[a]	Gene marker[b]	Number of positive wells[c]	Number of positive samples[c]	Concentration of positive samples (gene copies/L)	
					Median	Range
No host specificity	Pepper mild mottle virus	replication-associated protein	13	14	14	2–3,811
	Cryptosporidium spp.	18s rRNA	2	2	<1	<1–1
	Cryptosporidium parvum	18s rRNA	13	13	<1	<1−14
	Giardia duodenalis group B	β-giardin	2	2	<1	<1
	Campylobacter jejuni	*mapA*	1	1	<1	<1
	Salmonella spp.	*invA*	3	3	6	<1–13
	Salmonella spp.	*ttr*	5	5	10	5–59
	E. coli (pathogenic)	*eae*	1	1	4	4
	Shiga toxin producing bacteria	*stx1*	1	1	16	16
	Shiga toxin producing bacteria	*stx2*	1	1	1	1
	Rotavirus group C	*VP6*	3	3	50	45–1,301
	Any nonspecific marker	—	37	46	5	<1–3,811
All	Any fecal marker	—	79	82	2	<1–49,818

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Note: —, Any of the gene markers within the specified group.

[a]Microbial markers analyzed but not detected: human adenovirus groups B, C, D, and F; human enterovirus; human norovirus genogroups I and II; human polyomavirus; *Cryptosporidium bovis*; bovine adenovirus; bovine coronavirus; and bovine viral diarrhea virus types 1 and 2.

[b]Primers, probes, and references for qPCR assays are reported in Table S1.

[c]Totals are less than the sum of individual markers because some wells and samples were positive for more than one marker.

The 60% fecal contamination rate could be an overestimate because we limited well selection to those wells previously positive for coliforms or with nitrate-N concentrations >10mg/L to favor successful completion of the study objective, that is, identify fecal sources of contamination. On the other hand, 60% could be an underestimate, because 95% of the wells were sampled only once, and detection probability was shown to increase the more frequently a well was sampled in one study (Atherholt et al. 2015).

Comparing the fecal contamination rate of our study wells with rates from other studies is confounded by differences in hydrogeological setting, well type, sampling season, the number of wells, the number of samples per well, and the types and number of fecal microorganisms tested. Five studies approximate our study design, setting, or type and number of fecal microbes and can provide some context. Among 50 private wells in seven hydrogeological districts of Wisconsin, 8% were positive for human enteric viruses (Borchardt et al. 2003b). Private wells completed in fractured Silurian dolomite in Ontario, Canada (11 wells), and fractured bedrock in Pennsylvania, USA (5 wells), had microbes of fecal origin in 45% and 100%, respectively (Allen et al. 2017; Murphy et al. 2020). Ninety-six percent of public wells tested in Minnesota, USA, for similar types and number of fecal organisms were positive (Stokdyk et al. 2020), and, as in the present study, *Cryptosporidium* was the most frequently detected pathogen, suggesting it is more common in groundwater than previously thought (Stokdyk et al. 2019). Last, in a comprehensive review of groundwater studies conducted in Canada and the United States, Hynds et al. (2014a) reported that of 12,616 public and private wells tested, at least one enteric pathogen was detected in 15%. Although comparisons among studies are abstruse, the weight of evidence suggests fecal contamination of drinking water wells is not uncommon.

Fecal contamination stemmed from both human wastewater and bovine manure sources. Human wastewater was present in 33 wells, and bovine manure was present in 44 wells (Table 3). Nine wells were contaminated by both fecal sources, human and bovine. Of the 37 wells (46 samples) positive for nonspecific markers, 11 wells (13 samples) did not have coincident detections for human- or bovine-specific markers, indicating that for these wells and samples the fecal source was unknown.

Previous studies have found human-specific and bovine-specific *Bacteroidales* genetic markers detected together in the same private wells (Krolik et al. 2014; Felleiter et al. 2020) and wells and springs (Diston et al. 2015). Nine private wells completed in the dolomite aquifer of six Wisconsin counties were positive for *Bacteroidales* markers specific to human, bovine, or swine fecal material (Zhang et al. 2014).

Identifying which fecal source, human or bovine, was the greatest contributor to groundwater fecal contamination in the county is not possible from our MST data. The proportion of samples positive for human or bovine markers varied by sampling period, which is to say by season, groundwater level, and timing of manure applications (Figure 3). Beginning 1 January 2016, Kewaunee County banned manure applications during the 1 January–15 April period on all fields with bedrock depths <6.1m. The proportion of wells positive for bovine-specific markers likely depends on the timing and location of well sampling relative to the ban regulations. Groundwater recharge is also important (see below). Therefore, both human and bovine fecal sources contribute to contamination, and the fecal source that appears to bear the most responsibility for contamination depends on sample timing.

Human-specific HF183 *Bacteroides* (28 samples) and ruminant *Bacteroides* (36 samples) were the most common fecal markers, and all samples positive for these were successfully sequenced to confirm *Bacteroides* host identities (see Table S6, Table S7). The Rum-2-Bac marker is specific to ruminants, not cattle alone. However, two lines of evidence suggest the detected Rum-2-Bac markers were indeed from dairy manure: *a*) All amplicons (676 bp) from Rum-2-Bac-positive samples matched *Bacteroidales* or *Bacteroides* species from cattle feces with percent identities greater than 98% and E-scores of zero; and *b*) The only other abundant ruminants in Kewaunee County are approximately 16,000 white tail deer (Wisconsin Department of Natural Resources 2018). Deer excrete 261g/d fecal material (McCullough 1982), which for the Kewaunee County landscape equals 1.3×106 kg/y. In comparison, the land-applied cattle manure in the county is 1.76×109 kg/y (see Supplemental Material, Cattle Manure Volume Produced Annually in Kewaunee County), more than 1,000 times greater than that of deer, suggesting the more probable groundwater contaminant is cattle manure.

Rotavirus group A subtyping was successful for distinguishing human from bovine fecal sources in our study, but that may not always be possible. The human rotavirus vaccine, RotaTeq, contains five human–bovine reassortment strains (Matthijnssens et al. 2010), and because the G6 (bovine) strain can be shed in human stool after oral vaccination (Higashimoto et al. 2018), the fecal source cannot be distinguished when that strain is detected (i.e., vaccine shed into septic systems or G6 wild type in dairy manure). However, our study wells were not positive for the G6 strain, because subtyping analysis revealed rotavirus G1 [P8], which is typically associated with human rotavirus infection (Pitzer et al. 2011), or G10 [P11], a subtype associated with rotavirus infections in cattle (Papp et al. 2013). (Two wells were positive for both subtypes.) Whether the G1 [P8] rotavirus we detected is wild type or vaccine is uncertain, but it indicates a human fecal source regardless.

The human pathogens we detected in private wells have been previously reported in groundwater, except rotavirus group C. Rotavirus group C is zoonotic (unlike group A) and has been found in American cattle and children (Tsunemitsu et al. 1992; Jiang et al. 1995). One-third of young adults in the United States may experience infection in their lifetimes (Riepenhoff-Talty et al. 1997). Twenty wells (15%) were positive for rotaviruses (groups A and C), and rotavirus group C and bovine-related rotavirus group A had the highest concentrations (Table 3), suggesting groundwater in northeastern Wisconsin may be a common reservoir for the sharing and possible reassortment of rotavirus strains among people and cattle.

Risk Factors for Private Well Contamination–Univariable Association Tests

All univariable association tests between private well contamination outcomes and risk factors are reported in Tables S8–S13. Summary statistics of risk factor values are reported in Tables S14–S16.

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Sinkholes and rock ledges were associated with well contamination of all five investigated contaminants (coliforms, nitrate, human-specific, bovine-specific, and any fecal markers), but these risk factors were excluded from multivariable analyses for several reasons: *a*) sinkholes and rock ledges were highly correlated with bedrock depth; *b*) sinkhole and ledge locations were determined by field inspections by county staff, and 20% of fields had not been inspected; and *c*) inspections did not include residential properties, biasing the data toward agricultural fields.

Risk Factors for Well Contamination with Nitrate or Coliforms

All land use risk factors eligible for multivariable modeling of nitrate and coliform contamination were related to agriculture (Table 4), suggesting agricultural activities were the primary sources for these contaminants. Septic system density in univariable tests was, at times, associated with coliform and nitrate contamination (see Tables S8 and S9). However, the associations were negative (i.e., implausible and therefore not eligible for model inclusion), likely because more land with housing and septic systems meant there was less land nearby with agricultural activities. Rayne et al. (2019) made a similar observation, showing that when an agricultural field near Madison, Wisconsin, was developed into a housing subdivision with septic systems, the number of monitoring wells with NO_3^--N>10mg/L declined.

Table 4

Multivariable modeling of land use and bedrock risk factors as related to detection probabilities and concentrations of coliforms and nitrate in private wells.

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Sampling period	Contaminant and outcome measurement[a] (n)[b]	Risk factor	Univariable model p-value	Multivariable model			
				Risk factor median[c]	Risk factor range[c]	Coefficient or trend[d,e]	p-value[f]
Groundwater recharge	Coliforms detection (315)	Bedrock depth	0.0090	7.6	0–56.4	Negative	0.0001
		NMP field distance[g]	0.036	42	0–723	−0.002	0.20
		Manure storage distance	0.14	899	46–3,728	−0.00008	0.63
		Agricultural field area within 229m	0.072	12.7	0–16.4	−0.008	0.77
	NO−3-N>10mg/L detection (318)	NMP field area within 229m	0.0013	7.1	0–15.9	0.1	0.024
		NMP field distance	0.14	42	0–724	0.002	0.38
		Manure storage distance	0.082	928	46–3,728	−0.0002	0.49
		Bedrock depth	0.0028	7.6	0–56.4	Negative	0.082
	NO−3-N concentration (200)	NMP field area within 914m	0.071	141.7	10.2–235.7	Positive	0.29
		Bedrock depth	0.0063	5.0	0–56.4	Negative	0.0065

Sampling period	Contaminant and outcome measurement[a] (*n*)[b]	Risk factor	Univariable model p-value	Multivariable model			
				Risk factor median[c]	Risk factor range[c]	Coefficient or trend[d,e]	p-value[f]
No groundwater recharge	Coliforms detection (395)	Manure storage distance	0.0014	878	48–7,054	−0.0005	0.0062
		Agricultural field distance, dichotomous	0.15	NA	NA	0.3	0.24
		Agricultural field distance, continuous	0.081	24	0–805	−0.003	0.34
		NMP field area within 229m	0.059	7.4	0–15.6	0.008	0.75
		Bedrock depth	0.12	12.2	0–61	−0.006	0.42
	Coliforms concentration (76)	NMP field distance	0.0026	36	0–554	Negative	0.0050
	NO_3^--N>10mg/L detection (399)	Bedrock depth	<0.0001	12.2	0–61	Negative	0.021
		NMP field area within 457m	0.014	33.3	0–62.4	0.008	0.48
		NMP field distance	0.082	40	0–836	−0.001	0.66

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Note: NA, Not applicable; NMP, nutrient management plan. Univariable model *p*-values used for selecting risk factors are included for reference; complete univariable statistics are provided in Tables S8 and S9. Risk factor eligibility for inclusion in multivariable models is described in statistical methods.

[a]Univariable analyses for: *a*) coliform concentration, groundwater recharge; and *b*) nitrate concentration, no recharge, showed no eligible variables for multivariable modeling; therefore, these models are missing from the table.

[b]n=number of samples in multivariable model.

[c]Units for distance and depth are meters; area is hectares.

[d]In lieu of reporting multiple coefficients for spline-represented variables, we report the overall trend (positive or negative).

[e]Interpretation of coefficient linear terms: change in ln(detection probability) or change in ln(concentration) for a unit change in the risk factor.

[f]The composite *p*-value is reported for spline-represented variables.

[g]Fields with NMPs likely receive manure and inorganic fertilizer inputs.

The area of fields with NMPs within 229m was positively associated with having a well with NO_3^--N>10mg/L during groundwater recharge. This association was adjusted for three other risk factors: distance to manure storage, distance to NMP field, and bedrock depth (Table 4). For instance, wells surrounded by 15 ha of NMP fields within 229m, compared with zero hectares, had a 458% increase in the probability of having NO_3^--N concentrations >10mg/L (27.2% vs. 5.9%) (Figure 4A). Approximately 80% of the agricultural field area in Kewaunee County follows NMPs (D. Bonness, Kewaunee County Land and Water Conservation Director, personal communication). Because we did not have data on manure and inorganic fertilizer applications, we used county records of NMPs to identify fields likely receiving these inputs.

During the no-recharge period, bedrock depth had the strongest association with the detection of wells with NO_3^--N>10mg/L (adjusted for distance to NMP field and area of NMP fields within 457m). Wells with bedrock depths ≥40m had nearly 0% probability of NO_3^--N>10mg/L compared with 18% probability for wells with bedrock depths of zero (Figure 4B). Bedrock depth was also a significant risk factor for nitrate concentrations in wells during recharge (Table 4).

In a U.S. nationwide study of nitrate in 1,230 wells, Nolan (2001) identified risk factors within 500-m radii encircling wells and tested associations by multivariable logistic regression, an approach similar to ours. Significant risk factors were nitrogen fertilizer loading, percent cropland, population density, percent well-drained soils, depth to the seasonally high water table, and rock fractures within an aquifer. Our results are consistent with other studies that have associated groundwater nitrate contamination with agricultural-related risk factors, including agricultural land use (Eckhardt and Stackelberg 1995; Lichtenberg and Shapiro 1997; Nolan and Hitt 2006; Lockhart et al. 2013; Zirkle et al. 2016), animal feeding operations (Toetz 2006; Wheeler et al. 2015), dairy manure lagoons (Lockhart et al. 2013), and swine manure lagoons (Messier et al. 2014), but contrast with studies that associated nitrate with septic systems (Lichtenberg and Shapiro 1997; Gardner and Vogel 2005). Our study differs from previous nitrate work in that we dichotomized the nitrate outcome for log-binomial regression using the U.S. EPA health-based MCLG as the threshold; other studies used much lower thresholds, 4mg/L or lower (Eckhardt and Stackelberg 1995; Nolan 2001; Gardner and Vogel 2005).

Coliforms multivariable modeling showed the primary risk factors for detection were bedrock depth during groundwater recharge and distance to the nearest manure storage during the no-recharge period (Table 4). The concentration of coliforms was associated with only one risk factor: distance to the nearest NMP field (Table 4).

Coliform detection in wells during recharge became less likely the deeper the bedrock to depths of 10m (Figure 4C). Wells in locations with 10-m bedrock depth were 67% less likely to have coliform detections in comparison with wells with bedrock at the land surface (18.3% vs. 55.6%).

During the no-recharge period, coliform detection decreased with increasing distance between private wells and manure storage sites (Figure 4D). For example, in comparison with wells located 48m from manure storage (the minimum distance observed), the coliform detection probability for wells 4,000m distant decreased 87% (37.8% vs. 4.8%). Distance to manure storage was also a covariate in the multivariable models for coliform detection and nitrate detection during groundwater recharge (Table 4).

According to records maintained by the Kewaunee County Land and Water Conservation Department, there are 277 manure storage structures in the county, mostly lagoons ranging in size from 0.01 to 2.06 ha and typically 3.7m deep. Lagoon design specifications allow bottom leakage rates of 47,000L/ha/d (NRCS 313), equivalent to 3.4×107L/y for a 2-ha lagoon. Coliform concentrations in dairy manure are on the order of 106 CFU/g wet manure (Blaustein et al. 2015). Groundwater velocities in the Silurian dolomite fractures have been measured as high as 115 to 600m/d (Bradbury and Muldoon 1992; Bradbury et al. 2001), suggesting leaked manure could deliver coliforms to private wells 1,600m distant (1 mi) in 3 to 14 d.

However, one confounder to consider is a possible negative association between manure storage distance and land-applied manure volume. Transporting manure by tanker truck for land application is costly and time-consuming. More distant fields may receive less land-applied manure. Data on manure application volumes and locations in Kewaunee County are sparse, so discriminating between mechanisms (lagoon leakage vs. applied manure volume) is not possible.

Although we cannot identify the mechanism underlying the association between coliform contamination and manure storage, the relationship is consistent with previous studies (Li et al. 2015; Yessis et al. 1996). Previous studies have also linked the occurrence of coliforms and other indicator bacteria in wells to other agriculture-related factors, including proximity to farm animal operations (Allevi et al. 2013) or agricultural point sources (e.g., farmyards, animal holding facilities, manure storage) (Hynds et al. 2014b; Fennell 2017; Goss et al. 1998; Li et al. 2015) and the density of livestock (Invik et al. 2019; O'Dwyer et al. 2018). Moreover, Óhaiseadha et al. (2017) showed that laboratory-confirmed verotoxigenic *E. coli* infections in Ireland were positively associated with private well usage and cattle density. Our study differed from previous work in that we used GIS to measure continuous-scaled (i.e., not dichotomous or ordinal) "distance to" and "area of" agricultural activities with respect to study well locations.

Risk Factors for Well Contamination with Human Fecal Markers

Human fecal contamination of private wells was modeled with four variables, of which the median groundwater depth 14 d prior to sampling had the strongest association with contamination (Table 5). For example, the detection probability for human fecal contamination increased to 35% from 11%, with a 1.4-m decrease in median groundwater depth 14 d prior to sampling. Density of neighboring septic system drainfields was another risk factor. These two risk factors are in agreement with the fact that septic systems are the primary source of human fecal wastes on the rural county landscape, and that shallower groundwater depth gives microbes shorter travel distance from the bottom of septic drainfields to the top of the groundwater table. Likewise, bedrock depth, which reflects the distance microbes must travel to reach the fractured bedrock, was associated with the concentration of human markers (Table 5).

Table 5

Multivariable modeling of land use and bedrock risk factors as related to detection probabilities and concentrations of genetic markers of host-specific and fecal-associated microbes in private wells.

Fecal marker source and outcome measurement (n)[a]	Risk factor	Univariable model p-value	Multivariable model			
			Risk factor median[b]	Risk factor range[b]	Coefficient or trend[c,d]	p-value[e]
Human marker detection (137)	Drainfield septic systems, count within 229m	0.038	2	0–10	0.09	0.11
	Groundwater depth, 14-d antecedent, median	0.0003	1.2	0.3–1.6	−0.9	0.011
	Rainfall, 2-d antecedent, cumulative	0.0093	14	0–37	Positive	0.69
	Bedrock depth	0.051	6.1	0–46.6	Negative	0.13
Human marker concentration (33)	Bedrock depth	0.011	4.3	0.3–36.6	Negative	0.011
Bovine marker detection (138)	Groundwater recharge, 7-d antecedent, cumulative	0.0041	50	0–60	Positive	0.0092

Fecal marker source and outcome measurement (n)[a]	Risk factor	Univariable model p-value	Multivariable model			
			Risk factor median[b]	Risk factor range[b]	Coefficient or trend[c,d]	p-value[e]
Bovine marker concentration (41)	Agricultural field area within 229m	0.029	11.6	3.7–16.4	Positive	0.024
	Bedrock depth	0.0019	5.2	0–29	−0.1	0.0006
Any fecal marker[f] detection (137)	Drainfield septic systems, count within 229m	0.0036	2	0–10	Positive	0.036
	Rainfall, 2-d antecedent, cumulative	0.12	14	0–37	Positive	0.19
	Manure storage distance[g]	0.94	687	71–3,728	−0.0004	0.036
	Bedrock depth	0.027	6.1	0–46.6	−0.06	0.0058
	Manure storage distance times bedrock depth interaction	0.045	NA	NA	Negative	0.024
Any fecal marker concentration (77)	Agricultural field area within 229m	0.035	12.7	1.1–16.4	Positive	0.097
	Manure storage distance	0.083	762	113–3,728	−0.0001	0.76
	Bedrock depth	0.0003	4.6	0–36.6	−0.08	0.002

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Note: NA, Not applicable. Univariable model *p*-values used for selecting risk factors are included for reference; complete univariable statistics are provided in Table S10. Risk factor eligibility for inclusion in multivariable models is described in statistical methods.

[a]n=number of samples in multivariable model.

[b]Units for distance and depth are meters; rainfall and recharge are millimeters; area is hectares.

[c]In lieu of reporting multiple coefficients for spline-represented variables we report the overall trend (positive or negative).

[d]Interpretation of coefficient linear terms: change in ln(detection probability) or change in ln(concentration) for a unit change in the risk factor.

[e]The composite *p*-value is reported for spline-represented variables.

[f]"Any fecal marker" includes all microorganisms regardless of host specificity.

[g]Included in multivariable model because of its significant interaction with bedrock depth.

One other possible human fecal source was septage (i.e., wastewater pumped from septic tanks) land-applied to approved agricultural fields. Tests of association between septage-applied fields and well contamination were ambiguous, suggesting septage was not an important risk factor (see Supplemental Material, Septage Land-Applied Fields—Univariable Associations). County records show during the study period only 10 fields equaling 110 ha received 2.57×106L septage. In contrast, septic systems are located throughout the county and the volume of untreated effluent released to the subsurface was calculated to be 6.79×108L per year (see Supplemental Material, Septic System Effluent Volume Released Annually in Kewaunee County).

Septic system effluent contamination of groundwater with fecal indicator bacteria and pathogenic viruses and bacteria is well documented in the literature (Hagedorn et al. 1981; Yates 1985: Nicosia et al. 2001; Katz et al. 2010; Hynds et al. 2012; Lusk et al. 2017). In one study, vaccine poliovirus was introduced into the tank of a new conventional septic system, and the virus was cultured in multiple samples over time in a monitoring well 6m down-gradient from the edge of the drainfield (Alhajjar et al. 1988). More recently, detection in groundwater of the human-specific markers HF183 and HumM2 has been linked with septic system effluent (Schneeberger et al. 2015; Murphy et al. 2020). Groundwater-borne disease outbreaks (Yates 1985; Beller et al. 1997; Borchardt et al. 2011) and endemic diarrheal illness (Borchardt et al. 2003a) have also been associated with septic systems.

As early as 1977 the U.S. EPA recommended that to minimize groundwater contamination septic system density should not exceed 40 systems per square mile (1 system/6.5 ha or 0.15 systems/ha) (U.S. EPA 1977). Three subsequent studies have suggested septic system density should not exceed 5, 1–2.5, and 3.5–6 systems/ha (Reneau 1979; Gardner et al. 1997; Morrissey et al. 2015). In the fractured dolomite aquifer of our study, as the number of septic drainfields within 229m of private wells increased from zero to 10, the probability of human fecal contamination increased 2.5 times, from 13% to 33% (Figure 5A), with the upper limit (10 septic drainfields) equivalent to 0.6 systems/ha. This relationship was adjusted for groundwater depth, rainfall, and bedrock depth (Table 5). (In Figure 5A the count of one drainfield represents the well contamination probability from a household's own drainfield, 14%.)

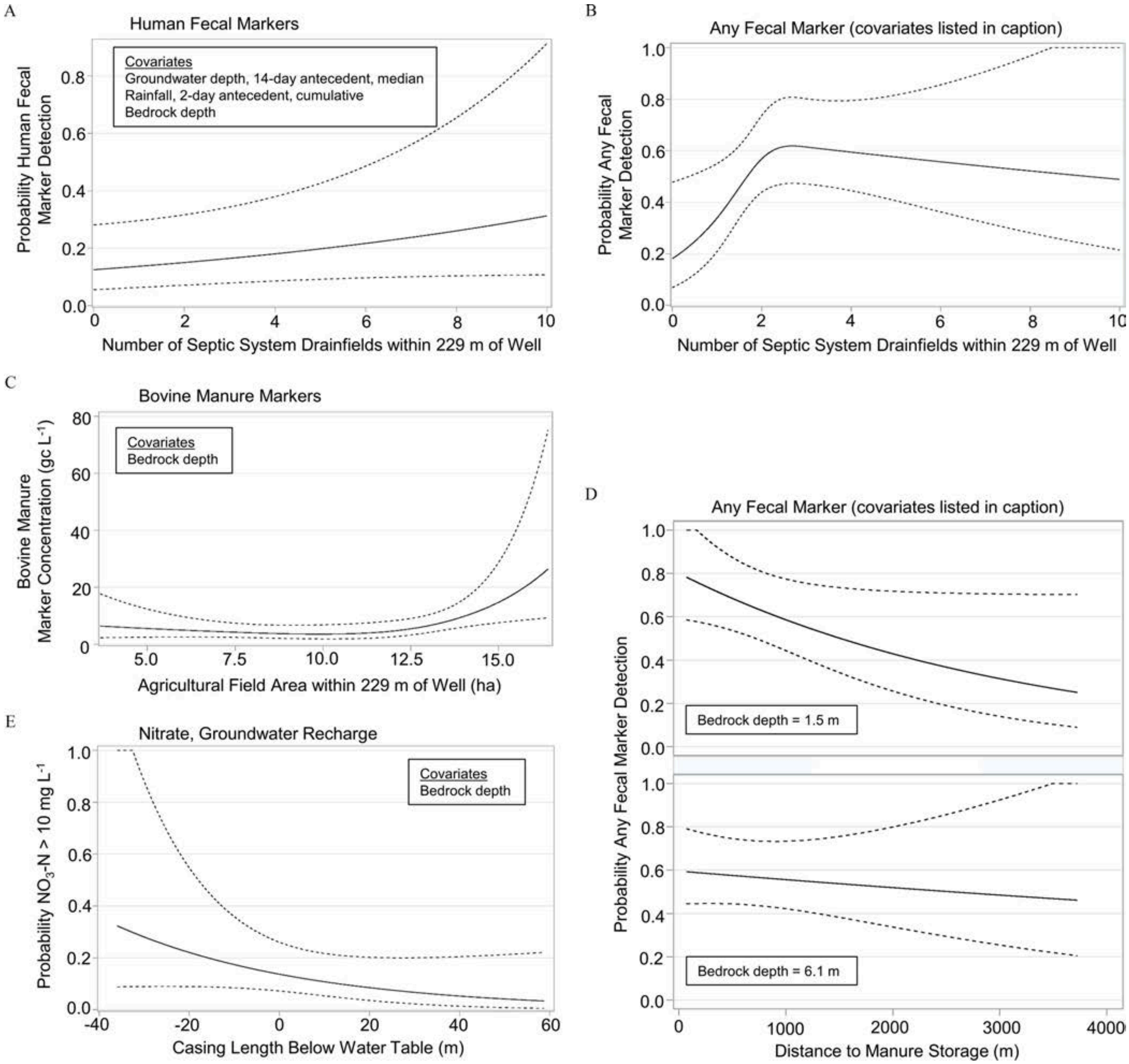

Figure 5.

Key risk factors regressed on private well contamination probability (log-binomial regression) or concentration (gamma regression): (A) detection probability for human-specific markers; (B) detection probability for any fecal marker; covariates: manure storage distance, bedrock depth, manure storage distance times bedrock depth interaction, rainfall 2-d antecedent cumulative; (C) estimated bovine-specific marker concentration (mean sum); (D) interaction between manure storage distance and bedrock depth for any fecal marker detection probability; covariates: septic system drainfields within 229m of well, rainfall 2-d antecedent cumulative; (E) detection probability of NO_3^--N>10mg/L. Black line: regression estimates. Dashed lines: 95% pointwise confidence limits. Coefficients and *p*-values are reported in Table 5. Covariates in the multivariable models were fixed at their median values for the purpose of plotting.

Considering other vulnerable aquifers, Blaschke et al. (2016) estimated the distance between septic systems and private wells needed for 12-log10 virus removal to achieve a risk of 10^{-4} infections/person/y, and their lower setback distance estimates for gravel and coarse gravel aquifers were 66m and 1,000m, respectively (equivalent to densities of 0.7 and 0.003 systems/ha). For limestone aquifers similar to our study site, Morrissey et al. (2015) derived a recommendation of 3.5 systems/ha from groundwater flow modeling of indicator bacteria and nitrate, and Masciopinto et al. (2008) estimated the setback required for 7-log10 virus reduction from municipal wastewater injected into sinkholes was 8,000m. Although previous work was based on indicators and nitrate or log removal of viruses, our model is based on the probability of contamination by fecal waste specific to humans.

Risk Factors for Well Contamination with Bovine Manure Markers

The detection probability of bovine-specific markers increased during periods of groundwater recharge (Table 5), as infiltrating precipitation and snowmelt carried manure from the surface to the water table. An increase from 0 to 40 millimeters cumulative recharge 7 d prior to sampling increased the detection probability of bovine markers from 13% to 50%.

Agricultural risk factors were not associated with the detection probability of bovine markers but were associated with those markers' *concentrations* (see Table S10), and of these the area of agricultural fields within 229m of wells had the strongest association. When the area exceeded 13 ha, bovine marker concentration increased (Figure 5C).

The reason we found associations between fecal sources and detection probability of human markers but not bovine markers likely stem from differences in release patterns between septic systems and manure. Septic system locations are fixed and known with certainty; the systems operate every day, continually releasing household wastewater to the subsurface. In contrast, manure applications vary in location, timing, and volume; manure could be applied near a well on one day and then not again that year. Unlike manure field applications, manure storages are like septic systems: The locations are fixed and known, meaning our distance measurements between manure storages and study wells had minimal error. This may have contributed to our finding that the "distance to manure storage" risk factor was relevant in five multivariable models.

Because manure application records were incomplete (only large farms are required to report applications), we assumed all agricultural fields near wells were potential sources of manure at the time of sampling, which was likely true for only some fields, resulting in misclassification. However, when the model was restricted to only bovine-positive samples, this restriction removed any chance of misclassification (i.e., positivity indubitably showed manure must be near the well), which likely explains why we were able to link agricultural field area to bovine marker concentration. The impact of misclassification of manured sites may have been lessened for contaminant detection models constructed with more positive samples (i.e., greater statistical power). These models (coliforms, nitrate-N, and any fecal marker) did indeed identify agricultural risk factors.

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Risk Factors for Well Contamination with Markers for Any Fecal Microbe

The any fecal marker category included the 82 samples (79 wells) positive for any of the 24 microbial markers found in the fecal material of humans, bovines and ruminants, or other vertebrate hosts (Table 3). Multivariable modeling showed detection of any marker in this category was associated with well proximity to locations of both human and bovine fecal material, namely septic drainfields and manure storages. The model included two other risk factors: rainfall and bedrock depth (Table 5). Similar findings were reported by O'Dwyer et al. (2018), who showed septic system density, cattle density, rainfall, and karst bedrock in Ireland were associated with private well contamination with *E. coli*.

Any fecal marker detection probability increased by a factor of three when septic drainfields increased from zero to two within 229m of wells; additional drainfields did not further increase the detection probability (Figure 5B). Manure storage distance from wells was associated with fecal contamination after accounting for its interaction with bedrock depth; for wells closer to manure storage, the probability of detecting any fecal marker increased more steeply at shallow bedrock depth (Figure 5D).

To model the concentration outcome of any fecal marker, only positive samples were included, reducing statistical power compared to the detection outcome model. Nonetheless, bedrock depth was strongly associated with fecal marker concentration after adjusting for manure storage distance and the area of agricultural fields within 229m of wells (Table 5).

The multivariable models for any fecal marker encapsulate the key study finding: Fecal contamination in the county's private wells stems from both septic systems and manure, and contamination is exacerbated by shallow bedrock depth and elevated rainfall. Both fecal sources release untreated wastes to the landscape at noteworthy volumes. Septic system drainfields in the county are estimated to release into the subsurface 6.79×108 L of household wastewater per year, and the county's cattle population produces approximately 1.74×109 L manure (fecal and urine combined) per year (see Supplemental Material, "Septic System Effluent Volume Released Annually in Kewaunee County, Cattle Manure Volume Produced Annually in Kewaunee County").

Precipitation as a Risk Factor for Private Well Contamination

There is ample evidence showing precipitation favors microbial contamination of private wells. Precipitation quantity in the period preceding sampling was positively associated with the occurrence in private wells of indicator bacteria (Hynds et al. 2012; O'Dwyer et al. 2014; Procopio et al. 2017; Invik 2019) human enteric viruses (Allen et al. 2017) and the human-specific *Bacteroides* marker HF183 (Murphy et al. 2020). The antecedent precipitation periods associated with contamination varied between 30 (Invik et al. 2019) and 5 d (Hynds et al. 2012), and even shorter periods of rainfall (24 h) may be associated with contamination of vulnerable aquifers (Morrissey et al. 2015). In our study 2-d antecedent cumulative rainfall was more strongly associated than 7- or 14-d periods with detection of any fecal marker and markers specific to humans (see Table S10). However, when rainfall was included in multivariable models it was not as strongly associated to contamination as the other risk factors (Table 5).

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Well Construction Risk Factors Related to Contamination

Well construction risk factor modeling did not identify a single overriding factor. Of 14 possible multivariable models (combinations of contaminant type, recharge, and outcome measurement) only six had any variables that met the univariable screening criteria (Table 6; and see Tables S11, S12, and S13). Four of the six models involved nitrate, suggesting well construction was more related to nitrate than microbial contamination. Statistical power may have been an issue, particularly for human and bovine markers, as construction data on file with the state government were not available for 35% of study wells. Nevertheless, the quality of the well construction data was good. Our data were derived from bona fide construction records instead of relying on well-owner recall. Summary statistics for all well construction data are reported in Tables S14–S16.

Table 6

Multivariable modeling of well construction risk factors as related to detection probabilities and concentrations of coliforms, any fecal-associated marker, and nitrate in private wells.

			Multivariable model			
Contaminant and outcome measurement (n)[a]	**Risk factor**	**Univariable model p-value**	**Risk factor median[b]**	**Risk factor range[b]**	**Coefficient or trend[c,d]**	**p-value[e]**
Any fecal marker[f] detection (83)	Casing depth	0.15	17.7	12.2–48.2	Negative	0.31
	Open interval length	0.13	29.0	2.1–79.6	Positive	0.24
	Bedrock depth	0.027	4.6	0–46.6	−0.02	0.26
Coliforms concentration, recharge (47)	Well depth	0.047	48.8	18.3–100.6	Negative	0.59
	Casing depth	0.057	18.9	12.2–80.2	None	0.91
	Groundwater depth at construction	0.0004	12.2	1.8–36.6	Negative	0.0038
	Well age	0.0042	24	5–49	0.04	0.016
NO_3^--N>10mg/L detection, recharge (201)	Casing length below water table	0.040	8.5	–36–58.8	−0.02	0.13
	Bedrock depth	0.0028	6.4	0–55.2	Negative	0.28

Contaminant and outcome measurement (n)[a]	Risk factor	Univariable model p-value	Multivariable model			
			Risk factor median[b]	Risk factor range[b]	Coefficient or trend[c,d]	p-value[e]
NO_3^--N concentration, recharge (124)	Well age	0.15	22	2–80	Positive	0.16
	Bedrock depth	0.0063	4.7	0–31.4	Negative	0.11
NO_3^--N>10mg/L detection, no-recharge (251)	Casing depth	0.12	18.9	6.1–126.5	0.01	0.65
	Casing length below water table	0.02	8.8	–36–117.3	−0.02	0.33
	Bedrock depth	<0.0001	10.1	0.3–54.3	Negative	0.07
NO_3^--N concentration, no-recharge (127)	Casing depth	0.043	18.0	6.1–126.5	−0.008	0.57
	Casing length below water table	0.054	5.5	–19.8–117.3	Negative	0.74
	Bedrock depth	0.0019	6.7	0.3–49.4	Negative	0.0088

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Note: Univariable model p-values used for selecting risk factors are included for reference; complete univariable statistics are provided in Tables S11, S12, and S13. Risk factor eligibility for inclusion in multivariable models is described in statistical methods.

[a]n=number of samples in multivariable model.

[b]Units for length and depth are meters; age is in years.

[c]In lieu of reporting multiple coefficients for spline-represented variables we report the overall trend (positive or negative).

[d]Interpretation of coefficient linear terms: change in ln(detection probability) or change in ln(concentration) for a unit change in the risk factor.

[e]The composite p-value is reported for spline-represented variables.

[f]"Any fecal marker" includes all microorganisms regardless of host specificity.

Casing depth was included in more multivariable well construction models than any other variable; minimum depths specified in well construction codes are believed to prevent contamination. However, its independent effect in the presence of other risk factors in the well construction models was equivocal; associations were weak, and trends were inconsistent (positive, negative, and none)

(Table 6). Casing length below water table was the second most frequently included risk factor, and its trends were consistent; longer casing into the aquifer reduced NO_3^--N contamination. For example, increasing casing length from 36m above to 59m below the water table decreased the probability of NO_3^--N contamination >10mg/L during recharge by 90% (Figure 5E). Placing the casing bottom deeper into the aquifer likely results in nitrate that is infiltrating from the land surface to be further diluted before it enters the well. Of 453 study wells that had data on casing length below water table, 77 wells (17%) had casings that ended above the water table, providing no dilution benefit.

Older wells tend to have greater nitrate and bacterial contamination (Yessis et al. 1996; Goss et al. 1998), but in our study, of the 14 possible multivariable models, well age was associated only with coliforms concentration during recharge (Table 6). Changes in State code in 1988 improved well construction reporting, so our construction data skewed toward newer wells (median age approximately 20 y) that comply with recent construction regulations (e.g., only one well had casing depth less than the State minimum of 12.2m.)

Well depth is frequently identified in groundwater studies as an important factor affecting nitrate and microbial contamination. Deeper wells have less nitrate (Glanville et al. 1997; Lichtenberg and Shapiro 1997; Goss et al. 1998; Allevi et al. 2013; Swistock et al. 2013; Lockhart et al. 2013; Warner and Arnold 2010), coliforms (Gonzales 2008; Goss et al. 1998; Allevi et al. 2013), *E. coli* (O'Dwyer et al. 2018), and human viruses (Allen et al. 2017). Warner and Arnold (2010) found that nitrate concentrations among 378 private wells in the glacial aquifer system in the United States (of which Kewaunee County is part) had less spatial and temporal variation than the variation contributed by well depth. They suggest deeper wells have older groundwater with lower dissolved oxygen favoring denitrification. Well depth was not associated with nitrate contamination in our study wells, likely because the aquifer is oxic at least to 70m (Bradbury and Muldoon 1992).

Hynds et al. (2012) showed that well design and construction were more important than septic systems, geological setting, or precipitation in explaining the variability of thermotolerant coliform contamination in private wells in Ireland. Our findings differ. Overall, well construction was not strongly associated with nitrate and microbial contamination of private wells in the Silurian dolomite aquifer of northeastern Wisconsin. Nor are our findings unique. In a study of 180 randomly selected private wells in northeastern Ohio, well age and well depth determined from construction records were not associated with coliform contamination (Won et al. 2013). Many studies that have investigated the link between well construction and contamination included dug wells and sand points (Yessis et al. 1996; Goss et al. 1998) or wells that lacked adequate sealing between the casing and well annulus, a condition that would allow direct ingress of surface contaminants (Hynds et al. 2012; Fennell 2017). In contrast, for our study wells that have construction data, all were drilled, none were dug, and all were sealed with grout. For wells such as these, in this hydrogeological setting, it appears differences in construction have less impact on contamination than other factors.

Utility and Generalizability of Findings

We have shown private household wells open to the Silurian dolomite aquifer in Kewaunee County, Wisconsin, were contaminated with nitrate, coliform bacteria, and diverse taxa of fecal-associated microbes, some of which were pathogenic. Contamination rates depended on bedrock depth, land use, groundwater recharge, rainfall, and to a lesser extent factors related to well construction. Our examination of risk factors was comprehensive, and multivariable modeling allowed each risk factor to be evaluated for its independent effects in the presence of other factors. In addition, risk factors were analyzed as continuous-scaled variables, which aids interpretation and is amenable for policymaking, for example, establishing setback distances between private wells and agricultural fields, allowable septic system densities, or minimum bedrock depths for manure applications.

Our findings likely apply to other regions that depend on the Silurian dolomite aquifer and where agricultural and exurban land uses affect groundwater quality. The aquifer is regionally extensive and an important water supply for public, domestic, and commercial uses in six U.S. states: Wisconsin, Illinois, Iowa, Michigan, Indiana, and Ohio (USGS 2016). The Silurian dolomite aquifer in Canada extends from Lake Huron to Niagara Falls and supplies water to nearly 800,000 people in southern Ontario (Allen et al. 2017). In northeast Wisconsin the aquifer is emblematic of an open-access resource and the “tragedies” that can result when the resource becomes degraded by competing interests. Understanding how the aquifer is contaminated—the sources, extent, and factors involved—may contribute to the broader appreciation that this essential resource is shared among all who depend on it.

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Acknowledgments

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References

- Alhajjar BJ, Stramer SL, Cliver DO, Harkin JM. 1988. Transport modelling of biological tracers from septic systems. Water Res 22(7):907–915, 10.1016/0043-1354(88)90028-0. [CrossRef] [Google Scholar]
- Allen AS, Borchardt MA, Kieke BA Jr., Dunfield KE, Parker BL. 2017. Virus occurrence in private and public wells in a fractured dolostone aquifer in Canada. Hydrogeol J 25(4):1117–1136, 10.1007/s10040-017-1557-5. [CrossRef] [Google Scholar]
- Allevi RP, Krometis L-AH, Hagedorn C, Benham B, Lawrence AH, Ling EJ, et al.. 2013. Quantitative analysis of microbial contamination in private drinking water supply systems. J Water Health 11(2):244–255, PMID: 23708572, 10.2166/wh.2013.152. [PubMed] [CrossRef] [Google Scholar]
- American Public Health Association. 1995. Standard Methods for the Examination of Water and Wastewater, 19th ed. Washington, DC: American Public Health Association. [Google Scholar]
- Atherholt TB, Korn LR, Louis JB, Procopio NA. 2015. Repeat sampling and coliform bacteria detection rates in New Jersey domestic wells. Groundwater Monit R 35(2):70–80. 10.1111/j.1745-6584.2012.00997.x, 10.1111/gwmr.12094. [CrossRef] [CrossRef] [Google Scholar]
- Bay Lake Regional Planning Commission. 2016. Kewaunee County 20-Year Comprehensive Plan Update. http://media.baylakerpc.org/media/51229/kewaunee%20county%20comp%20plan_farm%20preservation%20plan%20full%20document-%2001182017.pdf [accessed 10 September 2019].
- Blaschke AP, Derx J, Zessner M, Kirnbauer R, Kavka G, Strelec H, et al.. 2016. Setback distances between small biological wastewater treatment systems and drinking water wells against virus contamination in alluvial aquifers. Sci Total Environ 573:278–289, PMID: 27570196, 10.1016/j.scitotenv.2016.08.075. [PubMed] [CrossRef] [Google Scholar]
- Blaustein RA, Pachepsky YA, Hill RL, Shelton DR. 2015. Solid manure as a source of fecal indicator microorganisms: release under simulated rainfall. Environ Sci Technol 49(13):7860–7869, PMID: 26011817, 10.1021/acs.est.5b01095. [PubMed] [CrossRef] [Google Scholar]
- Borchardt MA, Bradbury KR, Alexander EC, Archer J, Braatz L, Forest B, et al.. 2011. Norovirus outbreak caused by a new septic system in a dolomite aquifer. Ground Water 49(1):85–97, PMID: 20199588, 10.1111/j.1745-6584.2010.00686.x. [PubMed] [CrossRef] [Google Scholar]
- Borchardt MA, Muldoon MA, Hunt RJ, Bonness DE, Firnstahl AD, Kieke BA, et al.. 2019. Assessing Groundwater Quality in Kewaunee County, Wisconsin and Characterizing the Timing and Variability of Enteric Pathogen Contamination Within the Dolomite Aquifer in Northeastern Wisconsin. DNR Project Number 227, Final Report. Wisconsin Geology and Natural History Survey, Open-File Report 2019-05, 2019.
- Bradbury KR, Muldoon MA. 1992. Hydrogeology and groundwater monitoring of fractured dolomite in the Upper Door Priority Watershed, Door County, Wisconsin. Wisconsin Geological and Natural History Survey Open File Report, WOFR 1992-2, https://wgnhs.wisc.edu/catalog/publication/000776/resource/wofr199202 [accessed 22 June 2019].

September 3, 2024
Clean Water Organizations Comments Exhibit 28

- Bradbury KR, Rayne TW, Muldoon MA. 2001. Field verification of capture zones for municipal wells at Sturgeon Bay, Wisconsin. Final Report to the Wisconsin Department of Natural Resources. https://wgnhs.uwex.edu/pubs/000823/ [accessed 22 June 2019].
- Bromley DW, Cernea MM. 1989. The Management of Common Property Natural Resources: Some Conceptual and Operational Fallacies (English). World Bank Discussion Papers; No. WDP 57. Washington, DC: World Bank Group, http://documents.worldbank.org/curated/en/548811468740174575/The-management-of-common-property-natural-resources-some-conceptual-and-operational-fallacies [accessed 15 March 2018]. [Google Scholar]
- Center for Watershed Science and Education Wisconsin. 2018. Well Water Viewer: Private Well Data for Wisconsin. University of Wisconsin – Stevens Point. https://www.uwsp.edu/cnr-ap/watershed/Pages/WellWaterViewer.aspx [accessed 29 March 2018].
- DiSimone LA. 2009. Quality of Water from Domestic Wells in Principal Aquifers of the United States, 1991-2004. U.S. Geological Survey Scientific Investigations Report 2008–5227. http://pubs.usgs.gov/sir/2008/5227 [accessed 21 November 2019].
- Diston D, Sinreich M, Zimmermann S, Baumgartner A, Felleisen R. 2015. Evaluation of molecular- and culture-dependent MST markers to detect fecal contamination and indicate viral presence in good quality groundwater. Environ Sci Technol 49(12):7142–7151, PMID: 25871525, 10.1021/acs.est.5b00515. [PubMed] [CrossRef] [Google Scholar]
- Eckhardt DAV, Stackelberg PE. 1995. Relation of ground-water quality to land use on Long Island, New York. Groundwater 33(6):1019–1033, 10.1111/j.1745-6584.1995.tb00047.x. [CrossRef] [Google Scholar]
- Erb K, Ronk E, Koundinya V, Luczaj J. 2015. Groundwater quality changes in a karst aquifer of northeastern Wisconsin, USA: reductions of brown water incidence and bacterial contamination resulting from implementation of regional task force recommendations. Resources 4(3):655–672, 10.3390/resources4030655. [CrossRef] [Google Scholar]
- Final Report of the Northeast Wisconsin Karst Task Force. 2007. Erb K, Steiglitz R, eds. https://cdn.shopify.com/s/files/1/0145/8808/4272/files/G3836.pdf [accessed 12 February 2020].
- Felleiter S, McDermott K, Hall G, Sheth P, Majury A. 2020. Exploring private water wells for fecal sources and evidence of pathogen presence in the context of current testing practices for potability in Ontario. Water Qual Res J Can 55(1):93–105, 10.2166/wqrj.2019.035. [CrossRef] [Google Scholar]
- Fennell C. 2017. The impact of domestic wastewater treatment system effluent on private water wells: an evaluation of contamination fingerprinting techniques. Thesis. University of Dublin, Trinity College, Dublin. [Google Scholar]
- Gardner KK, Vogel RM. 2005. Predicting ground water nitrate concentration from land use. Ground Water 43(3):343–352, PMID: 15882326, 10.1111/j.1745-6584.2005.0031.x. [PubMed] [CrossRef] [Google Scholar]
- Gardner TED, Geary P, Gordon IAN. 1997. Ecological sustainability and on-site effluent treatment systems. Australas J Environ Manag 4(2):144–156, 10.1080/14486563.1997.10648378. [CrossRef] [Google Scholar]
- Garson GD. 2013. Generalized Linear Models/Generalized Estimating Equations, 2013 Edition. Asheboro, NC: Statistical Associates Publishers. [Google Scholar]

September 3, 2024
Clean Water Organizations Comments Exhibit 28

- Gibson KE, Schwab KJ, Spencer SK, Borchardt MA. 2012. Measuring and mitigating inhibition during quantitative real time PCR analysis of viral nucleic acid extracts from large-volume environmental samples. Water Res 46(13):4281–4291, PMID: 22673345, 10.1016/j.watres.2012.04.030. [PubMed] [CrossRef] [Google Scholar]
- Glanville TD, Baker JL, Newman JK. 1997. Statistical analysis of rural well contamination and effects of well construction. Trans ASAE 40(2):363–370, 10.13031/2013.21281. [CrossRef] [Google Scholar]
- Goss MJ, Barry DAJ, Rudolph DL. 1998. Contamination in Ontario farmstead domestic wells and its association with agriculture: 1. Results from drinking water wells. J Contam Hydrol 32(3–4):267–293, 10.1016/S0169-7722(98)00054-0. [CrossRef] [Google Scholar]
- Hagedorn C, Mc Coy EL, Rahe TM. 1981. The potential for ground water contamination from septic effluents. J Environ Qual 10(1):1–8, 10.2134/jeq1981.00472425001000010001x. [CrossRef] [Google Scholar]
- Hastie TJ, Tibshirani RJ, Friedman JH. 2001. The Elements of Statistical Learning: Data Mining, Inference, and Prediction. New York: Springer-Verlag. [Google Scholar]
- Healy RW, Cook PG. 2002. Using groundwater levels to estimate recharge. Hydrogeol J 10(1):91–109, 10.1007/s10040-001-0178-0. [CrossRef] [Google Scholar]
- Hynds PD, Misstear BD, Gill LW. 2012. Development of a microbial contamination susceptibility model for private domestic groundwater sources. Water Resour Res 48:W12504, 10.1029/2012WR012492. [CrossRef] [Google Scholar]
- Hynds P, Misstear BD, Gill LW, Murphy HM. 2014b. Groundwater source contamination mechanisms: physicochemical profile clustering, risk factor analysis and multivariate modeling. J Contam Hydrol 159:47–56, PMID: 24583518, 10.1016/j.jconhyd.2014.02.001. [PubMed] [CrossRef] [Google Scholar]
- Invik J, Barkema HW, Massolo A, Neumann NF, Cey E, Checkley S. 2019. Escherichia coli contamination of rural well water in Alberta, Canada is associated with soil properties, density of livestock and precipitation. Can Water Resour J 44(3):248–262, 10.1080/07011784.2019.1595157. [CrossRef] [Google Scholar]
- Iturriza-Gómara M, Kang G, Gray J. 2004. Rotavirus genotyping: keeping up with an evolving population of human rotaviruses. J Clin Virol 31(4):259–265, PMID: 15494266, 10.1016/j.jcv.2004.04.009. [PubMed] [CrossRef] [Google Scholar]
- Jiang B, Dennehy PH, Spangenberger S, Gentsch JR, Glass RI. 1995. First detection of group C rotavirus in fecal specimens of children with diarrhea in the United States. J Infect Dis 172(1):45–50, PMID: 7797945, 10.1093/infdis/172.1.45. [PubMed] [CrossRef] [Google Scholar]
- Katz BG, Griffin DW, McMahon PB, Harden HS, Wade E, Hicks RW, et al.. 2010. Fate of effluent-borne contaminants beneath septic tank drainfields overlying a karst aquifer. J Environ Qual 39(4):1181–1195, PMID: 20830905, 10.2134/jeq2009.0244. [PubMed] [CrossRef] [Google Scholar]
- Krolik J, Evans G, Belanger P, Maier A, Hall G, Joyce A, et al.. 2014. Microbial source tracking and spatial analysis of *E. coli* contaminated private well waters in southeastern Ontario. J Water Health 12(2):348–356, PMID: 24937229, 10.2166/wh.2013.192. [PubMed] [CrossRef] [Google Scholar]

September 3, 2024
Clean Water Organizations Comments Exhibit 28

- Li X, Atwill ER, Antaki E, Applegate O, Bergamaschi B, Bond RF, et al.. 2015. Fecal indicator and pathogenic bacteria and their antibiotic resistance in alluvial groundwater of an irrigated agricultural region with dairies. J Environ Qual 44(5):1435–1447, PMID: 26436261, 10.2134/jeq2015.03.0139. [PubMed] [CrossRef] [Google Scholar]
- Lichtenberg E, Shapiro LK. 1997. Agriculture and nitrate concentrations in Maryland community water system wells. J Environ Qual 26(1):145–153, 10.2134/jeq1997.00472425002600010022x. [CrossRef] [Google Scholar]
- Lockhart KM, King AM, Harter T. 2013. Identifying sources of groundwater nitrate contamination in a large alluvial groundwater basin with highly diversified intensive agricultural production. J Contam Hydrol 151:140–154, PMID: 23800783, 10.1016/j.jconhyd.2013.05.008. [PubMed] [CrossRef] [Google Scholar]
- Lohr SL. 2010. Sampling: Design and Analysis. 2nd ed. Boston, MA: Brooks/Cole. [Google Scholar]
- Lusk MG, Toor GS, Yang Y-Y, Mechtensimer S, De M, Obreza TA. 2017. A review of the fate and transport of nitrogen, phosphorus, pathogens, and trace organic chemicals in septic systems. Crit Rev Environ Sci Technol 47(7):455–541, 10.1080/10643389.2017.1327787. [CrossRef] [Google Scholar]
- Madadgar O, Nazaktabar A, Keivanfar H, Zahraei Salehi T, Lotfollah Zadeh S. 2015. Genotyping and determining the distribution of prevalent G and P types of group a bovine rotaviruses between 2010 and 2012 in Iran. Vet Microbiol 179(3–4):190–196, PMID: 26072368, 10.1016/j.vetmic.2015.04.024. [PubMed] [CrossRef] [Google Scholar]
- Masciopinto C, La Mantia R, Chrysikopoulos CV. 2008. Fate and transport of pathogens in a fractured aquifer in the Salento area, Italy. Water Resour Res 44:W01404, 10.1029/2006WR005643. [CrossRef] [Google Scholar]
- Matthijnssens J, Joelsson DB, Warakomski DJ, Zhou T, Mathis PK, van Maanen MH, et al.. 2010. Molecular and biological characterization of the 5 human-bovine rotavirus (WC3)-based reassortant strains of the pentavalent rotavirus vaccine, RotaTeq. Virology 403(2):111–127, PMID: 20451234, 10.1016/j.virol.2010.04.004. [PubMed] [CrossRef] [Google Scholar]
- McNutt LA, Wu C, Xue X, Hafner JP. 2003. Estimating the relative risk in cohort studies and clinical trials of common outcomes. Am J Epidemiol 157(10):940–943, PMID: 12746247, 10.1093/aje/kwg074. [PubMed] [CrossRef] [Google Scholar]
- Mieszkin S, Yala JF, Joubrel R, Gourmelon M. 2010. Phylogenetic analysis of *Bacteroidales* 16S rRNA gene sequences from human and animal effluents and assessment of ruminant faecal pollution by real-time PCR. J Appl Microbiol 108(3):974–984, PMID: 19735325, 10.1111/j.1365-2672.2009.04499.x. [PubMed] [CrossRef] [Google Scholar]
- Morrissey PJ, Johnston PM, Gill LW. 2015. The impact of on-site wastewater from high density cluster developments on groundwater quality. J Contam Hydrol 182:36–50, PMID: 26331764, 10.1016/j.jconhyd.2015.07.008. [PubMed] [CrossRef] [Google Scholar]
- Muldoon MA, Bradbury KR. 2010. Assessing seasonal variations in recharge and water quality in the silurian aquifer in areas with thicker soil cover. Wisconsin groundwater management practice monitoring project, DNR-198. http://digital.library.wisc.edu/1711.dl/EcoNatRes.MuldoonAssess [accessed 22 June 2019].

September 3, 2024
Clean Water Organizations Comments Exhibit 28

- Muldoon MA, Simo JA, Bradbury KR. 2001. Correlation of high-permeability zones with stratigraphy in a fractured-dolomite aquifer, Door County, Wisconsin. Hydrogeol J 9(6):570–583, 10.1007/s10040-001-0165-5. [CrossRef] [Google Scholar]
- Murphy HM, McGinnis S, Blunt R, Stokdyk J, Wu J, Cagle A, et al.. 2020. Septic systems and rainfall influence human fecal marker and indicator organism occurrence in private wells in southeastern Pennsylvania. Environ Sci Technol 54(6):3159–3168, PMID: 32073835, 10.1021/acs.est.9b05405. [PubMed] [CrossRef] [Google Scholar]
- Nennich TD, Harrison JH, VanWieringen LM, Meyer D, Heinrichs AJ, Weiss WP, et al.. 2005. Prediction of manure and nutrient excretion from dairy cattle. J Dairy Sci 88(10):3721–3733, PMID: 16162547, 10.3168/jds.S0022-0302(05)73058-7. [PubMed] [CrossRef] [Google Scholar]
- NGWA (National Groundwater Association). 2020. Groundwater Use in the United States of America. NGWA USA Groundwater Use Factsheet 01-13-2020. https://www.ngwa.org/docs/default-source/default-document-library/groundwater/usa-groundwater-use-fact-sheet.pdf?sfvrsn=5c7a0db8_4 [accessed 1July 2020].
- Nicosia LA, Rose JB, Stark L, Stewart MT. 2001. A field study of virus removal in septic tank drainfields. J Environ Qual 30(6):1933–1939, PMID: 11789999, 10.2134/jeq2001.1933. [PubMed] [CrossRef] [Google Scholar]
- NOAA (National Oceanic and Atmospheric Administration/National Centers for Environmental Information). n.d.. Climate at a Glance: Global Mapping. https://www.ncdc.noaa.gov/cag/ [accessed 18 March 2021].
- Nolan BT, Hitt KJ. 2006. Vulnerability of shallow groundwater and drinking-water wells to nitrate in the United States. Environ Sci Technol 40(24):7834–7840, PMID: 17256535, 10.1021/es060911u. [PubMed] [CrossRef] [Google Scholar]
- Nolan BT. 2001. Relating nitrogen sources and aquifer susceptibility to nitrate in shallow ground waters of the United States. Groundwater 39(2):290–299, 10.1111/j.1745-6584.2001.tb02311.x. [PubMed] [CrossRef] [Google Scholar]
- NRCS 313 (Natural Resources Conservation Service Conservation Practice Standard 313). Waste Storage Facility, Wisconsin Natural Resources Conservation Service, Madison, WI, January 2014.
- O’Dwyer J, Dowling A, Adley CC. 2014. Microbiological assessment of private groundwater-derived potable water supplies in the Mid-West Region of Ireland. J Water Health 12(2):310–317, PMID: 24937225, 10.2166/wh.2014.178. [PubMed] [CrossRef] [Google Scholar]
- O’Dwyer J, Hynds PD, Byrne KA, Ryan MP, Adley CC. 2018. Development of a hierarchical model for predicting microbiological contamination of private groundwater supplies in a geologically heterogeneous region. Environ Pollut 237:329–338, PMID: 29499576, 10.1016/j.envpol.2018.02.052. [PubMed] [CrossRef] [Google Scholar]
- Procopio NA, Atherholt TB, Goodrow SM, Lester LA. 2017. The likelihood of coliform bacteria in NJ domestic wells based on precipitation and other factors. Ground Water 55(5):722–735, PMID: 28369797, 10.1111/gwat.12518. [PubMed] [CrossRef] [Google Scholar]

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- Rasmuson A, Erickson B, Borchardt M, Muldoon M, Johnson WP. 2020. Pathogen prevalence in fractured versus granular aquifers and the role of forward flow stagnation zones on pore-scale delivery to surfaces. Environ Sci Technol 54(1):137–145, PMID: 31770489, 10.1021/acs.est.9b03274. [PubMed] [CrossRef] [Google Scholar]
- Rayne TW, Bradbury KR, Krause JJ. 2019. Impacts of a rural subdivision on groundwater quality: results of long-term monitoring. Ground Water 57(2):279–291, PMID: 29603208, 10.1111/gwat.12666. [PubMed] [CrossRef] [Google Scholar]
- Reneau RB Jr. 1979. Changes in concentrations of selected chemical pollutants in wet, tile-drained soil systems as influenced by disposal of septic tank effluents. J Environ Qual 8(2):189–196, 10.2134/jeq1979.00472425000800020011x. [CrossRef] [Google Scholar]
- Shaver RH, Ault CH, Ausich WI, Droste JB, Horowitz AS, James WC, et al.. 1978. The Search for a Silurian Reef Model, Great Lakes Area: Indiana Department of Natural Resources Geological Survey Special Report 15. https://igws.indiana.edu/bookstore/details.cfm?Pub_Num=SR15 [accessed 29 June 2019].
- Sherrill MG. 1979. Contamination Potential in the Silurian Dolomite Aquifer, Eastern Wisconsin. USGS Water-Resources Investigations Open-File Report, 78–108. Reston, VA: USGS, 10.3133/wri78108 [accessed 1 October 2018]. [CrossRef] [Google Scholar]
- Sherrill MG. 1978. Geology and ground Water in Door County, Wisconsin, with Emphasis on Contamination Potential in the Silurian Dolomite. U.S. Geological Survey Water-Supply Paper 2047. Reston, VA: USGS. [Google Scholar]
- Stokdyk J, Spencer SK, Walsh J, de Lambert J, Firnstahl AD, Anderson A, et al.. 2019. *Cryptosporidium* incidence and surface water influence of groundwater supplying public water systems in Minnesota, USA. Environ Sci Technol 53(7):3391–3398, PMID: 30895775, 10.1021/acs.est.8b05446. [PubMed] [CrossRef] [Google Scholar]
- Stokdyk JP, Firnstahl AD, Walsh JF, Spencer SK, de Lambert JR, Anderson AC, et al.. 2020. Viral, bacterial, and protozoan pathogens and fecal markers in wells supplying groundwater to public water systems in Minnesota, USA. Water Res 178:115814, PMID: 32325219, 10.1016/j.watres.2020.115814. [PubMed] [CrossRef] [Google Scholar]
- Toetz D. 2006. Nitrate in ground and surface waters in the vicinity of a concentrated animal feeding operation. Arch Hydrobiol 166(1):67–77, 10.1127/0003-9136/2006/0166-0067. [CrossRef] [Google Scholar]
- U.S. EPA. (United States Environmental Protection Agency). 1977. The Report to Congress–Waste Disposal Practices and Their Effects on Ground Water. CgD-78-120: B-166506. Washington, DC: U.S. Environmental Protection Agency. [Google Scholar]
- U.S. EPA. 2020. National Primary Drinking Water Regulations. 40 CFR Part 141 https://www.ecfr.gov/cgi-bin/text-idx?tpl=/ecfrbrowse/Title40/40cfr141_main_02.tpl [accessed 16 April 2020].
- U.S. General Accounting Office. 1997. Information on the Quality of Water Found at Community Water Systems and Private Wells. United States GAO/RCED-97-123.. Washington, DC: U.S. Government Accounting Office. [Google Scholar]
- USDA NASS (U.S. Department of Agriculture National Agricultural Statistics Service). 2002. Census of Agriculture. http://agcensus.mannlib.cornell.edu/AgCensus/getVolumeOnePart.do?year=2002&part_id=1015&number=49&title=Wisconsin [accessed 2 November 2018].

September 3, 2024
Clean Water Organizations Comments Exhibit 28

- USDA NASS. 2017. Census of Agriculture. www.nass.usda.gov/AgCensus [accessed 8 December 2020].
- USGS (U.S. Geological Survey). 2016. U.S. Geological Survey Ground Water Atlas of the United States, https://pubs.usgs.gov/ha/ha730/ Page last modified 30-Nov-2016 [accessed 27 May 2020].
- USGS (U.S. Geological Survey). 2020. USGS Water Data for the Nation: U.S. Geological Survey National Water Information System Database. 10.5066/F7P55KJN [accessed 15 September 2018]. [CrossRef]
- Warner KL, Arnold TL. 2010. Relations that Affect the Probability and Prediction of Nitrate Concentration in Private Wells in the Glacial Aquifer System in the United States. U.S. Geological Survey Scientific Investigations Report 2010–5100, https://pubs.usgs.gov/sir/2010/5100/ [accessed 1 April 2020].
- Wisconsin Department of Health Services. n.d. Drinking Water: Manure in Private Wells. https://www.dhs.wisconsin.gov/water/manure.htm [accessed 10 May 2021].
- Wisconsin Department of Natural Resources. 2018. Deer Abundance and Densities in Wisconsin Deer Management Units. https://dnr.wi.gov/topic/hunt/maps.html/falldeerperdr.pdf [accessed 12 May 2021].
- Won G, Gill A, LeJeune JT. 2013. Microbial quality and bacteria pathogens in private wells used for drinking water in northeastern Ohio. J Water Health 11(3):555–562, PMID: 23981882, 10.2166/wh.2013.247. [PubMed] [CrossRef] [Google Scholar]
- Yates MV. 1985. Septic tank density and ground-water contamination. Groundwater 23(5):586–591, 10.1111/j.1745-6584.1985.tb01506.x. [CrossRef] [Google Scholar]
- Yessis J, McColl RS, Seliske P. 1996. Methodological aspects of monitoring for microbial contamination of drinking water from private wells: the water quality program of the region of waterloo. Can Water Resour J 21(3):221–228, 10.4296/cwrj2103221. [CrossRef] [Google Scholar]
- Zhang Y, Kelly W, Panno SV, Liu W-T. 2014. Tracing fecal pollution sources in karst groundwater by Bacteriodales genetic biomarkers, bacterial indicators, and environmental variables. Sci Total Environ 490:1082–1090, PMID: 24922611, 10.1016/j.scitotenv.2014.05.086. [PubMed] [CrossRef] [Google Scholar]

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Bacteria | Minnesota Pollution Control Agency

pca.state.mn.us/pollutants-and-contaminants/bacteria

Countless bacteria can be found in land, water, humans, and animals. Most bacteria are beneficial, serving as food for larger organisms and playing critical roles in natural processes such as organic matter decomposition and food digestion. But about 10% of bacteria, such as *E. coli,* are harmful and, if ingested by humans, can cause sickness or even death.

Sources

Bacteria in Minnesota lakes and streams mainly come from sources such as failing septic systems, wastewater treatment plant releases, livestock, and urban stormwater. Waste from pets and wildlife is another, lesser source of bacteria.

Human health and environmental concerns

In addition to bacteria, human and animal waste may contain pathogens such as viruses and protozoa that could be harmful to humans and other animals. The behavior of bacteria and pathogens in the environment is complex. Levels of bacteria and pathogens in a body of water depend not only on their source, but also on weather, current, and water temperature. As these factors fluctuate, the level of bacteria and pathogens in the water may increase or decrease. Some bacteria can survive and grow in the environment while many pathogens tend to die off with time.

Monitoring, reporting, and regulations

Testing for specific disease-producing bacteria or other pathogens is difficult, expensive, and time-consuming. The MPCA tests for fecal coliform and *E. coli* bacteria, which are commonly found in fecal waste and are easy to measure. They are often used as "indicator organisms" to denote the potential presence of fecal waste. Although using indicator bacteria to assess the presence of pathogens in not a perfect process, it is the best available option at this time. Lakes and streams in Minnesota meet water quality standards if they have a monthly geometric mean less than 126 colony-forming units of *E. coli* per 100 milliliters of water, between April and October.

Most lakes and streams in Minnesota meet water quality standards for bacteria. MPCA uses the *E. coli* water quality standard to identify water bodies that may be contaminated with fecal waste. Higher levels of *E. coli* in the water may or may not be accompanied by higher levels of pathogens and an increased risk of harm; varying survival rates of bacteria make is impossible to definitively state when pathogens are present. See the Minnesota Department of Health Waterborne Illness web page for more information on how to reduce your risk for waterborne illnesses when swimming, boating, or wading.

Is my lake or stream safe for swimming?

Minnesota does not have a list of "safe" bodies of water for recreation. Sometimes a city or county health department will close a swimming beach due to bacterial contamination. Conditions can change over time, and state water-testing efforts are not frequent enough to be time current, particularly in streams and rivers. If you have questions about a specific beach, check with the proper beach authority for their current information and recommendations.

Check with your city or county environmental services to see if your local lake is tested on a regular basis. Two examples of local testing programs:

- Minnesota Lake Superior Beach Monitoring Program
- Hennepin County public swimming beaches

Addressing bacterial contamination

Some bacteria and pathogens will always be present in surface waters. While most of the bacteria and pathogens from fecal waste in the water will die off over time, some may survive. Pathogens from fecal waste generally die off in the environment much faster than bacteria. While there is not a way to rid water bodies of all pathogens, we can reduce bacteria in surface waters by combining the efforts of many individuals and groups. The best methods include:

- Controlling runoff on feedlot properties and where manure is spread on farmland
- Repairing or replacing failing septic systems
- Improving wastewater treatment processes at some facilities
- Controlling erosion with practices such as conservation tillage and riparian buffers
- Rotational livestock grazing, which reduces both sedimentation and fecal coliform concentrations
- Urban stormwater management – runoff detention, infiltration, and street sweeping

Many government entities and groups across Minnesota are working to better understand sources of bacteria in water and mitigate them. Some examples include:

- Pollutant reduction studies that lead to limits on bacteria discharged by wastewater treatment facilities to lakes and streams
- Feedlot runoff controls and other conservation practices installed by farmers because of permit requirements or a statewide water quality certification program
- County and state programs to bring failing sewer systems into compliance

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The Animal Feed Industry's Impact on the Planet

independentmediainstitute.org/2024/01/29/the-animal-feed-industrys-impact-on-the-planet

January 30, 2024

January 29, 2024

> *Click here to read the article on the Observatory.*

The diet of factory-farmed animals is linked to environmental destruction around the globe.

By *Vicky Bond*

In some parts of the continental United States, you might drive through a nearly unchanging landscape for hours. Stretching for miles and miles, vast swaths of soil are dedicated to growing crops—corn, grains, fruits, and vegetables that make up the foundation of our food system.

The process seems highly efficient, producing enormous quantities of food every year. But only a small percentage of these crops will go toward feeding humans. According to a 2013 study conducted by researchers at the Institute on the Environment at the University of

Minnesota and published in the journal Environmental Research Letters, a mere 27 percent of crop calorie production in the United States actually feeds humans. So what happens to the rest?

Some crops are used for the production of ethanol and other biofuels. But the vast majority—more than 67 percent of crop calories grown in the U.S.—are used to feed animals raised for human consumption.

Rather than feeding people, these crops feed the billions of chickens, cows, pigs, and other animals who live and die on factory farms. And that's a problem.

The issue is that feeding humans indirectly—essentially, making animals the caloric middlemen—is a highly inefficient use of food. "For every 100 calories of grain we feed animals, we get only about 40 new calories of milk, 22 calories of eggs, 12 of chicken, 10 of pork, or 3 of beef," writes Jonathan Foley, PhD, executive director of the nonprofit Project Drawdown, for National Geographic. "Finding more efficient ways to grow meat and shifting to less meat-intensive diets… could free up substantial amounts of food across the world."

This shift in growing and consuming food more sustainably has become especially important, with up to 783 million people facing hunger in 2022, according to the United Nations. Research indicates that if we grew crops exclusively for humans to consume directly we could feed an additional 4 billion people worldwide.

Farming has always loomed large in American politics, history, and identity. But the idyllic farming we may imagine—rich piles of compost, seedlings poking through the soil, and flourishing gardens of diverse fruits and vegetables—has transformed into factory farming, a highly industrialized system far removed from earth and soil. Animal feed is essential for the sustenance of this industry—supplying the cattle feedlots, broiler chicken sheds, and egg factories that increasingly make up the foundation of our food system.

What Factory-Farmed Animals Eat

Take a moment to picture a farm animal enjoying dinner. Are you imagining a cow grazing on grass or perhaps a chicken pecking at the ground, foraging for seeds and insects? In today's factory farming system, the "feed" these animals eat is far removed from their natural diets. Rather than munching on grass or insects, most animals on factory farms eat some type of animal feed—a cost-effective mixture of grains, proteins, and often the addition of antibiotics designed to make them grow as quickly as possible.

The ingredients in animal feed don't just matter to the animals' health. They also impact human health—especially since the average American consumes 25 land animals yearly. Researchers have noted that animal feed ingredients are "fundamentally important" to human health impacts. As author and journalist Michael Pollan puts it: “We are what we eat, it is often said, but of course that’s only part of the story. We are what what we eat eats too.”

So, what are the main ingredients used in animal feed today?

Corn and Other Grains

In 2019, farmers planted 91.7 million acres of corn in the U.S. This equals 69 million football fields of corn. How can so much land be devoted to a single crop—especially something many people only eat on occasion?

The answer is that corn is in almost everything Americans eat today. It’s just there indirectly—in the form of animal feed, corn-based sweeteners, or starches. The U.S. is the world’s largest producer, consumer, and exporter of corn. And a large percentage of all that corn is used for animal feed, supplying factory farms across the country.

While “cereal grains”—such as barley, sorghum, and oats—are also used for animal feed, corn is by far the number one feed grain used in the U.S., accounting for more than 96 percent of total feed grain production. Corn supplies the carbohydrates in animal feed, offering a rich energy source to increase animals’ growth.

Unfortunately, what this system offers in efficiency it lacks in resilience. Numerous researchers have expressed concern about the vulnerability of the food supply that is so reliant on a single crop. “Under these conditions, a single disaster, disease, pest, or economic downturn could cause a major disturbance in the corn system,” notes Jonathan Foley in another article for Scientific American. “The monolithic nature of corn production presents a systemic risk to America’s agriculture.”

Soybeans

When you think about soybeans, you might imagine plant-based foods like tofu and tempeh. However, the vast majority of soybeans are used for animal feed. Animal agriculture uses 97 percent of all soybean meal produced in the United States.

While corn is rich in carbohydrates, soybeans are the world’s largest source of animal protein feed. Similar to corn, Americans might not eat a lot of soybeans in the form of tofu, tempeh, and soy milk—in fact, 77 percent of soy grown globally is used to feed livestock, and only 7 percent of it is used directly for human consumption, states a 2021 Our World in Data article—but they do consume soy indirectly through animal products like meat and dairy.

Soy production comes at a high cost to the environment. It is heavily linked to deforestation, driving the destruction of forests, savannahs, and grasslands—as these natural ecosystems are converted to unnatural farmland—and “putting traditional, local livelihoods at risk.”

Critical habitats, like the Cerrado savannah in Brazil, are being razed to clear space for soybean production to meet the global demand for animal feed. More than half of the Cerrado's 100 million hectares of native landscape has already been lost, with livestock and soybean farming being major contributors to this destruction.

"Most soybean-driven land conversions in Brazil have happened in the Cerrado," said Karla Canavan, vice president for commodity trade and finance at World Wildlife Fund, in 2022. "The corridor [Cerrado] is like an inverted forest that has enormous roots and is a very important carbon sink. … Unfortunately, more than 50 percent of the Cerrado has been already converted into soybean farmlands."

It's a common misconception that plant-based soy products like tofu drive global deforestation. In reality, the vast majority of soy is used for animal feed. To fight this tragic habitat destruction, it's far more effective to replace meat with soy-based alternatives.

Animal Protein and Waste

Editor's note: The following section contains graphic descriptions that may disturb some readers.

It's not just plants like corn and soybeans that go into animal feed. The factory farming industry has a long history of feeding animals waste and proteins from other animals. In 2014, outrage ensued when an investigation by the Humane Society of the United States revealed that pig farmers were feeding animals the intestines of their own piglets. At a huge factory farm in Kentucky, workers were filmed eviscerating dead piglets and turning their intestines into a puree that was being fed back to mother pigs.

This wasn't even an isolated atrocity. The executive director of the American Association of Swine Veterinarians in 2014 commented that the practice was "legal and safe" and was meant to immunize the mother pigs against a virus called porcine epidemic diarrhea, according to the New York Times. Pigs aren't the only animals who are effectively turned into cannibals by the factory farming industry.

Farmers were only prohibited from feeding cow meat to other cows following concerns about bovine spongiform encephalopathy (BSE), more commonly known as mad cow disease. The U.S. Department of Agriculture notes on its website that BSE may have been caused by feeding cattle protein from other cows. The practice was banned in 1997—but, notably, only because of the risks to human health and not out of concern for the cows.

Antibiotics

Another key ingredient in animal feed likely doesn't come to mind when you think about animal nutrition. This ingredient is antibiotics, commonly used in the food given to animals across the country.

On factory farms, animals are confined in extremely crowded, filthy facilities—the perfect conditions for spreading illness and disease. Not only do antibiotics allow animals to survive the conditions in these facilities but they also encourage animals to grow unnaturally large and fast. Drugs are administered through food and water, starting when the animals are just a few days old.

The meat industry's excessive antibiotic use has directly been linked to antimicrobial resistance (AMR), a massive threat to human health. As bacteria are killed off, the surviving that remain gradually learn how to survive the attacks, becoming resistant to antibiotics over time.

AMR means that conditions that should be easy and affordable to treat—like ear infections—can become life-threatening. It's "one of today's biggest threats to global health, food security, and development," according to the World Health Organization, states a News-Medical article, and it's projected to kill four times as many people per year as COVID-19 did in 2020, according to the British Society for Antimicrobial Chemotherapy.

Additives and Preservatives

Along with the mixture of corn, soybeans, and a cocktail of antibiotics, animal feed may also contain a plethora of additives and preservatives. The Code of Federal Regulations provides a long list of additives legally permitted in animals' food and drinking water. These include "condensed animal protein hydrolysate" (produced from meat byproducts of cattle slaughtered for human consumption), formaldehyde, and petrolatum—to name a few.

Unfortunately, many of these additives and preservatives have been linked to adverse human health impacts. For example, formaldehyde, which is classified as a known human carcinogen by the National Toxicology Program, is commonly used in animal feed to reduce salmonella contamination. In 2017, following concerns about farmworkers being exposed to the harmful substance, the European Commission voted to ban feed producers from using formaldehyde as an additive in animal feed.

Animal Feeding Operations

To understand the true impact of animal feed, we must look at animal feeding operations. Of all the animals in our food system today, 99 percent live on factory farms—enormous, vertically integrated operations designed to make as much profit as possible (at the expense of animals, people, and the environment). The transition to using animal feed has been closely intertwined with the transition to this type of large-scale factory farming.

The official term for a factory farm is concentrated animal feeding operation or CAFO. As the name implies, these operations are laser-focused on feeding large numbers of animals until they reach "slaughter weight," after which they are killed and turned into products.

The faster an animal reaches slaughter weight, the more quickly the industry profits. So factory farms have dialed in on the most efficient way to feed animals in the shortest amount of time. Rather than grazing on pasture, animals are confined in stationary cages or crowded sheds and given feed that will increase their growth rates—even while it hurts their health.

Take cows, for example. Along with sheep and other grazing animals, they are known as "ruminants"—because they have a rumen, an organ perfectly designed to transform grass into protein. But the industry feeds cows corn instead of grass because it brings them to "slaughter weight" much faster than grazing. Sadly, this high-starch diet can disturb a cow's rumen, causing pain with severe bloat, acidosis (or heartburn), and other types of stomach upset.

When it comes to feeding animals on factory farms these are some key industry terms to know:

- **Growth rates:** This is the rate at which an animal grows or how quickly the animal reaches "slaughter weight." Sadly, most factory farm animals are bred to grow so quickly that their health suffers. Chickens raised for meat frequently develop bone deformities, muscle diseases like white striping, and heart problems. Many chickens have difficulty walking, or even just standing due to painful lameness as a consequence of their fast growth rate.
- **Feed conversion ratio:** This is the ratio between the amount of feed an animal eats and the amount of body weight that an animal gains. In other words, a feed conversion ratio is the industry's effort to feed animals as little as possible to make them grow as quickly as possible.
- **Selective breeding:** This is the practice of breeding two animals to produce offspring with a desired trait. For example, the poultry industry breeds birds who quickly develop outsized breast muscles. In the meat industry, selective breeding is generally used to optimize both feed conversion ratio and growth rates.

Animal Feed Industry Impacts

Overall, factory farming is incredibly resource-intensive and harmful to the environment. From agricultural runoff to water waste and pollution, CAFOs are responsible for some of humanity's worst climate impacts.

"Livestock farms generate about 70 percent of the nation's [United States] ammonia emissions, plus gases that cause global warming, particularly methane," according to the Public Broadcasting Service. The practice of growing crops for animal feed is one of the worst drivers of environmental destruction—leaving biodiversity loss, deforestation, and greenhouse gas emissions in its wake.

Deforestation

Growing crops necessary to feed huge numbers of animals to support human meat consumption requires vast amounts of land, which results in massive deforestation. Forests worldwide are systematically being cleared and replanted with monocrops (such as the corn and soybeans mentioned earlier) to meet the demand for animal products—and therefore, animal feed.

Brazil, for example, is the world's biggest beef exporter. In the Amazon rainforest—nearly two-thirds of which is part of Brazil—crops for animal feed are one of the primary drivers of deforestation, damaging an essential habitat for countless species. Deforestation rates have averaged nearly 2 million hectares yearly since 1995 in the Amazon, or about seven football fields every minute.

Meanwhile, farmland expansion accounts for 90 percent of deforestation worldwide, "including crops grown for both human and animal consumption, as well as the clearing of forests for animal grazing," according to a July 2022 article in Sentient Media.

Deforestation eliminates one of our best defenses against climate change as healthy, intact forests provide a crucial ecosystem service: carbon sequestration. Forests safely store more carbon than they emit, making them powerful "carbon sinks" critical to maintaining a stable climate. When we destroy forests for farmland and other uses, we remove that carbon sink and release all the carbon into the atmosphere that had been stored there.

Biodiversity Loss and Extinction Threat

Naturally, deforestation goes hand in hand with biodiversity loss—of which animal agriculture is also a key driver. A 2021 study found that land use conversions to support the "global food system" are a primary driver of biodiversity loss. Tragically, researchers project that more than 1,000 species will lose at least a quarter of their habitats by 2050 if meat consumption continues at the same rate.

At the UN Biodiversity Conference (COP15) in Montreal in December 2022, delegates warned that if our land-intensive eating habits don't change, more and more critical species will go extinct. As author and journalist Michael Grunwald points out in the New York Times: "[W]hen we eat cows, chickens, and other livestock, we might as well be eating macaws, jaguars, and other endangered species."

Water Use

Along with vast amounts of land, growing crops for animal feed requires enormous quantities of water. In the U.S. alone, more than 60 percent of freshwater was used to grow crops in 2012, and around 2.5 trillion gallons per year of water was used for animal feed in the same year. Corn, soybeans, and the other grains used in animal feed require about 43 times more water than grass or roughage, which animals could access if they were allowed to graze.

Soil Degradation

The intensive farming practices required to grow vast amounts of crops—like corn and soybeans—even take a toll on the soil.

Healthy soil contains millions of living organisms, which naturally replenish and recycle organic material and nutrients. Soil filters water, stores carbon, and allows for carbon, nitrogen, and phosphorus cycles that are critical for life on Earth.

But intensive farming practices, like growing "monocultures" (huge amounts of one crop like corn or soybeans), can degrade soil and deplete critical nutrients. Not only do these farming practices prevent soil's natural processes but they can also reduce the amount of carbon stored in soil—a huge problem in the face of climate change. Intensive agriculture, closely intertwined with factory farming, damages the soil beyond repair.

Change Is Possible

The impacts of our animal-based food production system are far-reaching and complex. The intensive farming practices that supply animal feed for factory farms are destroying our water, air, and soil—and harming countless animals raised in food supply chains. But there is hope. It's not too late to build a better food system from the ground up.

The movement to build a healthier food system is growing every day. Around the world, people are advocating for systemic change—from plant-based food options to better treatment of farmed animals. In fact, according to a March 2022 article in Phys.org, "switching to a plant-based diet in high-income nations would save an area the size of the EU worldwide." Moreover, if just one person follows a vegan diet, an average of 95 animals will be spared each year, according to the book, *Ninety-Five: Meeting America's Farmed Animals in Stories and Photographs*.

Concerned citizens and consumers can also hold corporations accountable for animal abuse and environmental degradation—by pressuring companies to adopt more sustainable practices. Already, several large meat producers and fast food and supermarket chains have stopped keeping pigs in gestation crates after people expressed "disgust" at the practice. According to the New York Times, "[T]he tide is turning because consumers are making their preferences known."

> *Click here to read the article on the Observatory.*

This article was produced by Earth | Food | Life.

***Vicky Bond** is a veterinary surgeon, animal welfare scientist, and the president of The Humane League, a global nonprofit organization working to end the abuse of animals raised for food through institutional and individual change. She is a contributor to the Observatory. Follow her on Twitter @vickybond_THL.*

Photo Credit: Albert Bridge / Wikimedia Commons

Understanding Global Warming Potentials

epa.gov/ghgemissions/understanding-global-warming-potentials

January 12, 2016

Greenhouse gases (GHGs) warm the Earth by absorbing energy and slowing the rate at which the energy escapes to space; they act like a blanket insulating the Earth. Different GHGs can have different effects on the Earth's warming. Two key ways in which these gases differ from each other are their ability to absorb energy (their "radiative efficiency"), and how long they stay in the atmosphere (also known as their "lifetime").

Starting in 1990, the Intergovernmental Panel on Climate Change (IPCC) used the Global Warming Potential (GWP) to allow comparisons of the global warming impacts of different gases. Specifically, it is a measure of how much energy the emission of 1 ton of a gas will absorb over a given period of time, relative to the emission of 1 ton of carbon dioxide (CO_2). The larger the GWP, the more that a given gas warms the Earth compared to CO_2 over that time period. The time period usually used for GWPs is 100 years. GWPs provide a common unit of measure, which allows analysts to add up emissions estimates of different gases (e.g., to compile a national GHG inventory), and allows policymakers to compare emissions reduction opportunities across sectors and gases.

- CO_2, by definition, has a GWP of 1 regardless of the time period used, because it is the gas being used as the reference. CO_2 remains in the climate system for a very long time: CO_2 emissions cause increases in atmospheric concentrations of CO_2 that will last thousands of years.

- Methane (CH_4) is estimated to have a GWP of 27-30 over 100 years. CH_4 emitted today lasts about a decade on average, which is much less time than CO_2. But CH_4 also absorbs much more energy than CO_2. The net effect of the shorter lifetime and higher energy absorption is reflected in the GWP. The CH_4 GWP also accounts for some indirect effects, such as the fact that CH_4 is a precursor to ozone, and ozone is itself a GHG.
- Nitrous Oxide (N_2O) has a GWP 273 times that of CO_2 for a 100-year timescale. N_2O emitted today remains in the atmosphere for more than 100 years, on average. (Learn why EPA's U.S. Inventory of Greenhouse Gas Emissions and Sinks uses a different value.)
- Chlorofluorocarbons (CFCs), hydrofluorocarbons (HFCs), hydrochlorofluorocarbons (HCFCs), perfluorocarbons (PFCs), and sulfur hexafluoride (SF_6) are sometimes called high-GWP gases because, for a given amount of mass, they trap substantially more heat than CO_2. (The GWPs for these gases can be in the thousands or tens of thousands.)

Explore the questions and answers below to learn more about global warming potentials (GWPs).

Frequently Asked Questions

- Why does the IPCC definition of GWP differ from the definitions used in ISO (e.g., 14044 and 21930:2017) and related Environmental Product Declarations and Product Category Rules?
- Why do GWPs change over time?
- Why are GWPs presented as ranges?
- What GWP estimates does EPA use for GHG emissions accounting, such as the *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (Inventory) and the Greenhouse Gas Reporting Program?
- Are there alternatives to the 100-year GWP for comparing GHGs?

Why does the IPCC definition of GWP differ from the definitions used in ISO (e.g., 14044 and 21930:2017) and related Environmental Product Declarations and Product Category Rules?

> The International Organization for Standardization (ISO) community differs in its definition and use of the term Global Warming Potential (GWP) from that used by IPCC. This ISO approach is applied in Environmental Production Declaration (EPD), Product Category Rules (PCR), Buy Clean Policies, and related programs. This definition and use are inconsistent with how GWP is defined by the IPCC and used in many international GHG accounting efforts, including national reporting by Parties to the UNFCCC and Paris Agreement.
>
> The ISO and relevant communities use the term "GWP" as an impact category to refer to the embodied greenhouse gases of a specific product or product-level GHG emission intensities (see, e.g., ISO 21930:2017). This specific use of GWP by the EPD community refers to the total greenhouse gas emissions directly associated with the production of a product, including the upstream activities of extraction and transport of raw materials. This type of calculation can also be described with terms such as "embodied GHG equivalent" or "GHG footprint." The product GWP measure is reported in CO_2-equivalents per functional unit in EPDs, PCRs, etc. However, the ISO calculation of CO_2-equivalents requires the use of the original GWP as defined by IPCC, thereby making the EPD/ISO GWP inherently confusing as it uses both meanings of the term GWP simultaneously.
>
> To reduce confusion, the use of the term "Global Warming Potential" or "GWP" that fall outside the IPCC definition or use—i.e., a measure of the relative climate impact of a given greenhouse gas relative to the impact of carbon dioxide (as defined on this page)—should include a definition of the non-IPCC usage of the term to distinguish it from the original established IPCC definition. In the case of how ISO and relevant communities use the term GWP, it should be clearly explained that the specific meaning in that context refers to "embodied GHG equivalent," "embodied GHG emissions," or "carbon equivalent footprint," as applicable. This context is especially important if the document uses both different meanings of the term "GWP" such as in the ISO/EPD context.

Why do GWPs change over time?

> EPA and other organizations will update the GWP values they use occasionally. This change can be due to updated scientific estimates of the energy absorption or lifetime of the gases or to changing atmospheric concentrations of GHGs that result in a change in the energy absorption of 1 additional ton of a gas relative to another.

Why are GWPs presented as ranges?

> In the most recent report by the Intergovernmental Panel on Climate Change (IPCC), multiple methods of calculating GWPs were presented based on how to account for the influence of future warming on the carbon cycle. For this Web page, we are presenting the range of the lowest to the highest values listed by the IPCC.

What GWP estimates does EPA use for GHG emissions accounting, such as the *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (Inventory) and the Greenhouse Gas Reporting Program?

> The EPA considers the GWP estimates presented in the most recent IPCC scientific assessment to reflect the state of the science. In science communications, the EPA will refer to the most recent GWPs. The GWPs listed above are from the IPCC's Sixth Assessment Report, published in 2021.
>
> The EPA's *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (Inventory) complies with international GHG reporting standards under the United Nations Framework Convention on Climate Change (UNFCCC). UNFCCC guidelines now require the use of the GWP values from the IPCC's Fifth Assessment Report (AR5), published in 2013. The Inventory also presents emissions by mass, so that CO_2 equivalents can be calculated using any GWPs, and emission totals using more recent IPCC values are presented in the annexes of the Inventory report for informational purposes.
>
> The data collected by EPA's Greenhouse Gas Reporting Program is generally reported in mass units of greenhouse gas and is used in the Inventory. The Reporting Program, generally uses GWP values from the AR4 to determine whether facilities exceed reporting thresholds and to publish data in CO_2 equivalent values. The Reporting Program collects data about some industrial gases that do not have GWPs listed in the AR4; for these gases, the Reporting Program uses GWP values from other sources, such as the AR5.
>
> EPA's CH_4 reduction voluntary programs also use CH_4 GWPs from the AR5 report for calculating CH_4 emissions reductions through energy recovery projects, for consistency with the national emissions presented in the Inventory.

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Are there alternatives to the 100-year GWP for comparing GHGs?

The United States primarily uses the 100-year GWP as a measure of the relative impact of different GHGs. However, the scientific community has developed a number of other metrics that could be used for comparing one GHG to another. These metrics may differ based on timeframe, the climate endpoint measured, or the method of calculation.

For example, the 20-year GWP is sometimes used as an alternative to the 100-year GWP. Just like the 100-year GWP is based on the energy absorbed by a gas over 100 years, the 20-year GWP is based on the energy absorbed over 20 years. This 20-year GWP prioritizes gases with shorter lifetimes, because it does not consider impacts that happen more than 20 years after the emissions occur. Because all GWPs are calculated relative to CO_2, GWPs based on a shorter timeframe will be larger for gases with lifetimes shorter than that of CO_2, and smaller for gases with lifetimes longer than CO_2. For example, for CH_4, which has a short lifetime, the 100-year GWP of 27–30 is much less than the 20-year GWP of 81–83. For CF_4, with a lifetime of 50,000 years, the 100-year GWP of 7380 is larger than the 20-year GWP of 5300.

Another alternate metric is the Global Temperature Potential (GTP). While the GWP is a measure of the heat absorbed over a given time period due to emissions of a gas, the GTP is a measure of the temperature change at the end of that time period (again, relative to CO_2).The calculation of the GTP is more complicated than that for the GWP, as it requires modeling how much the climate system responds to increased concentrations of GHGs (the climate sensitivity) and how quickly the system responds (based in part on how the ocean absorbs heat).

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Nitrate contamination of Minnesota waters shows little sign of going away, despite years of effort

www2.startribune.com/nitrate-pollution-minnesota-groundwater-farm-fertilizer-mpca-wells-epa/600310942

Jeff Hargarten, Jennifer Bjorhus, Star Tribune

Farm pollution persists despite hundreds of millions spent to clean it up.

By Jeff Hargarten and Jennifer Bjorhus Star Tribune
November 28, 2023 — 6:20pm

Brian Peterson, Star Tribune file

Utica, Minn., a farm town of about 200 people surrounded by fields in Winona County, is preparing to dig a new, deeper well to find clean water because the city's water is contaminated with nitrate. It's one example of the state's stubborn problem with reducing nitrate contamination from farm pollution.

Minnesota has spent hundreds of millions of dollars and decades of effort to reduce nitrate that's contaminating drinking water and rivers. The progress so far: negligible.

The main source of the nitrate is nitrogen fertilizer, a pillar of production agriculture that includes animal manure and synthetic chemicals. Farmers apply tens of thousands of tons of fertilizers to their fields every year, and what isn't absorbed by crops can seep into aquifers and any runoff can end up in rivers.

Despite numerous programs designed to encourage farmers to change their ways, purchases of fertilizer keep growing. In many parts of Minnesota farm country, drinking water wells and streams carry that legacy: A decades-old state law limits how much nitrate is allowed in drinking water, although some researchers now say that level needs to be much stricter to protect people.

The three agencies tasked with keeping Minnesota waters clear of harmful levels of nitrate acknowledge that the situation isn't improving, particularly for private wells in the vulnerable topography of the state's hilly southeastern corner. In that region, frustrated residents have called for the federal government to intervene on what environmental groups call a public health emergency — and the EPA recently responded with a directive that Minnesota clean up its act.

A lack of progress

Nitrate levels of 10 milligrams per liter of water or higher have violated federal health standards since the 1960s, since those concentrations are known to cause the potentially life-threatening condition methemoglobinemia, or blue baby syndrome, that starves infants of oxygen.

But there's a push to reduce the state and federal nitrate standard from the 10 mg/l limit, given growing research around links to cancer and other damaging health impacts from drinking water with nitrate at half the legal maximum concentration, or even lower.

Community drinking water supplies, which serve cities, towns and mobile home parks, are regularly tested to assure nitrate levels are below the state and federal health limit.

While those with the highest nitrate concentrations have taken action to reduce it, about 177,000 Minnesotans still lived in communities with average readings above 3 milligrams of nitrate per liter of water as of 2022, levels considered by health authorities to be caused by human activity, not nature.

At least 400,000 Minnesotans in more than 100 communities live in areas where water has tested at least once for elevated nitrate levels since 2013. They're mostly spread across central and southern parts of the state.

Separately, there are some 980,000 private wells in Minnesota, according to the Minnesota Well Owners Organization. And people who rely on them for drinking water are on their own to have them tested and, if necessary, find remedies.

Far more Minnesotans could be affected by elevated nitrate levels in their water, but a lack of one central testing agency means it is difficult to gather and compare data.

The volunteer private well tests the Department of Agriculture has helped run show the problem is widespread. In southeast Minnesota from 2008 through 2018, about 8% to 15% of the hundreds of private wells tested each year showed nitrate pollution above the 10 mg/L health limit. In 2021, about 30% of those private wells showed results above 3 milligrams.

In the 14-county Central Sands Region from 2011-2018, about 3% to 5% of the hundreds of private wells tested each year were polluted with nitrate above the 10 mg/L limit.

Public drinking water systems — not private wells — that violate federal nitrate contamination standards must report them to the EPA. Those violations in Minnesota totaled 34 last year in the EPA's Safe Drinking Water Information System and included gas stations, bars and churches.

Impaired rivers and streams

Nitrate also endangers fish and other aquatic life when it leaches into lakes, streams and rivers.

The nitrate entering the Mississippi River contributes to the huge oxygen-starved dead zone in the Gulf of Mexico. As part of the Hypoxia Task Force of states up and down the river, Minnesota has pledged to cut the nitrate in the Mississippi by 20% by 2025. But nitrate has actually risen in spots, as it has in most of the state's major rivers.

Lawmakers directed the Minnesota Pollution Control Agency in 2010 to set limits on nitrate to protect fish and aquatic life. It hasn't happened. It would be too expensive for small wastewater treatment plants, and wouldn't effectively reduce the nitrate from the farms it has no power to regulate, the agency told the Star Tribune.

About 5% — or 165 miles — of Minnesota's rivers and streams used for drinking water are impaired by nitrogen and/or phosphorus as of 2022, meaning they don't meet federal quality standards. In all, the EPA lists more than 300 bodies of water across the state including parts of the Minnesota, Mississippi and St. Croix rivers, as well as other streams and rivers, as threatened or impaired by nitrogen and phosphorus and in need of a restoration plan.

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Spending with little impact

Hundreds of millions in federal and state funding has paid for nitrate research, efforts to change farming and other practices and nitrate filtration systems for water supplies in Hastings, Cold Spring, Adrian and four other cities.

That's paid for Nitrogen Smart farmer training in the past, water research, conservation programs, source water protection work and guidance for farmers on adopting best management practices — and that's just a few examples. The state covers this list in its five year progress reports on the state's 2014 Nutrient Reduction Strategy to cut nitrogen and phosphorus in waters.

The state's Clean Water Fund, part of the sales-tax funded Legacy Amendment, has directed at least $148 million to the nitrate problem since 2010, according to a Star Tribune analysis, and is just one of several spending sources.

None of it appears to have made a dent in the overall demand for nitrogen fertilizer. As cropland has expanded, farmers bought a record high 824,000 tons of nitrogen fertilizer in 2020, the most recent year for which data is available, according to sales tracked by the state Department of Agriculture.

Agency response

The responsibility for reducing nitrate lies mostly with three state agencies: Minnesota Department of Health, the Minnesota Department of Agriculture and the Minnesota Pollution Control Agency (MPCA). All said their efforts will pay off eventually.

The MPCA blamed climate change's effect on precipitation for the failure to show progress on nitrate reduction.

"It will take time to see the benefit of this work, especially as more frequent and extreme weather events caused by climate change are both masking our progress and worsening the nitrate problem by forcing nitrate pollution off lands, into groundwater, rivers, and downstream," said MPCA spokeswoman Andrea Cournoyer.

The Health Department said 30 years of data doesn't show increasing nitrate violations in the public water supplies it watches, but that it's a "different story" for private well owners in certain highly vulnerable parts of the state.

The Agriculture Department agrees that in parts of southeast Minnesota, the nitrate in private water wells "has been going up slowly for decades."

"Nowhere in the U.S. is a state tackling nitrate issues like Minnesota," the agriculture department said.

Some in southeast Minnesota, a land of heavy agriculture and a porous karst geography, say they can't wait any longer for help. A group from Dodge, Goodhue, Fillmore, Houston, Mower, Olmstead, Wabasha and Winona counties asked the EPA to declare a public health emergency because state and local authorities haven't controlled nitrate pollution of groundwater.

About 80,000 residents in those counties rely on private wells for their drinking water and about 300,000 use public water systems, according to the request for help, filed in April by the Minnesota Center for Environmental Advocacy, the Minnesota Well Owners Association and others.

The EPA responded with a letter this month, warning Minnesota's three responsible agencies of possible enforcement actions if they don't enact measures to better warn residents of nitrate dangers, provide bottled water and develop plans to reduce nitrate pollution in the region.

Further reading

Sources: Environmental Protection Agency, EPA Safe Drinking Water Information System, Minnesota Pollution Control Agency, Minnesota Department of Health, Minnesota Department of Agriculture, Environmental Working Group, Star Tribune reporting and analysis

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Commentary | Open Access

Confronting our Agricultural Nonpoint Source Control Policy Problem

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Research Impact Statement: Reform of agricultural nonpoint source pollution policies is necessary to make progress in achieving water quality goals.

Abstract

Federal and state agricultural and environmental agencies have spent enormous sums since the 1990s to reduce nonpoint source (NPS) water pollution from agriculture. Yet, water quality problems are pervasive, and agriculture is a major cause. The lack of progress is often attributed to insufficient funding for pollution control practices relative to the scale of the problem. However, we attribute the lack of progress to shortcomings in agricultural NPS pollution control policy. We illustrate our argument after considering nearly four decades of federal, state, and local efforts to reduce agricultural NPS pollution to the Chesapeake Bay. Additional funding for current programs, absent fundamental program reform, is unlikely to produce reductions from agriculture needed to achieve desired water quality outcomes.

INTRODUCTION

The 1972 Federal Clean Water Act ushered in a new era of state and federal regulation, supported by enormous public and private spending directed at restoring, in the words of the Act "the physical, biological and chemical integrity of the nation's waters." Now, 50 years later, States (U.S.) rivers and streams are in poor biological condition, 25% are in fair condition, and only 28% are in good condition (USEPA 2017). Agricultural nonpoint source (NPS) pollution is often a principal cause of water quality impairments. The 2017 USEPA National Water Quality Inventory lists agricultural NPS pollution as the leading cause of water quality impairments in rivers and streams, the third-largest cause for lakes, the second largest for wetlands, and a major contributor to contamination of estuaries and groundwater (USEPA 2017). Agriculture is the largest source of nutrients contributing to the eutrophication of the Gulf of Mexico, the Chesapeake Bay, and the Great Lakes (Goolsby et al. 1999; Howarth 2008).

Policies and programs to reduce agricultural NPS pollution rely primarily on agricultural producers voluntarily implementing pollution control practices, encouraged by technical and financial assistance from federal and state programs. These programs have achieved only limited successes in reducing agricultural NPS loads (Sprague et al. 2011; Shortle et al. 2012; Ator et al. 2020). This widely acknowledged gap between NPS reductions achieved and the amount needed to meet water quality goals often is attributed to insufficient funding for existing technical and financial assistance programs (DeGood 2020).

We argue that increased funding is not enough. The limited success of NPS programs is embedded in the structure of the programs, and how these programs guide and direct choices; choices made by agricultural producers, technical assistance providers who advise producers, and water quality program managers. We illustrate our argument with the Chesapeake Bay Program (CBP) and conclude that fundamental policy reforms will be needed for achieving substantial reductions in agricultural NPS loads.

EXISTING APPROACHES TO ADDRESS AGRICULTURAL NPS POLLUTION

Programs to encourage agricultural NPS load reductions can take a variety of forms (Segerson 2013; OECD 2017; Shortle 2017; Pannell and Classen 2020; Shortle et al. 2021). With some exceptions, [1] conventional agricultural NPS policy in the U.S. rests on the premise that agricultural producers voluntarily decide how to manage their operations and whether or how to reduce to NPS pollution. To encourage producers to implement NPS pollution control practices, information programs inform them of the best management practices (BMPs) intended to improve water quality, and points them to government funding and sometimes private funding available for implementing the practices. Because BMPs can be costly, and in many cases reduce producers' net income, programs typically encourage implementation by sharing implementation costs (Shortle et al. 2021). Federal and state technical assistance

Environmental Quality Incentives Program and there are state programs that also cost-share BMP implementation. Cost sharing is typically for a portion of the cost of BMP installation and in limited circumstances some annual operation and maintenance costs (Ribaudo and Shortle 2019; Ribaudo 2001).

The Limited Success of NPS Programs: Illustration from the Chesapeake Bay

The CBP illustrates the current agricultural NPS pollution policy conundrum. Since the 1980s, nitrogen and phosphorus were identified as the primary pollutants limiting attainment of desired Bay water quality outcomes and agricultural NPS was identified early on as a major contributor of nutrient loads. In 1990, nutrient reduction targets for meeting water quality goals were set, these targets were revised in the 2000s, and brought under federally mandated nutrient limits in 2010.[2] Through all these years, policymakers understood that without agricultural NPS load reductions nutrient reduction targets and desired water quality outcomes would be unattainable.

For over three decades, federal and state governments have been committed to funding the types of conventional technical and financial cost share programs described above, hoping to encourage BMP implementation and meet agricultural NPS reduction targets. In fact, the CBP has been successful in increasing federal and state funding to support these programs, including securing a special federal appropriation of $256 million for the NRCS Chesapeake Bay Watershed Initiative (Natural Resource Conservation Service 2021). Recently, there have been renewed efforts to increase NPS program funding, arguing that more funding for these conventional programs will finally secure Bay water quality goals (Northey 2021).

Also of note is that the CBP has invested substantial resources to build a state-of-the-art model to evaluate and inform water quality managers decision-making (Hood et al. 2021). With respect to NPS pollution, the CBP watershed model is the basis for prioritizing BMP implementation and for crediting progress toward meeting NPS load reduction targets. As BMP implementation is reported, the CBP model credits NPS reductions by multiplying model-based estimates of nutrient runoff (pounds per acre) by an assigned BMP removal efficiency and the number of acres treated by the BMP. The model calculates nutrient runoff as an average over a relatively large area (~20,000 acres) for different land use types (crop, hay, etc.). The BMP pollutant removal efficiencies are generally a single number (e.g., 30% N removal for a riparian buffer) applied across the watershed. In the CBP model, these removal efficiencies are usually generated by expert judgment from a group of subject matter authorities (Stephenson et al. 2018).

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the CBP model estimates, all pollution controls (implemented since 1985) have reduced nitrogen (N) loads by nearly 100 million pounds and phosphorus loads 14 million pounds per year. Over three-quarters of N reductions have come from wastewater treatment plants and from reductions in atmospheric deposition (CBP 2021). Point sources have reduced P loadings about 80% since 1985, which represents about 70% of the estimated P reductions achieved since 1985. The CBP identifies a need for another 50 million pounds of N reductions by 2025 to meet water quality standards, noting that these reductions must come primarily from agricultural NPS.

Meanwhile, statistical analyses of monitoring data suggest that the CBP model may be overestimating the nutrient reductions achieved by the cumulative impact of agricultural BMPs. Ator et al. (2019) found little evidence that agricultural NPS loads declined between 1992 and 2012. Another statistical analysis of monitoring data found that while P loads are declining in some regions of the Bay watershed, those improvements were offset by increases in agricultural P sources in other areas (Fanelli et al. 2019; Kleinman et al. 2019; Ator et al. 2020). While the limited response in observed pollutant reductions could be due the time that is required for NPS reductions to produce ambient water quality outcomes, the so-called “lag times,” evidence suggests that another cause is at play: our agricultural NPS programs are not as effective as expected. The CBP is not alone in confronting this NPS challenge. Reductions from BMP implementation predicted by models routinely over estimate measured reductions (Osmond et al. 2012; Lintern et al. 2020). The challenge of measuring reductions in NPS loads in response to BMP adoption is one of the most fundamental and common challenges confronting large-scale water quality programs (Osmond et al. 2012; Boesch 2019; Lintern et al. 2020).

CHALLENGES WITH AGRICULTURAL NPS INCENTIVES

The continued failure to meet agricultural NPS reduction goals is not simply due to a lack of funding or a lack of effort. To a significant degree, the problem lies with the incentives inherent in conventional program design. These incentives influence choices made by producers and technical service providers that often limit the implementation of cost-effective BMPs in the locations that produce the greatest NPS loads. The following illustrations of NPS incentive challenges are drawn from the CBP, but these challenges are common across most large-scale water quality programs.

Agricultural Producers Face Limited Financial Incentives to Address NPS Pollution

sharing for practices is unlikely to result in producers and technical service providers seeking to identify water quality problem areas, and then implementing the most effective BMPs. Consider a BMP that produces little agronomic benefit to a producer's operation, promises significant low-cost nutrient reductions but requires substantial upfront capital investment and ongoing operation and maintenance expenditures. BMPs such as stream buffers, denitrifying bioreactors, stream fencing, and manure storage/treatment can generate substantial nutrient reductions at relatively low costs (Price et al. 2021; Stephenson et al. 2021). From a strictly financial perspective, agricultural producers will not install and operate a technology with few on-farm benefits and that costs them money (even if cost-shared). The structure of our cost-share programs does not directly pay producers for what is needed: pollutant reductions.

Program Managers Have Limited Ability and Incentives to Target NPS Hotspots

Many studies demonstrate that relatively small portions of the agricultural landscape produce most of the agricultural load. The way NPS loads are counted and reductions are credited is a disincentive for program managers to identify and treat these high loss areas. Suppose that 80% of nutrient losses on a 250-acre farm is coming from only 25 acres. The CBP crediting system and technical assistance programs provide few incentives for technical service providers and producers to focus on those 25 acres. If the 250 acres is in the same land use (say corn), CBP crediting gives the same reduction credit whether the BMP is placed on any of the 225 low loss acres, or the 25 high loss acres. Furthermore, conventional programs typically require that agricultural producers develop conservation plans for the entire farm operation to be eligible for program benefits. A producer willing to aggressively treat only the 25 high loss acres might not want or need a whole farm plan and, under current program guidelines, the producer would be ineligible for financial assistance without a plan that covers the entire farm.

Technical Service Providers Are Not Rewarded for Loads Reduced

Technical service providers serve as the conduit between the entity funding BMP implementation and producers, providing engineering, installation, and maintenance assistance to producers, and facilitating financial assistance. This structure, the technical service provider as a liaison, provides no direct incentive for a service provider to prioritize reductions from difficult and often high loading areas. Suppose a service provider can work with two neighboring producers. One producer has low nutrient losses and willingly adopts conservation practices. The other producer has high nutrient losses and is reluctant to participate in government programs. Such diversity of producer behavior is real and can be substantial (Ribaudo 2015). One recent study in a portion of the Chesapeake Bay watershed as measured by contracts processed. When programs, as in the CBP, credit nutrient reductions by BMPs installed, the same NPS reduction credit applies to both producers. Spending time with a reluctant, but high nutrient loss producer is a poor investment of the service providers time when the measure of success is participants enrolled and BMPs installed.

Technical Service Providers and Water Quality Managers cannot “Go Big”

Cost-share programs typically cap the amount of assistance that can be received by an individual agricultural producer. While distributing funding over more participants helps engage more producers in the conservation program, such funding limitations restrict what water quality managers and technical service providers can do to address larger scale NPS issues. For example, at a regional level, nutrient losses tend to be highest in areas with nutrient mass imbalances, where nutrient imports, in the form of fertilizers and animal feed, exceed the ability of the local cropping system to utilize the nutrients. The use of conventional BMPs, most of which do not address excessive nutrient mass imbalances, offers limited potential to reduce NPS loads. Regional animal waste management systems (manure conversion, waste to energy projects, transport) offer opportunities to address regional nutrient mass imbalances, but given the large upfront and ongoing maintenance and operation cost, and lack of on farm benefits associated with such systems limit their uptake.

Barriers to Innovation Exist in Current Program Structure

Incentives for innovation in NPS technologies and management are weak. Under conventional cost-share programs, entrepreneurs face limited profit opportunities to develop innovative NPS control practices because conventional agricultural cost-share programs create no buyers for such products. Producers have no incentive to pay for these technologies (unless there are on-farm benefits) and water quality managers have no means to pay for them given the requirement that costs must be shared.

Water quality managers, agricultural producers, and technical service providers have few incentives to invest in actions that produce more certain load reductions. Consider a producer who wants to implement a BMP where pollutant removal can be more readily measured or observed, for example in situ nutrient extraction (measurement of aquatic biomass harvest), direct treatment of runoff or water (influent and effluent from bioreactors), or manure conversion technologies, among others (Rose et al. 2015; Stephenson and Shabman 2017; Stephenson et al. 2018). Consider another example of a producer who is willing to demonstrate intermediate outcomes from conservation activities, such as changes in soil nutrient levels or

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POLICY REFORM IS NEEDED TO MAKE PROGRESS IN ADDRESSING NPS POLLUTION

Conventional NPS program designs create limited incentives for program managers, technical service providers, or producers to care about whether BMPs provide the expected NPS load reductions. As a result, it is unlikely that significant progress will be made on NPS load reduction without fundamental policy and programmatic change (Shortle et al. 2012, 2021; Ribaudo and Shortle 2019). The most fundamental change would replace the current program premise that producers decide both whether and how to control their pollution, with a new premise that a producer or group of producers is obligated to limit their pollution but has discretion and flexibility in deciding how that limit is met.

Whatever the program premise, first, reform must shift the focus from practices to outcomes. Incentive systems that reward quantifiable nutrient reductions or observable water quality outcomes, such as “pay-for-performance” (“pay-for-success”) systems, may better motivate agricultural producers to seek out and implement practices that result in the largest NPS reductions. Payment for performance programs can be designed in a variety of ways, but all should require that technical service providers also be able and willing to evaluate all NPS reduction options and develop plans for reducing pollutants. [3]

Second, the focus on outcomes through a “pay-for-performance” (“pay-for-success”) system will require establishing acceptable practices for quantifying either pollutant reduction or changes in water quality conditions. Outcomes can be documented by direct measurement, by indirect, but observable, indicators of pollutant loss potential (e.g. soil nutrient levels), or by using more sophisticated field-scale models to predict site-specific reductions from implemented BMPs. Measured outcomes can be used for determining when the producers would be paid under the pay for performance system or for determining if the limits are being met. Measured outcomes allow technical service providers to be rewarded for working with high loss producers and for targeting high loss areas, and measured outcomes mean water quality managers' report progress as quantified load reductions, or improvement in ambient water quality conditions.

A shift toward outcome-based program design should involve experimentation with innovative combinations of incentive systems and outcome-based measurement (Shabman et al. 2011). As one example, producer-led watershed cooperatives could be created with the assistance of technical service providers to achieve measurable water quality. Such organizations would be incentivized to achieve specific quantifiable, independently verified, water quality outcomes, for instance, at the outlet of small watersheds by offering reward or bonus payments made to the

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producers to identify and direct cost-share funds received from the NPS programs to the investments that yield the most effective, more certain pollutant reductions (Maille et al. 2009).

Third, as noted above, reform may require shifting away from the premise of conventional programs that producers decide based on financial or personal adoption benefits whether to limit NPS pollution, to program designs that obligate producers to limit their NPS pollution. Such mandatory limits must be structured to recognize the diversity in agriculture across scales and across production systems. Consider large regional nutrient mass imbalances from high concentrations of intensive livestock operations. In vertically integrated production systems, such as poultry and swine, manure ownership and management requirements could be assigned to the integrator, rather than individual producers working under contract with the integrator. The integrator would be responsible for meeting manure disposal requirements but would be allowed the flexibility and technical expertise to find cost-effective solutions for the treatment, transport, and use of the manure.

Reform may mean that some agricultural producers accept more responsibilities for delivering pollutant reductions. Reform can mean more funding to existing programs given that funding requests often exceed available program funds, but new funding must be dedicated to paying for outcomes. Reform must mean that water quality managers rely more on measured outcomes, rather than tallying BMPs installed when determining progress. Reform must mean that agencies invest in training technical service providers in new skills needed to execute new program designs and embrace changes to familiar program and reward systems.

The challenges to making this transition are many and transition will not come easily. Reform will meet resistance. Acknowledging the need for change is the first step, and that will require accepting that we cannot simply buy our way out of the problem by spending more money on conventional, voluntary programs.

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Kurt Stephenson: Conceptualization; writing – original draft. Leonard Shabman: Conceptualization; writing – original draft. James Shortle: Conceptualization; writing – review and editing. Zachary Easton: Conceptualization; writing – original draft.

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DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

LITERATURE CITED

Ator, S.W., J.D. Blomquist, J.S. Webber, and J.G. Chanat. 2020. "Factors Driving Nutrient Trends in Streams of the Chesapeake Bay Watershed." *Journal of Environmental Quality*. https://doi.org/10.1002/jeq2.20101.

PubMed | Web of Science® | Google Scholar

Ator, S.W., A.M. Garcia, G.E. Schwarz, J.D. Blomquist, and A.J. Sekellick. 2019. "Toward Explaining Nitrogen and Phosphorus Trends in Chesapeake Bay Tributaries, 1992–2012." *Journal of the American Water Resources Association* **55** (5): 1149–68.

CAS | Web of Science® | Google Scholar

Boesch, D. 2019. "Barriers and Bridges in Abating Coastal Eutrophication." *Frontiers in Marine Science* **6**. https://doi.org/10.3389/fmars.2019.00123.

Web of Science® | Google Scholar

CBP (Chesapeake Bay Program). 2021. " Chesapeake Progress: Modeled Nitrogen and Phosphorus Loads to the Chesapeake Bay." https://www.chesapeakeprogress.com/clean-water/watershed-implementation-plans.

Google Scholar

DeGood, K. 2020. *A Call to Action on Combating Nonpoint Source and Stormwater Pollution*. Center for American Progress. https://www.americanprogress.org/issues/economy/reports/2020/10/27/492149/call-action-combating-nonpoint-source-stormwater-pollution/
.

Google Scholar

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Fanelli, R.M., J.D. Blomquist, and R.M. Hirsch. 2019. "Point Sources and Agricultural Practices Control Spatial–Temporal Patterns of Orthophosphate in Tributaries to Chesapeake Bay." *Science of the Total Environment* **652**: 422–33.

PubMed | Web of Science® | Google Scholar

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.R. Hooper, D.R. Keeney, and G.J. Stensland. 1999. " Flux and Sources of Nutrients in the Mississippi Atchafalaya River Basin Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico: Silver Spring, MD, NOAA Coastal Ocean Office, NOAA Coastal Ocean Program Decision Analysis Series No. 17." 130 pp.

Google Scholar

Hood, R.R., G.W. Shenk, R.L. Dixon, S.M.C. Smith, W.P. Ball, J.O. Bash, R. Batiuk, et al. 2021. "The Chesapeake Bay Program Modeling System: Overview and Recommendations for Future Development." *Ecological Modeling* **456**: 109635.

Web of Science® | Google Scholar

Howarth, R.W. 2008. "Coastal Nitrogen Pollution: A Review of Sources and Trends Globally and Regionally." *Harmful Algae* **8** (1): 14–20.

CAS | Web of Science® | Google Scholar

Keiser, D.A., and J.S. Shapiro. 2019. "Consequences of the Clean Water Act and the Demand for Water Quality." *Quarterly Journal of Economics* **134** (1): 349–96.

Web of Science® | Google Scholar

Kleinman, P., R. Fanelli, B. Hirsch, A.R. Buda, Z.M. Easton, L. Wainger, C. Brosch, et al. 2019. "Phosphorus and the Chesapeake Bay — Lingering Issues and Emerging Concerns for Agriculture." *Journal of Environmental Quality* **48**: 1191–203.

CAS | PubMed | Web of Science® | Google Scholar

Kling, C.L. 2013. "State Level Efforts to Regulate Agricultural Nonpoint Sources of Water Quality Impairment." *Choices* **28** (3): 1–4.

Google Scholar

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Clean Water Organizations Comments Exhibit 33

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Diffuse Nutrient Pollution: Wicked Problems across Urban and Agricultural Watersheds." *Environmental Science & Technology*. https://doi.org/10.1021/acs.est.9b07511.

PubMed | Web of Science® | Google Scholar

Maille, P., A.R. Collins, and N. Gillies. 2009. "Performance-Based Payments for Water Quality: Experiences from a Field Experiment." *Journal of Soil and Water Conservation* **64** (3): 85A–87A.

Web of Science® | Google Scholar

Natural Resource Conservation Service. 2021. " Chesapeake Bay Watershed Initiative." https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/fb08_cp_cbwi.html https://www.nrcs.usda.gov/Internet/NRCS_RCA/reports/cp_nat.html.

Google Scholar

Northey, H. 2021. " Chesapeake Bay Farmers Facing Cleanup Deadline Seek USDA Boost." *E&E News*, September 20.

Google Scholar

OECD. 2017. *Diffuse Pollution, Degraded Waters: Emerging Policy Solutions*. Paris: OECD Studies on Water, OECD Publishing. https://doi.org/10.1787/9789264269064-en.

Google Scholar

Osmond, D., D. Meals, D. Hoag, M. Arabi, A. Luloff, G. Jennings, M. McFarland, J. Spooner, A. Sharpley, and D. Line. 2012. "Improving Conservation Practices Programming to Protect Water Quality in Agricultural Watersheds: Lessons Learned from the National Institute of Food and Agriculture–Conservation Effects Assessment Project." *Journal of Soil and Water Conservation* **67** (5): 122A–27A.

Web of Science® | Google Scholar

Pannell, D.J., and R. Classen. 2020. "The Roles of Adoption and Behavior Change in Agricultural Policy." *Applied Economic Perspectives and Policy* **42** (1): 31–41.

Web of Science® | Google Scholar

September 3, 2024
Clean Water Organizations Comments Exhibit 33

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Web of Science® Google Scholar

Price, E., T. H. Fleming, and L. Wainger. 2021. " Cost Analysis of Stormwater and Agricultural Practices for Reducing Nitrogen and Phosphorus Runoff in Maryland." Maryland Center for Environmental Science, Report number: TS-772-21. https://doi.org/10.13140/RG.2.2.28896.74246/1.

Google Scholar

Ribaudo, M. 2001. " Non-Point Source Pollution Control Policy in the USA." In *Environmental Policies for Agricultural Pollution Control*, edited by J.S. Shortle and D. Abler, 123–50. New York: CABI Publishing.

Google Scholar

Ribaudo, M. 2015. "The Limits to Voluntary Conservation Programs." *Choices* **30** (2): 1–5.

Google Scholar

Ribaudo, M., and J. Shortle. 2019. "Reflections on 40 Years of Applied Economics Research on Agriculture and Water Quality." *Agricultural and Resource Economics Review* **48** (3): 519–30.

Web of Science® Google Scholar

Rose, J., S. Bricker, S. Deonarine, J.G. Ferreira, T. Getchis, J. Grant, J.K. Kim, et al. 2015. *Nutrient Bioextraction*. New York: Encyclopedia of Sustainability Science and Technology, Springer.

Google Scholar

Segerson, K. 2013. "When is Reliance on Voluntary Approaches in Agriculture Likely to Be Effective?" *Applied Economic Perspectives and Policy* **35** (4): 565–92.

Web of Science® Google Scholar

Shabman, L., B. Rose, and K. Stephenson. 2011. "Environmental Services Programs for the Chesapeake Bay." *Choices* **26** (3).

Google Scholar

September 3, 2024
Clean Water Organizations Comments Exhibit 33

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Web of Science® Google Scholar

Shortle, J.S., M. Ollikainen, and A. Iho. 2021. *Water Quality and Agriculture: Economics and Policy for Agricultural Nonpoint Source Water Pollution*. Cham: Springer Nature.

Google Scholar

Shortle, J.S., M. Ribaudo, R.D. Horan, and D. Blandford. 2012. "Reforming Agricultural Nonpoint Pollution Policy in an Increasingly Budget-Constrained Environment." *Environmental Science & Technology* **46** (3): 1316–25.

CAS PubMed Web of Science® Google Scholar

Sprague, L.A., R.M. Hirsch, and B.T. Aulenbach. 2011. "Nitrate in the Mississippi River and its Tributaries, 1980 to 2008: Are we Making Progress?" *Environmental Science and Technology* **45** (17): 7209–16.

CAS PubMed Web of Science® Google Scholar

Stephenson, K., W. Ferris, E. Bock, and Z. Easton. 2021. "Treatment of Legacy Nitrogen as a Compliance Option to Meet Chesapeake Bay TMDL Requirements." *Environmental Science and Technology* **55** (20): 13593–601.

CAS PubMed Web of Science® Google Scholar

Stephenson, K., C. Hershner, B. Benham, Z.M. Easton, J. Hanson, S. Julius, and E. Hinrichs. 2018. " Consideration of BMP Performance Uncertainty in Chesapeake bay Program Implementation." STAC Publication Number 18-003, Edgewater, MD, 33 pp.

Google Scholar

Stephenson, K., and L. Shabman. 2017. "Nutrient Assimilation Services for Water Quality Credit Trading Programs: A Comparative Analysis with Nonpoint Source Credits." *Coastal Management* **45** (1): 24–43.

Web of Science® Google Scholar

USEPA (U.S. Environmental Protection Agency). 2017. " National Water Quality Inventory: Report to Congress." EPA 841-R-16-011.

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Citing Literature

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Groundwater quality

pca.state.mn.us/air-water-land-climate/groundwater-quality

1. Air, Water, Land, Climate
2. Water
3. Water quality

Image

Groundwater is the source of drinking water for about 75% of all Minnesotans and provides almost all of the water used to irrigate crops. Groundwater in parts of the central and southwestern regions of the state is contaminated with high nitrate concentrations from agriculture and, to a lesser extent, failing septic systems. Nitrate levels are higher in groundwater under agricultural land than water below urban areas. Groundwater availability in Minnesota varies by region. It is more difficult to access in the northeast, when it's available at all, and is scarce in some areas of the southwest.

Overall conditions

- The quality of groundwater varies around the state. Even within an aquifer, the quality can change at different depths. Near-surface groundwater in areas of high urban density or intensive agriculture is more likely to be contaminated by chloride or nitrate.
- The overuse of groundwater threatens surface water quality, and draws contaminated near-surface water into our drinking water aquifers.

Current regulations and voluntary best management practices will not be sufficient to maintain healthy groundwater and shield contaminated wells and aquifers from additional pollution. Even if all existing laws were followed to the letter, groundwater would still be subject to unacceptable levels of nutrients and other contaminants. Targeted action will be required to cut off unregulated sources of pollution.

Northeast Minnesota

Availability issues. Higher volume supplies of groundwater can be difficult to obtain in the northeast, compared to the central part of the state.

Central region

- **Availability good.** Groundwater is available throughout this region in volumes sufficient to satisfy residential use.
- **Nitrate pollution.** About 40 percent of the shallow wells (less than 30 feet deep) in the central region have higher nitrate concentrations than the EPA allows for drinking water.

Metro area and the southeast

- **Availability issues.** Though this region has multiple aquifers, groundwater availability is threatened by high consumption in the Twin Cities metro area.
- **Chloride pollution.** Groundwater in the Twin Cities metro area shows high concentrations of chloride.
- **Nitrate pollution.** Most of the sand and gravel aquifers in southern Minnesota have nitrate concentrations that exceed EPA guidelines for human health.

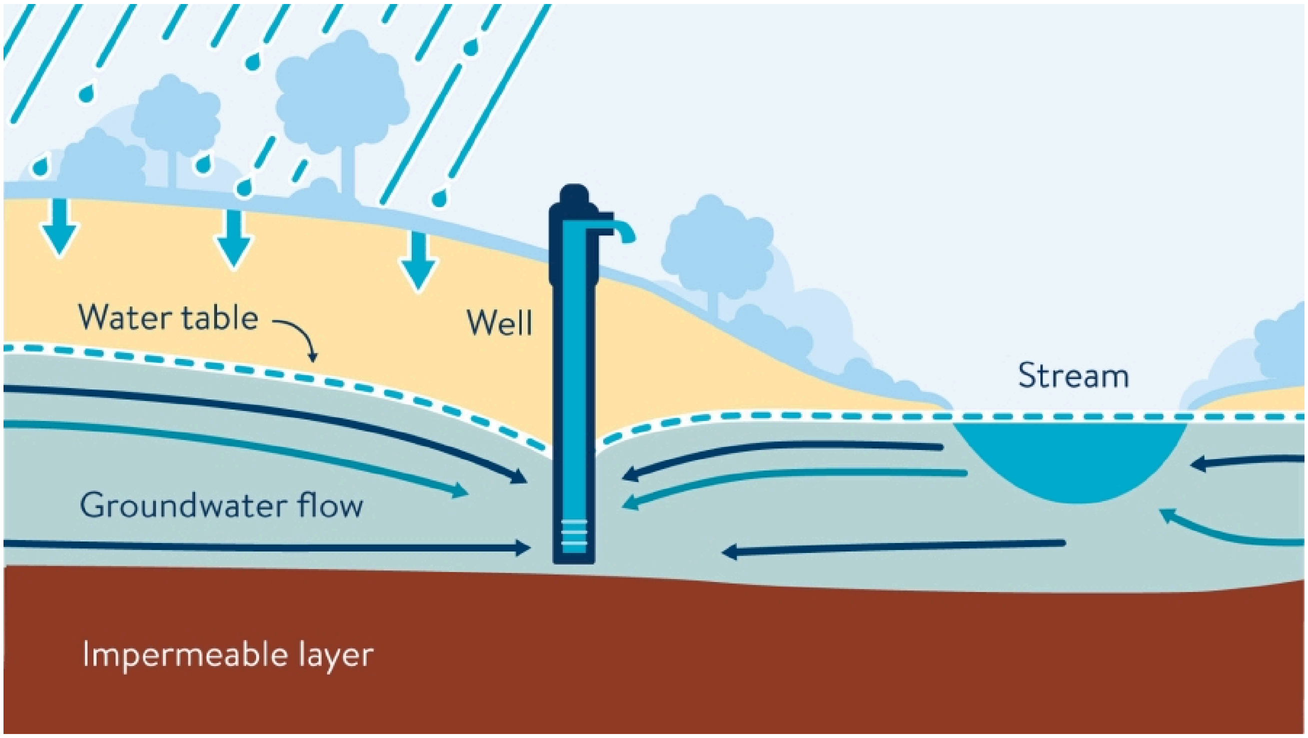
Western and southwestern Minnesota

Nitrate pollution. About 20 percent of the monitored shallow wells in the southwestern region have nitrate concentrations higher than the EPA allows for drinking water.

How groundwater affects surface water

Groundwater contamination and diminishing water levels in the ground can affect bodies of water on the surface. Groundwater feeds surface waters and helps maintain water levels during droughts. If groundwater is being used up and the water level in a stream goes down as a result, the pollutants in the stream will be concentrated, doing greater environmental damage.

The low water levels in Little Rock Creek north of St. Cloud illustrate how groundwater interacts with surface water. Heavy groundwater pumping in the area contributes to low stream flows in the summer, killing off fish. Downstream at Little Rock Lake, low water and excess nutrients cause massive summer algae blooms. The local soil and water conservation district is working with farmers on irrigation management strategies that will use less groundwater.



BEFORE THE
UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

Petition for Emergency Action Pursuant to the Safe Drinking Water Act, 42 U.S.C. § 300i, to Protect the Citizens of the Karst Region of Minnesota from Imminent and Substantial Endangerment to Public Health Caused By Nitrate Contamination of Underground Sources of Drinking Water.	EPA Docket No. ______________ April 24, 2023

Submitted on Behalf of Petitioners
Minnesota Center for Environmental Advocacy,
Environmental Working Group,
Minnesota Well Owners Organization,
Center for Food Safety,
Clean Up the River Environment,
Food & Water Watch,
Friends of the Mississippi River,
Izaak Walton League Minnesota Division
Land Stewardship Project,
Minnesota Trout Unlimited,
and
Mitchell Hamline Public Health Law Center

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I. Introduction

Petitioners respectfully petition the U.S. Environmental Protection Agency (EPA) to exercise its emergency powers established in Section 1431 of the Safe Drinking Water Act (SDWA), 42 U.S.C. § 300i, to address groundwater contamination that presents an imminent and substantial endangerment to the health of residents in southeastern Minnesota. Like many other parts of the Nation plagued by pollution from industrial agriculture, the residents in southeastern Minnesota are suffering from drinking water contamination. As detailed in this Petition, this region has an extensive and well-documented history of nitrate contamination in its underground sources of drinking water, which continues to put the health of residents at risk. The EPA must act now to address this too-long ignored health crisis and ensure clean drinking water for Minnesotans.

Southeastern Minnesota is particularly vulnerable to groundwater pollution due to its karst geography. According to the Minnesota Pollution Control Agency (MPCA):

> Southeastern Minnesota is characterized by an unusual type of geography called karst. It features rolling hills, hollows, caves, sinkholes, and dramatic bluffs and valleys. In karst landscapes, the distinction between groundwater and surface water is blurry. . . . [C]ontaminated surface water can easily become groundwater pollution, and pose a health risk to those using it for drinking.[1]

The "karst region" of southeastern Minnesota is depicted in Figure 1 below.[2]

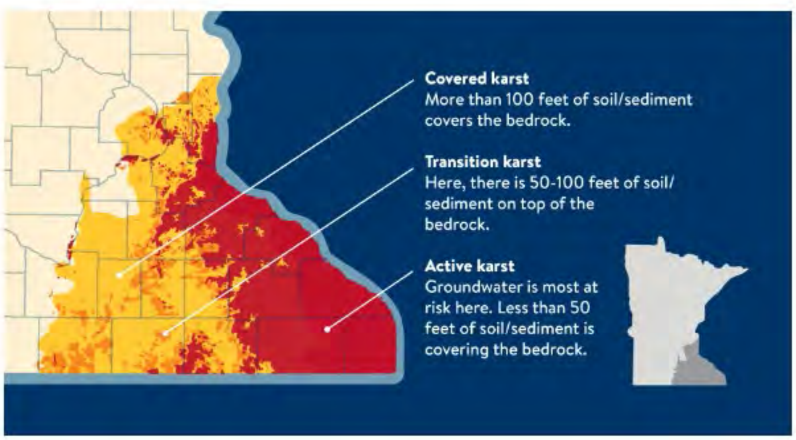

Figure 1: Minnesota's Karst Region
Based on a map created by E. Calvin Alexander, Jr., Yongli Gao, and Jeff Green

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[1] *Protecting water in karst regions*, MINN. POLLUTION CONTROL AGENCY, https://www.pca.state.mn.us/air-water-land-climate/protecting-water-in-karst-regions (last visited Apr. 13, 2023).
[2] *Id.*

The karst region[3] is a predominantly rural area of the State where many people rely on private wells, rather than public water supplies, for their drinking water.[4] All drinking water in this region—public and private—comes from groundwater aquifers. The population of the eight counties comprising this region is 380,513.[5] About 300,000 people in this area rely on community water systems while the remaining 80,000 use wells.[6] It is important to note that the populations more likely to be affected by nitrate contamination are people living in small towns, who are dependent on community water systems and private wells and who are also more likely to be of lower income.[7] The karst region of Minnesota is a community overburdened by pollution. The Administrator has called on EPA to strengthen the enforcement of cornerstone environmental statutes in these communities.[8]

This Petition is based on data that have been compiled by the Minnesota Department of Agriculture (MDA), the Minnesota Department of Health (MDH), the Minnesota Pollution Control Agency (MPCA), the Minnesota Department of Natural Resources (MDNR), Petitioner Minnesota Well Owners Organization, and Petitioner Environmental Working Group. The data demonstrate that nitrate concentrations in

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[3] The karst region does not follow county lines, but for purposes of data analysis, this Petition uses the eight counties of Dodge, Fillmore, Goodhue, Houston, Mower, Olmsted, Wabasha, and Winona as a substitute. These counties are all fully within what is considered the karst region.

[4] For information on community water systems in Minnesota that rely on groundwater *see Interactive Map: Community Water Systems*, MINN. DEP'T OF HEALTH, https://mndatamaps.web.health.state.mn.us/interactive/cwss.html (last visited Apr. 13, 2023). For further data on private wells in Minnesota, *see Minnesota Well Index*, MINN. DEP'T OF HEALTH, https://mnwellindex.web.health.state.mn.us/# (last visited Apr. 13, 2023).

[5] *See Minnesota Demographics*, CUBIT PLANNING, https://www.minnesota-demographics.com/counties_by_population (last visited Apr. 13, 2023).

[6] The population served by each community water system in the eight-county region system can be determined by clicking on MDH's water system map, *see Interactive Map: Community Water Systems*, MINN. DEP'T OF HEALTH, https://mndatamaps.web.health.state.mn.us/interactive/cwss.html (last visited Apr. 13, 2023).

[7] *Tap Water for 500,000 Minnesotans Contaminated With Elevated Levels of Nitrate*, ENV'T WORKING GRP. (Jan. 14, 2020), https://www.ewg.org/interactive-maps/2020_nitrate_in_minnesota_drinking_water_from_groundwater_sources/ [hereinafter EWG Tap Water Report]; *see also Interactive Maps: Poverty in Minnesota counties*, MINN. DEP'T OF HEALTH, https://mndatamaps.web.health.state.mn.us/interactive/poverty.html (last visited Apr. 14, 2023).

[8] Memorandum from Lawrence E. Starfield, Acting Assistant Adm'r of U.S. EPA, on Strengthening Enf't in Communities with Env't Just. Concerns to Office of Enf't and Compliance Assurance (Apr. 30, 2021), https://www.epa.gov/sites/default/files/2021-04/documents/strengtheningenforcementincommunitieswithejconcerns.pdf.

public water systems and underground sources of drinking water routinely exceed federal and state drinking water standards, putting the health of area residents at serious risk.

As explained in this Petition, the well-documented nitrate contamination of drinking water in the karst region necessitates prompt and decisive EPA emergency action under the SDWA. Elevated levels of nitrate in drinking water are known to increase the risk of a wide range of very serious health problems, including birth defects, blue-baby syndrome, various cancers, thyroid disease, and other maladies. This contamination poses an imminent and substantial threat to human health, and the problem is not getting any better.

Despite Minnesota applying for and being granted "primacy" under the SDWA, state and local officials have failed to do what is needed to correct the pervasive threat to human health. The data confirm that past voluntary measures employed by the State have been unsuccessful at reducing nitrate concentrations in crucial drinking water sources to below federal and state standards. EPA is fully empowered under the SDWA to take emergency action to protect human health in the karst region of Minnesota given present circumstances.

Because of its landscape features, groundwater quality in the karst region is largely driven by land use practices, and land use in this region is dominated by industrial row crop agriculture and feedlots. Petitioners request that EPA act to protect human health and effectuate the goals of the SDWA in the karst region of Minnesota through an investigation focused on the agricultural land uses that are most likely driving the contamination of drinking water resources. Specifically, Petitioners request that EPA issue orders, as necessary, to protect the health of people who use the drinking water, including, at a minimum, orders that require responsible contaminators to provide a free and safe alternative source of drinking water for impacted communities; orders that prohibit concentrated animal feeding operations (CAFOs) from expanding or constructing new operations until nitrate concentrations fall below unsafe levels; public notice of potential contamination events, such as manure land applications; an investigation to determine the specific entities and land use practices causing the contamination; a survey to identify public water systems, private supply wells, or ground water monitoring wells near potentially contaminated areas; monitoring of contaminants; control of the source of contaminants; and cleanup of contaminated soils endangering underground sources of drinking water. Petitioners further request that EPA seek injunctions through civil actions, as needed, to return the area's underground aquifers to a safe and drinkable condition.

II. Interests of Petitioners

Minnesota Center for Environmental Advocacy (MCEA) is a nonprofit environmental advocacy organization with offices in St. Paul and Duluth, Minnesota.

Since 1974, MCEA has defended Minnesota's natural resources, water, air and climate, and the health and welfare of Minnesotans. MCEA is driven by the principle that everyone has a right to a clean and healthy environment, and that decisions must be based on fact, science, and the law.

Environmental Working Group (EWG) is a nonprofit, nonpartisan organization that empowers people to live healthier lives in a healthier environment. For 30 years, EWG has harnessed its signature blend of research, advocacy, and unique educational tools to drive consumer choice and inspire civic action.

Minnesota Well Owners Organization (MNWOO) is a statewide nonprofit with a mission to help ensure safe drinking for Minnesota private well users who depend on groundwater for their private water systems and wells. MNWOO works with well users and partners with other non-governmental organizations, and local and state government units to build individual and community values for the protection, enhancement, and restoration of Minnesota groundwater through outreach, education, and advocacy. MNWOO's goal is to conduct free water quality screening clinics and provide professional help to connect and activate the community of well owners, land managers, water managers, and policy makers who steward Minnesota's groundwater. MNWOO seeks to remove the threats to safe drinking water on a foundation of accurate, up-to-date, and practical information that addresses the personal, community, economic, technical, legal, and policy barriers faced by private well owners seeking safe drinking water. MNWOO works to motivate private well owners and decision makers to take the individual and collective steps necessary to assure safe drinking water from all private wells for future generations.

Center for Food Safety (CFS) is a nonprofit environmental advocacy organization that aims to empower people and protect the environment from the harmful effects of industrial agriculture, including groundwater contamination from the concentration of industrial animal operations and their waste. CFS represents over a million members and supporters across the country, including over 9,000 members in Minnesota. CFS uses education, science-based advocacy, and litigation to address the negative environmental and public health effects of industrial agriculture.

Clean Up the River Environment (CURE) is a rural Minnesota nonprofit organization headquartered in the Minnesota River valley. CURE's mission is to protect and restore resilient rural landscapes and build vibrant, just, and equitable rural communities. CURE embodies three core practices: (1) awakening people's bonds with the natural world around them; (2) inclusively, strategically, and dialectically exploring issues and actions; and (3) systematically building communities of change at critical intersections of ecological and social wellbeing. Among CURE's values and guiding principles are that the capacity of communities to flourish is directly connected to the condition of the landscapes that embrace them; a moral responsibility to future generations to be good stewards of the ecosystems in which they live; and the human use of natural resources can be regenerative and a sustainable force. CURE, with its rural roots, is aware that the Dakota and Ojibwe Nations and other rural communities, already culturally, socially, and politically marginalized, are often most impacted by climate change, clean water scarcity, and environmental degradation. While local control is important to CURE, it is equally important that there is accountability to all Minnesotans and to future generations. Because rural communities are frontline communities when it comes to pollution from industrial agriculture, CURE requests that EPA exercise its broad emergency powers, per the SDWA, to address groundwater contamination in southeastern Minnesota. Too often industrial agriculture is given a pass on protections for our land and water, putting profits over people. CURE asks EPA to step in and be a voice for those communities impacted by groundwater contamination.

Food & Water Watch (FWW) is a national, nonprofit membership organization that mobilizes regular people to build political power to move bold and uncompromised solutions to the most pressing food, water, and climate problems of our time. FWW uses grassroots organizing, media outreach, public education, research, policy analysis, and litigation to protect people's health, communities, and democracy from the growing destructive power of the most powerful economic interests. FWW has long advocated for stronger regulation of factory farm pollution and industrial agribusiness to protect farmers, rural communities, and the environment.

Friends of the Mississippi River (FMR) engages people to protect, restore and enhance the Mississippi River and its watershed in the Twin Cities region. FMR's water quality and drinking water protection work focuses on addressing agricultural contamination of surface water and groundwater with a goal of ensuring all Minnesotans have access to clean, safe, and healthy waters.

For over 100 years, the Izaak Walton League has fought for clean air and water, healthy fish and wildlife habitat, and conserving special places for future generations. It was the first conservation organization with a mass membership. Today, the League plays a unique role in supporting citizens locally and shaping conservation policy nationwide. The League is a grass roots member organization that has led efforts for clean water legislation achieving initial success with the passage of federal water pollution acts in 1948, 1956 and finally the Clean Water Act of 1972. The League continues to advocate for preserving wetlands, protecting wilderness, and promoting soil and water conservation. Its Save Our Streams (SOS) program involves activists in all fifty states in monitoring water quality. The Minnesota Division of the Izaak Walton League of America is composed of 16 chapters located throughout the state of Minnesota. The League's broader mission is to conserve, restore, and promote the sustainable use and enjoyment of our natural resources, including soil, air, woods, waters, and wildlife. More specifically in regard to groundwater, by a resolution passed at the 1988 Annual Meeting, the Division went on record pointing out the need for better protection and management of the state's groundwater. While some protections have been put in place at the state level, it is clear that these have been inadequate. Greater federal protections are urgently needed.

Land Stewardship Project (LSP) is a private, nonprofit organization founded in 1982 to foster an ethic of stewardship for farmland, to promote sustainable agriculture, and to develop healthy communities. LSP is dedicated to creating transformational change in our food and farming system. LSP's work has a broad and deep impact, from new farmer training and local organizing to federal policy and community-based food systems development. At the core of all of LSP's work are the values of stewardship, justice, and democracy.

Minnesota Trout Unlimited (MNTU) is a nonprofit, nonpartisan conservation organization working to protect, restore, and sustain the watersheds and groundwater sources that support coldwater fisheries. For more than 60 years our members have advocated for clean water, both for recreational benefits and drinking. Minnesota trout streams are protected as Class 1 drinking water sources due to their close connection to groundwater. Nitrate contamination of southeast Minnesota groundwater and trout streams not only harms humans, but also the aquatic organisms on which these ecosystems depend. MNTU's several thousand Minnesota members regularly fish southeast streams and drink the water drawn from area aquifers.

Public Health Law Center (PHLC) is a nonprofit law and policy organization working to advance equitable public health policies through the power of law. For over 20 years, PHLC has fought to regulate and eliminate commercial tobacco, promote healthy food, support physical activity, and improve environmental health as a means of reducing chronic disease. PHLC partners with Tribal health leaders, federal agencies, health advocacy organizations, state and local governments, and many others to combat systems of institutional racism and create healthier communities across the country.

III. Legal Background

A. Safe Drinking Water Act

Congress enacted the SDWA as a powerful tool for protecting drinking water resources throughout the United States. Under the Act, EPA may delegate duties to state authorities to develop policies, regulations, and programs to ensure access to safe drinking water. On the federal level, the SDWA "requires EPA to protect the public from . . . drinking water contaminants."[9]

[9] *City of Portland v. Env't Prot. Agency*, 507 F.3d 706, 709 (D.C. Cir. 2007).

States may apply for, and EPA may delegate, "primacy" to states, which shifts significant authority and responsibility to state officials to implement the SDWA.[10] To assume primacy, the state is supposed to adopt regulations at least as stringent as EPA's national requirements, develop adequate procedures for enforcement and levying penalties, conduct inventories of water systems, maintain records and compliance data, and develop a plan for providing safe drinking water under emergency conditions.[11] While a state granted primacy has responsibility to implement the SDWA's provisions in that state, EPA retains emergency powers under Section 1431 of the SDWA to take actions necessary to abate imminent and substantial endangerment to the health of persons caused by drinking water contamination when state officials have failed to effectively do so on their own.

B. EPA's Emergency Powers

For EPA to exercise its Section 1431 authority, two conditions must be met. First, EPA must have received "information that a contaminant which is present in or likely to enter a public water system or an underground source of drinking water . . . may present an imminent and substantial endangerment to the health of persons."[12] Second, EPA must have received information that "appropriate State and local authorities have not acted to protect the health of such persons" in a timely and effective manner.[13]

1. Contaminant

The SDWA defines a contaminant as "any physical, chemical, biological, or radiological substance or matter in water."[14] While this broad definition does not require a substance to be regulated under the Act in order to be classified as a "contaminant," nitrate is listed as a contaminant with an established maximum contaminate level (MCL) of 10 mg/L.[15] An MCL is the "maximum permissible level of a contaminant in water which is delivered to any user of a public water system."[16] MCLs are promulgated after a determination by EPA based on the best available, peer-reviewed science and data that the regulation of the contaminant will reduce a threat to public health.[17] Establishing

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[10] 42 U.S.C. § 300g-2; 40 C.F.R. §§ 142.10–142.19 (primacy enforcement responsibility).
[11] ELENA H. HUMPHREYS & MARY TIEMANN, CONG. RES. SERV., RL31243, SAFE DRINKING WATER ACT (SDWA): A SUMMARY OF THE ACT & ITS MAJOR REQUIREMENTS 7 (2021), https://sgp.fas.org/crs/misc/RL31243.pdf.
[12] 42 U.S.C. § 300i; *see also* U.S. ENV'T PROT. AGENCY, UPDATED GUIDANCE ON EMERGENCY AUTHORITY UNDER SECTION 1431 OF THE SDWA 8 (2018) [hereinafter EMERGENCY AUTHORITY GUIDANCE].
[13] 42 U.S.C. § 300i; *see also* EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at 12-13.
[14] 42 U.S.C. § 300f(6).
[15] 40 C.F.R. § 141.62(b).
[16] 42 U.S.C. § 300f(3).
[17] 42 U.S.C. §§ 300g-l(b)(1)(A), (b)(3)(A).

nationwide, health-based MCLs is central to EPA's role in protecting drinking water under the SDWA.[18]

The MCL for nitrate was set at 10 mg/L to protect against blue-baby syndrome; however, recent studies have shown that even lower levels of nitrate can cause other health effects, including cancer and reproductive harm.[19] For example, recent studies have found statistically significant increased risks of colorectal cancer at drinking water levels far below the current MCL of 10 mg/L.[20]

2. Imminent & Substantial Endangerment

An endangerment from a contaminant is "imminent" if conditions that give rise to it are present, even if the actual harm may not be realized for years.[21] Courts have established that an "imminent hazard" may be declared at any point in a chain of events that may ultimately result in harm to the public.[22] Information presented to EPA need not demonstrate that residents are actually drinking contaminated water and becoming ill to warrant EPA exercising its Section 1431 emergency authority.[23] In other words, an actual injury need not have occurred for EPA to act, and to wait for such actual injury to befall the public would be counter to the precautionary intent behind the SDWA. Thus, while the threat or risk of harm must be "imminent" for EPA to act, actual and documented harm itself need not be.[24] While endangerments are readily determined to be imminent where MCL violations expose sensitive populations to a contaminant, contaminants that lead to chronic health effects may also cause "imminent endangerment."[25] In such cases, it is appropriate to consider the length of time a population has been or could be exposed to a contaminant.[26]

An endangerment is "substantial" "if there is a reasonable cause for concern that someone may be exposed to a risk of harm."[27] For instance, Congress has deemed an

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[18] 42 U.S.C. § 300g-1(b)(4)(B).

[19] *See, e.g.,* Mary. H. Ward et al., *Drinking Water Nitrate and Human Health: An Updated Review*, 15 INT'L J. ENV'T RSCH. & PUB. HEALTH 1557 (2018); Alexis Temkin et al., *Exposure-Based Assessment and Economic Valuation of Adverse Birth Outcomes and Cancer Risk Due to Nitrate in United States Drinking Water*, 176 ENV'T RSCH. 108442 (2019).

[20] *See, e.g.,* Jorg Schullehner et al., *Nitrate in Drinking Water and Colorectal Cancer Risk: A Nationwide Population-Based Cohort Study*, 143 INT'L J. CANCER 73 (2018).

[21] EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at 8 (citing *United States v. Conservation Chem. Co.*, 619 F. Supp. 162, 193-94 (W.D. Mo. 1985)).

[22] *Id.* n.15 (citing cases).

[23] *See Trinity Am. Corp. v. Env't Prot. Agency*, 150 F.3d 389, 399 (4th Cir. 1998).

[24] EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at 8.

[25] *Id.*

[26] *Id.*

[27] *Id.* at 11.

endangerment sufficiently substantial where a substantial likelihood exists that contaminants capable of causing adverse health effects will be ingested by consumers if preventative action is not taken.[28] As with imminence, EPA has made clear that actual reports of human illness resulting from contaminated drinking water are not necessary to establish substantial endangerment.[29]

C. Minnesota's Authority

Minnesota has several state agencies with jurisdiction over the quality of underground sources of drinking water: MDH, MDA, and MPCA are the primary ones. The graphic below shows the differing roles of these agencies.[30]

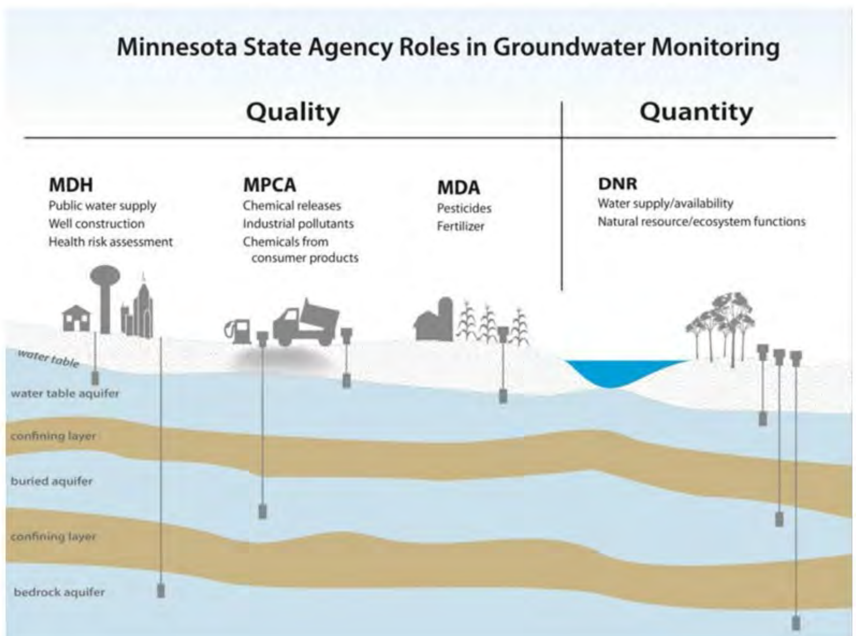

Figure 2: Agency Roles in Groundwater

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[28] *See* H.R. REP. NO. 93-1185, at 35-36 (1974).

[29] *See* EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at 11 (citing *United States v. North Adams*, 777 F. Supp. 61, 84 (D. Mass. 1991)).

[30] SHARON KROENING & SOPHIA VAUGHAN, MINN. POLLUTION CONTROL AGENCY, CONDITIONS OF MINNESOTA'S GROUNDWATER QUALITY 2013-2017, 4 (2019), https://www.pca.state.mn.us/sites/default/files/wq-am1-10.pdf [hereinafter MPCA GROUNDWATER QUALITY 2013-2017]. The graphic also depicts the MDNR, which controls water appropriation and has a role in agricultural drainage projects that affect public waters. MDNR also conducts some groundwater monitoring as part of is County Geologic Atlas program.

The MDH administers the Minnesota Well Code for the construction of new wells and borings[31] and Minnesota's SDWA.[32] EPA granted Minnesota primacy under the federal SDWA in 1976.[33] Although the SDWA allows states to set higher standards than the federal minimum, Minnesota state law sets the drinking water quality standard for nitrate at the same level as the federal standard: 10 mg/L.[34] Public water systems with nitrate levels over 10 mg/L must notify people who receive water from them.[35]

The MPCA's authority extends to discharges from point sources under its water pollution control laws.[36] Point sources include animal feeding operations, which, as discussed below, are a significant contributor of nitrate pollution to groundwater in the karst region. The MPCA regulates animal feeding operations with more than 1,000 animal units through the issuance of National Pollution Discharge Elimination System (NPDES) permits,[37] but smaller farms are unregulated. Finally, the MDA has statutory authority under the Minnesota Groundwater Protection Rule to regulate the use of pesticides and commercial fertilizer.[38]

D. EPA's Authority in Minnesota

Despite Minnesota's primacy under the SDWA, EPA retains emergency powers to abate present or likely contamination of public water systems (PWS) or underground sources of drinking water (USDW) when such contamination poses an imminent and substantial threat to human health and the state "ha[s] not acted to protect the health of [endangered] persons."[39]

EPA's Section 1431 authority extends to contaminated USDW and PWS that pose a threat to human health,[40] including sources that supply private wells.[41] EPA defines USDW as an aquifer or part of an aquifer "(1) [w]hich supplies any public water systems; or (2) which contains a sufficient quantity of ground water to supply a public water system; and (i) currently supplies drinking water for human consumption."[42] PWS are

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[31] MINN. R. 4725.0500–4725.7605.
[32] MINN. STAT. §§ 144.381–144.387.
[33] MINN. DEP'T OF HEALTH, MINNESOTA DRINKING WATER ANNUAL REPORT FOR 2021 2 (2022), https://www.health.state.mn.us/communities/environment/water/docs/report21.pdf.
[34] MINN. R. 4720.0350 (adopting national standards by reference).
[35] MINN. STAT. § 144.385.
[36] MINN. STAT. § 115.03.
[37] MINN. R. 7020.2003, subp. 2(B).
[38] MINN. STAT. § 103H.275; MINN. R. 1573.0010–1573.0090.
[39] 42 U.S.C. § 300i(a).
[40] *Id.*
[41] EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at 7-8.
[42] 40 C.F.R. § 144.3.

aquifers that provide water for human consumption and "ha[ve] at least fifteen service connections or regularly serve[] at least twenty-five individuals."[43] The drinking water for the hundreds of thousands of residents of the karst region of Minnesota comes from either private or community wells that rely on groundwater. The underground aquifers that supply these wells therefore qualify as USDW and PWS within the purview of the SDWA.

To abate endangerment to human health that arises despite a state's efforts to curtail it, Congress authorized EPA to, among other things, issue "such orders as may be necessary to protect the health of persons who are or may be users of" the affected drinking water supplies and to commence civil enforcement actions against entities causing threats to public health by contaminating drinking water supplies.[44] Petitioners ask EPA to use that authority here.

IV. Drinking Water Contamination in the Karst Region Constitutes an Endangerment under the SDWA and Necessitates Emergency Action by EPA

Nitrate contamination in Minnesota's karst region is a widespread issue that has stubbornly persisted through decades as state officials continuously fail to effectively address the problem. "Nitrate contamination of surface water and groundwater is a long-standing issue in the region. Impacts to municipal and private drinking water supplies by nitrate are widespread and well-documented."[45] According to MPCA, "[t]rends from the past 10, 20, and 40 years show that statewide . . . nitrate concentrations have generally been increasing."[46] Figure 3 is a MPCA graphic which shows that there are no areas of the state where nitrate trends in surface water have decreased between 2008 and 2017.[47] The main contributors to this problem are large-scale animal agriculture facilities and industrial row-crop agriculture which dominate land use within the area and that are not effectively addressed by existing regulations and policies promoting voluntary actions.

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[43] 42 U.S.C. § 300f(4)(A).

[44] EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at Attach. 2.

[45] ANTHONY C. RUNKEL ET AL., GEOLOGIC CONTROLS ON GROUNDWATER AND SURFACE WATER FLOW IN SOUTHEASTERN MINNESOTA AND ITS IMPACT ON NITRATE CONCENTRATIONS IN STREAMS, MINN. GEOLOGIC SURV., 4 (2013) [hereinafter RUNKEL 2013].

[46] DAVE WALL ET AL., MINN. POLLUTION CONTROL AGENCY, 5-YEAR PROGRESS REPORT ON MINNESOTA'S NUTRIENT REDUCTION STRATEGY 17 (2020), https://www.lrl.mn.gov/docs/2021/other/210420.pdf [hereinafter 5-YEAR PROGRESS REPORT].

[47] *Id.*

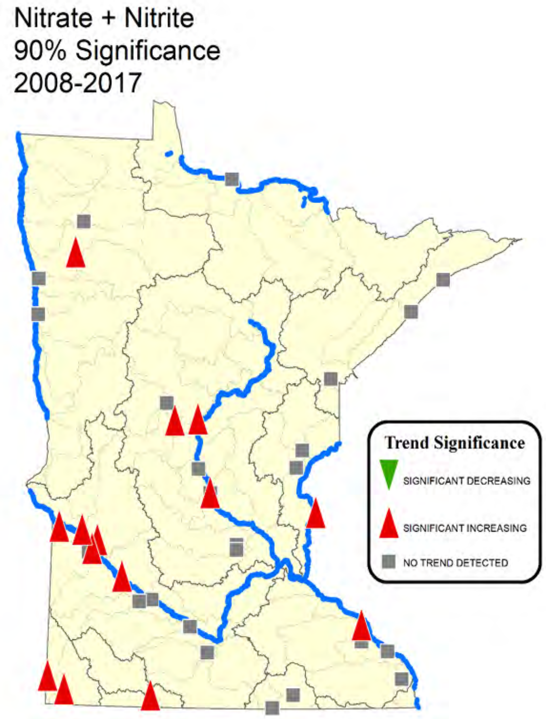

Figure 3: 5-year Progress on Nitrate

Emergency action by EPA is necessary to address the dangerous levels of nitrate in the karst region because the contamination poses an imminent and substantial risk to the health of more than 380,000 residents who rely on groundwater, and because Minnesota officials have failed to improve drinking water quality, despite knowing about the problem, for over 40 years.[48]

A. The Karst Region is Particularly Susceptible to Nitrate Pollution

Groundwater in the karst region is vulnerable to contamination because of the fluid interaction between groundwater and surface water. The rapid movement of water in and out of the ground in this region leaves a blurry distinction between groundwater and surface water that is compounded by Minnesota's multi-agency approach to drinking water policies, regulation, and funding. Specific karst features such as stream sinks and sinkholes that inject water into the ground and the springs that discharge groundwater to the surface are depicted in Figure 4.[49] "[N]ot only does karst aquifer groundwater flow rapidly (flows have been measured in miles per day versus the inches, or feet, per year common to sandstones), but contaminants in the groundwater are not

[48] 5-YEAR PROGRESS REPORT, *supra* note 46, at 17.
[49] RUNKEL 2013, *supra* note 45, at Fig. 3.

readily filtered out. As a result, contaminants can reach domestic wells located miles from the source of contamination."[50]

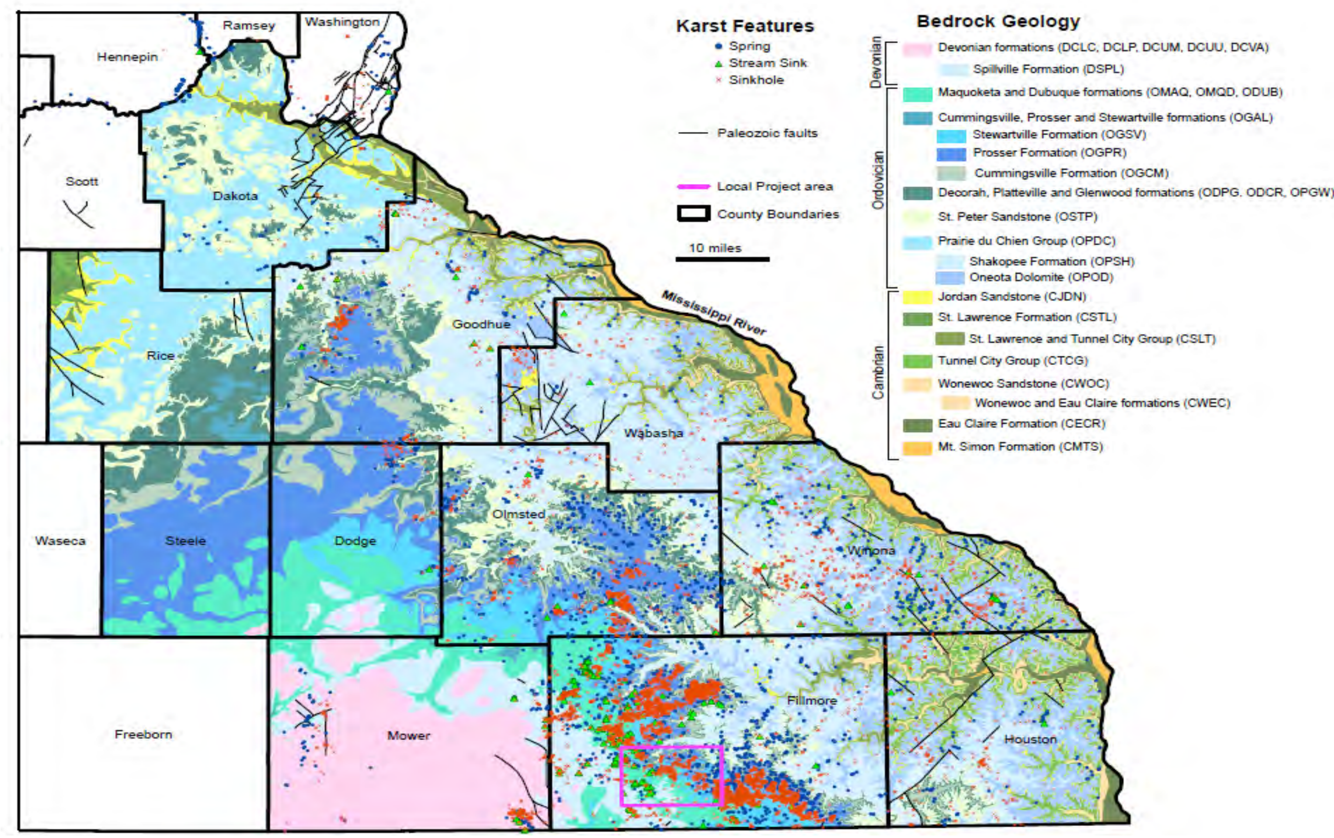

Figure 4: Karst Features

Nitrate pollution is particularly troublesome because nitrate is mobile in groundwater.[51] Nitrate mobility in karst regions can be largely determined by rainfall frequency and intensity.

Recent research indicates that up to 80% of nitrate loading in karst regions can be traced to fertilizers that are quickly flushed from soils into the karst and groundwater

[50] JEFFREY ST. ORES ET AL., GROUNDWATER POLLUTION PREVENTION IN SOUTHEAST MINNESOTA'S KARST REGION, 465 UNIV. OF MINN. EXTENSION BULL. 6 (1982), https://conservancy.umn.edu/bitstream/handle/11299/169069/mn_2000_eb_465.pdf?sequence=1 [hereinafter ORES 1982].

[51] MINN. POLLUTION CONTROL AGENCY, EFFECTS OF LIQUID MANURE STORAGE SYSTEM ON GROUNDWATER QUALITY 3 (2001), https://www.pca.state.mn.us/sites/default/files/rpt-liquidmanurestorage.pdf.

systems during rain events.[52] Water carries the excess nitrogen from fertilizers on the surface through the soil column and into the fractured karst bedrock, where oxygenated conditions facilitate conversion of nitrogen to nitrate.[53] Combining nitrogen intensive land uses with the karst region's heightened vulnerability to nitrate contamination is a major hazard.

As a result, "[g]roundwater in uppermost bedrock units, especially on the karstic plateaus that dominate the landscape of southeastern Minnesota, is typically nitrate-enriched, with concentrations commonly between 5-15 ppm."[54] Rural communities are particularly at risk since private wells are more likely to draw from shallow aquifers than public water systems, which can pull water from deeper wells and multiple sources.[55]

Minnesota officials have been aware of the vulnerability of this region for at least 80 years. "S.P. Kingston, a former Minnesota health official, noted in 1943 that the regional groundwater system in southeast Minnesota is particularly vulnerable to contamination from many sources."[56] And nitrate was identified as one of the contaminants of concern as early as 1982: "Many shallow wells in southeast Minnesota contain coliform bacteria and high nitrate levels—both indicators of possible contamination."[57] The evidence of nitrate contamination in the groundwater of this region is robust.

B. The Karst Region Has a Documented History of Nitrate Contamination

The karst region has an extensive history with nitrate contamination in groundwater aquifers. Although nitrate is a naturally occurring substance, the presence of nitrate in groundwater at concentrations above 3 parts per million or milligrams per liter is not natural and indicates an anthropogenic source of the nitrate.[58]

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[52] Fu-Jun Yue et al., *Rainfall and Conduit Drainage Combine to Accelerate Nitrate Loss from a Karst Agroecosystem: Insights from a Stable Isotope Tracing and High-Frequency Nitrate Sensing*, 186 WATER RSCH. 116388 (2020), https://doi.org/10.1016/j.watres.2020.116388.

[53] PHILIP MONSON, MINN. POLLUTION CONTROL AGENCY, AQUATIC LIFE WATER QUALITY STANDARDS DRAFT TECHNICAL SUPPORT DOCUMENT FOR NITRATE 1 (2022), https://www.pca.state.mn.us/sites/default/files/wq-s6-13.pdf.

[54] RUNKEL 2013, *supra* note 45, at 59.

[55] *Learn About Private Water Wells*, ENV'T PROT. AGENCY (Mar. 1, 2023), https://www.epa.gov/privatewells/learn-about-private-water-wells.

[56] ORES 1982, *supra* note 50, at 3.

[57] *Id.*

[58] *Nitrate in Drinking Water*, MINN. DEP'T OF HEALTH (DEC. 8, 2022), https://www.health.state.mn.us/communities/environment/water/contaminants/nitrate.html.

Regular sampling of wells to detect nitrate began over 30 years ago. Fifty-five wells in Winona County were first sampled in 1990 and 1991.[59] Twenty-five of the well samples were taken from the shallower Prairie du Chien aquifer and 30 were from the deeper Jordan aquifer. "Nitrate concentrations exceeded the 10 mg/l drinking water standard in 48 percent of Prairie du Chien wells and 3.2 percent of Jordan wells."[60] Fifteen to thirty years later, nothing had improved: testing data from wells sampled between 2005 to 2017 revealed that 49% of wells in agricultural areas of the state, installed near the water table, exceeded the MCL for nitrate.[61]

Petitioners present a compilation of data in this Petition that shows nitrate contamination in private wells in the karst region. The data were compiled by Petitioners EWG and MNWOO. In 2020, EWG used data from the Township Testing Program[62] conducted by MDA, a Volunteer Nitrate Monitoring Network,[63] and new well tests required by MDH since the Well Code was adopted in 1975.[64] EWG used the data to create an interactive map showing nitrate contamination by township.[65] The Township Testing Program sampled and analyzed over 32,000 private wells between 2017 and 2020. The Volunteer Nitrate Monitoring Network in the karst region began in 2008 with a network of 675 private drinking water wells. "Between February 2008 and August 2018, 13 sampling events occurred representing 5,421 samples."[66] And MDH provided EWG with location data and test results for each of the 45,598 wells sampled between 2009 and 2018.[67] Finally, MNWOO hosts well testing clinics that allow homeowners to test their

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[59] David B. Wall & Charles P. Regan, *Water Quality and Sensitivity of the Prairie du Chien-Jordan Aquifer in West-Central Winona County*, MINN. POLLUTION CONTROL AGENCY, ES1 (1991).
[60] *Id.*
[61] MPCA GROUNDWATER QUALITY 2013-2017, *supra* note 30, at 2, 15.
[62] MINN. DEP'T AGRIC., TOWNSHIP TESTING PROGRAM UPDATE - MAY 2022 (2022), https://www.mda.state.mn.us/sites/default/files/docs/2022-05/ttpupdate2022_05.pdf (hereinafter TOWNSHIP TESTING UPDATE 2022).
[63] MINN. DEP'T OF HEALTH, VOLUNTEER NITRATE MONITORING NETWORK: METHODS AND RESULTS (2012), https://www.health.state.mn.us/communities/environment/water/docs/swp/no3methods.pdf.
[64] MINN. R. 4725.0500–4725.7605.
[65] *Interactive Map: Nitrate in Minnesota Private Drinking Water from Groundwater Sources (2009-2018)*, ENV'T WORKING GRP., https://www.ewg.org/interactive-maps/2020_nitrate_in_minnesota_private_drinking_water_from_groundwater_sources/map/ (last visited Apr. 17, 2023).
[66] KIM KAISER ET AL., MINN. DEP'T OF AGRIC., NITRATE RESULTS AND TRENDS IN PRIVATE WELL MONITORING NETWORKS 2008-2018 2 (2019), https://wrl.mnpals.net/islandora/object/WRLrepository%3A3395/datastream/PDF/view.
[67] EWG Tap Water Report, *supra* note 7, at Methodology.

well water for nitrates and chlorides at no cost. MNWOO provided data from 119 different wells, from at least 24 townships from five counties in the karst region. To date, these data points do not appear in any other public record. The karst-region-specific data from these combined sources are depicted in Figure 5.

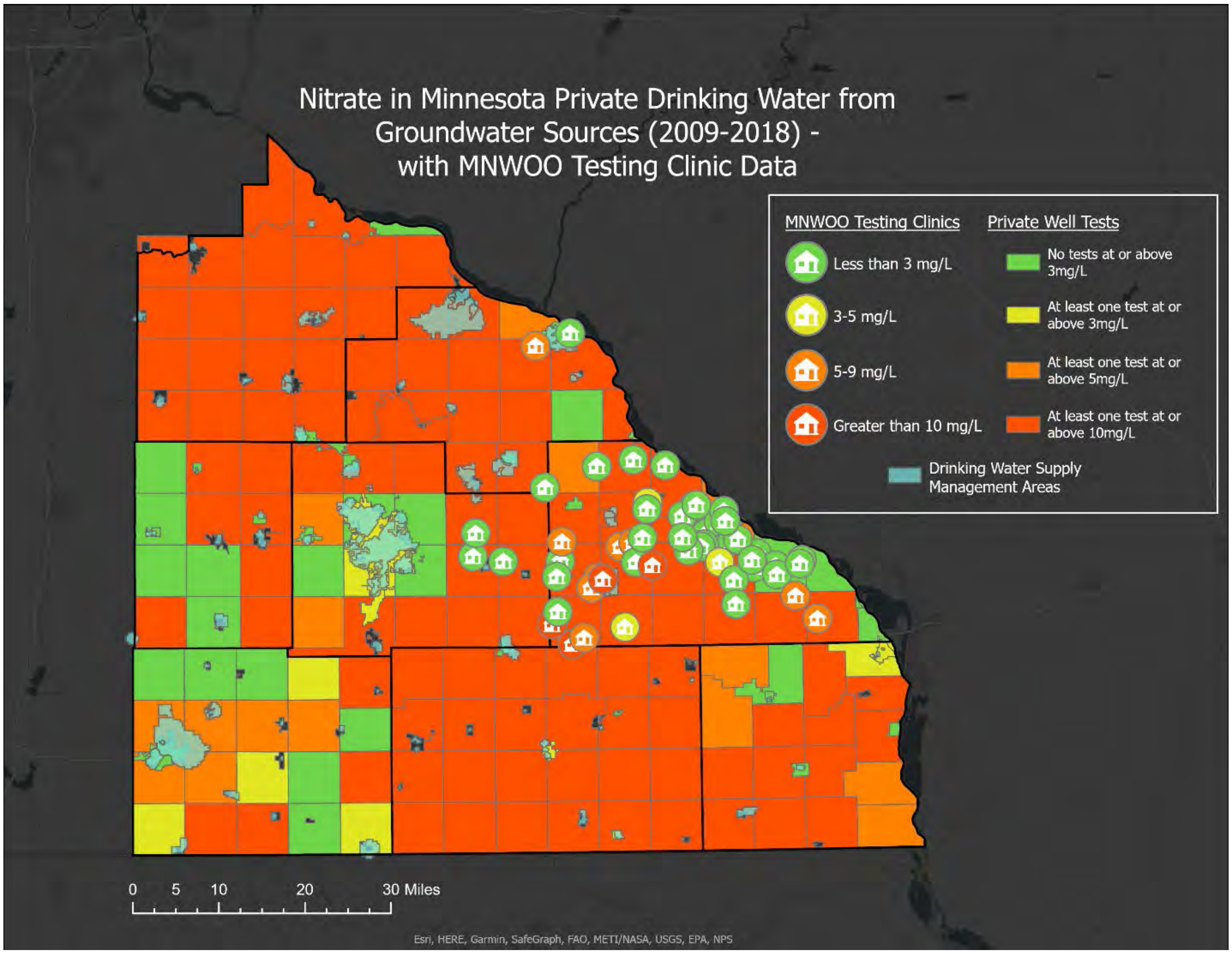

Figure 5: Private Well Contamination
Data from Township Testing Program, Southeast Volunteer Monitoring Network, MDH Well Index, and MNWOO clinic

Approximately 9% of the wells tested during the initial round of the Township Testing Program were found to have samples that exceeded the MCL for nitrate of 10mg/l. The multiple rounds of sampling and analysis also found a maximum nitrate concentration of 69.8 mg/L. The percentage of wells tested between 2008 and 2018 in the Volunteer Nitrate Monitoring Network (VNMN) above 10 mg/l ranged from a low of 7.5% in 2012 to a high of 14.6% in 2008. More recent data from the VNMN show that (among continuing participants) nitrate contamination continues: In 2019, 9% of wells tested above 10 mg/l, in 2020 it was 9.4% and in 2021 it was 8.5%.[68] The MNWOO clinic conducted in the karst region in February 2023 showed a 6% rate of nitrate contamination above 10 mg/L.

Figure 5 also depicts the location of the wells in comparison to the Drinking Water Supply Management Areas (DWSMAs). DWSMAs are defined geographic areas around public water supply wells that represent a 10-year travel time for water to reach the well. These areas are used by MDH and local communities in developing Well Head Protection Areas and are the geographic limitation for MDA's ability to protect groundwater under the Groundwater Protection Rule from commercial fertilizers and pesticides. As figure 5 demonstrates, many of the private wells in this region fall outside of a protected DWSMA. EPA needs to step in to afford private well owners protection against nitrate contamination.

It is also important to note that despite the additional protection available to protect PWS, many community water supplies with 25 or more connections to a well and many transient community water supplies like churches, campgrounds, and businesses in the area, are also affected by nitrate contamination. Petitioner EWG has also compiled Minnesota well testing data into an interactive map for public water systems,[69] and presents a karst-specific version of that map in Figure 6.

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68 *Southeast Minnesota Volunteer Monitoring Network*, MINN. DEP'T OF AGRIC., https://www.mda.state.mn.us/southeast-minnesota-volunteer-nitrate-monitoring-network (last visited Apr. 17, 2023).

69 *Interactive Map: Nitrate in Minnesota Public Drinking Water from Groundwater Sources (2009-2018)*, ENV'T WORKING GRP., https://www.ewg.org/interactive-maps/2020_nitrate_in_minnesota_public_drinking_water_from_groundwater_sources/map/ (last visited Apr. 17, 2023).

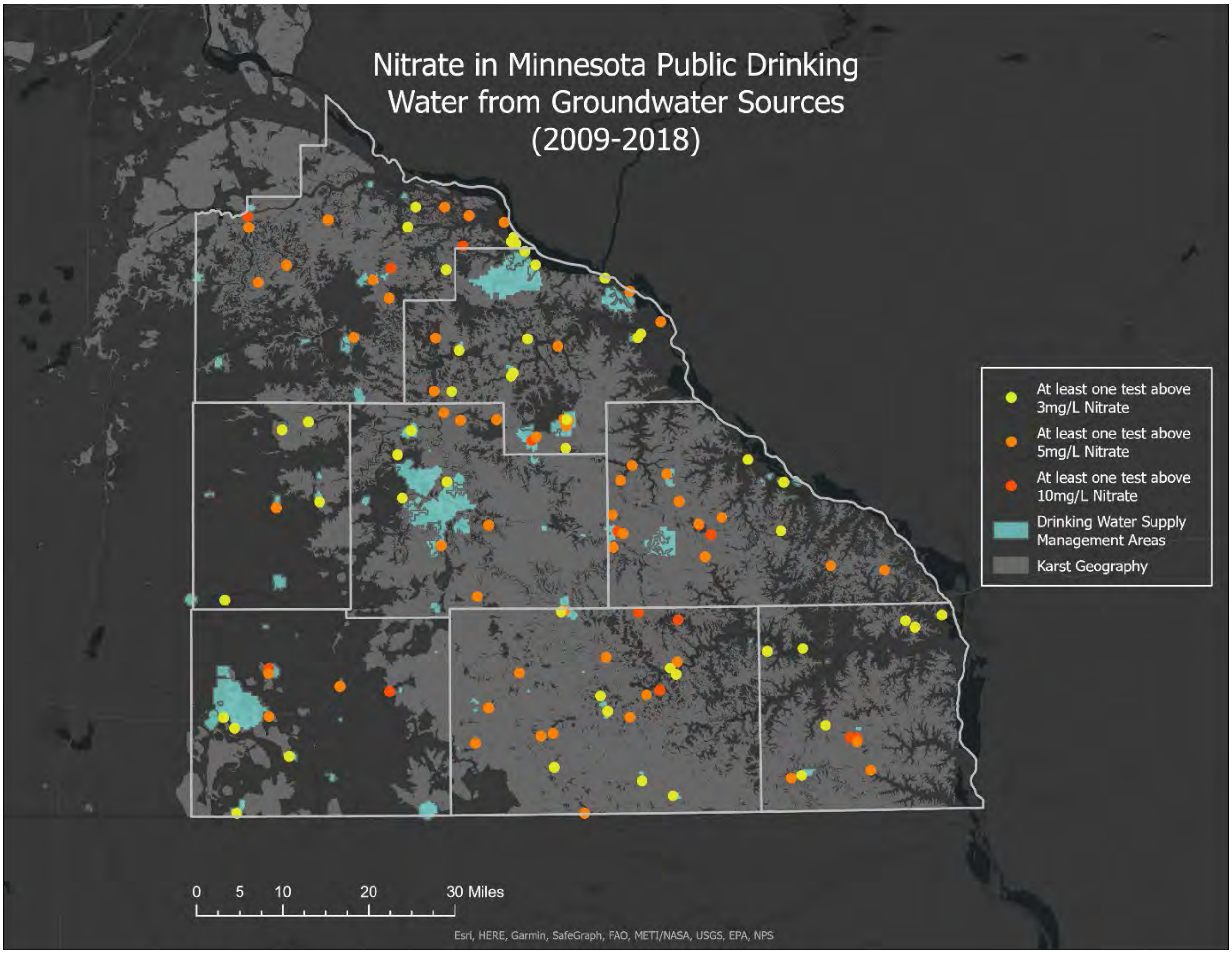

Figure 6: Public Drinking Water Contamination

In its 2020 analysis, EWG determined that groundwater-derived drinking water for an estimated 150,000 Minnesotans is contaminated with nitrate at levels over the legal limit. For 4,178 Minnesotans, the level is more than double the legal limit.[70] Cities in the karst region have long struggled with high nitrate concentrations in their drinking water. For example, the city of Lewiston has dug multiple deeper wells to try to eradicate nitrate from the city's water at a cost of approximately $1 million per well.[71] Had the city pursued a treatment system, the cost would have risen to $3.1 million, and doubled water rates for residents.[72]

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70 EWG Tap Water Report, *supra* note 7.
71 Elizabeth Baier, *Even in Region with Abundant Water, Residents Turn to Bottles and Try to Conserve*, MPR NEWS (Mar. 20, 2014), https://www.mprnews.org/story/2014/03/20/ground-level-beneath-the-surface-southeast-minnesota.
72 *Id.*

As another example, the city of Utica has two city wells, but as shown in the graph below, one well has been exceeding the 10 mg/L MCL since 2003 and is now for emergency use only. The other well, drilled in the late 1970s, began with a nitrate concentration of 3.9 mg/L, but that concentration has been steadily increasing and was as high as 8.6 mg/l in 2019.

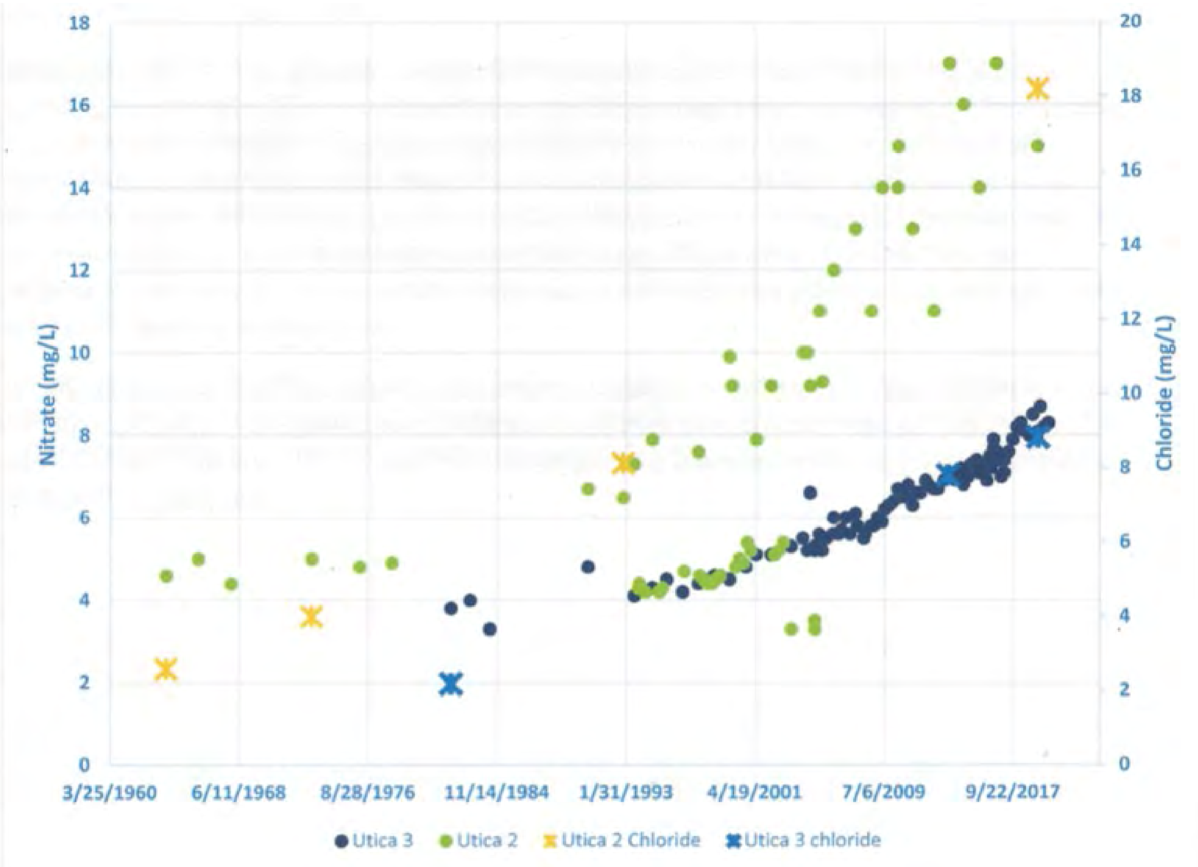

Figure 7: Utica City Well Contamination
Data from Minnesota Geological Survey

C. Under-Regulated Animal Feeding Operations and Industrial Row Crop Agriculture Are Dominant Land Use Activities and the Predominant Causes of Nitrate Contamination in the Karst Region

Most nitrate contamination in the karst region is caused by harmful agricultural practices on groundwater recharge areas that are not sufficiently addressed by Minnesota regulators. Despite evidence of adverse impacts on groundwater and public health caused by manure storage, the excessive or poorly timed application of manure, and animal feeding operations under MPCA, industrial row-crop agriculture under MDA, or the wellhead protections under MDH, Minnesota has had inadequate state and local regulation for decades, resulting in a public health crisis that requires emergency action from EPA. The root cause of this pollution is public policy that makes polluting actions cheaper and easier than sustainable practices. The vast majority of farmers care deeply about stewardship of the land, but our policies do not reflect that same stewardship.

1. Animal Agriculture

Within the boundaries of Houston, Fillmore, Mower, Dodge, Olmsted, Wabasha, Winona, and Goodhue counties, there are currently approximately 3,170 animal feedlot operations that are required to register with MPCA's Feedlot program, with more added every year.[73] In addition, as depicted in the map below, many more feedlots are located in this area that fall below the number of animal units that require a permit or registration.

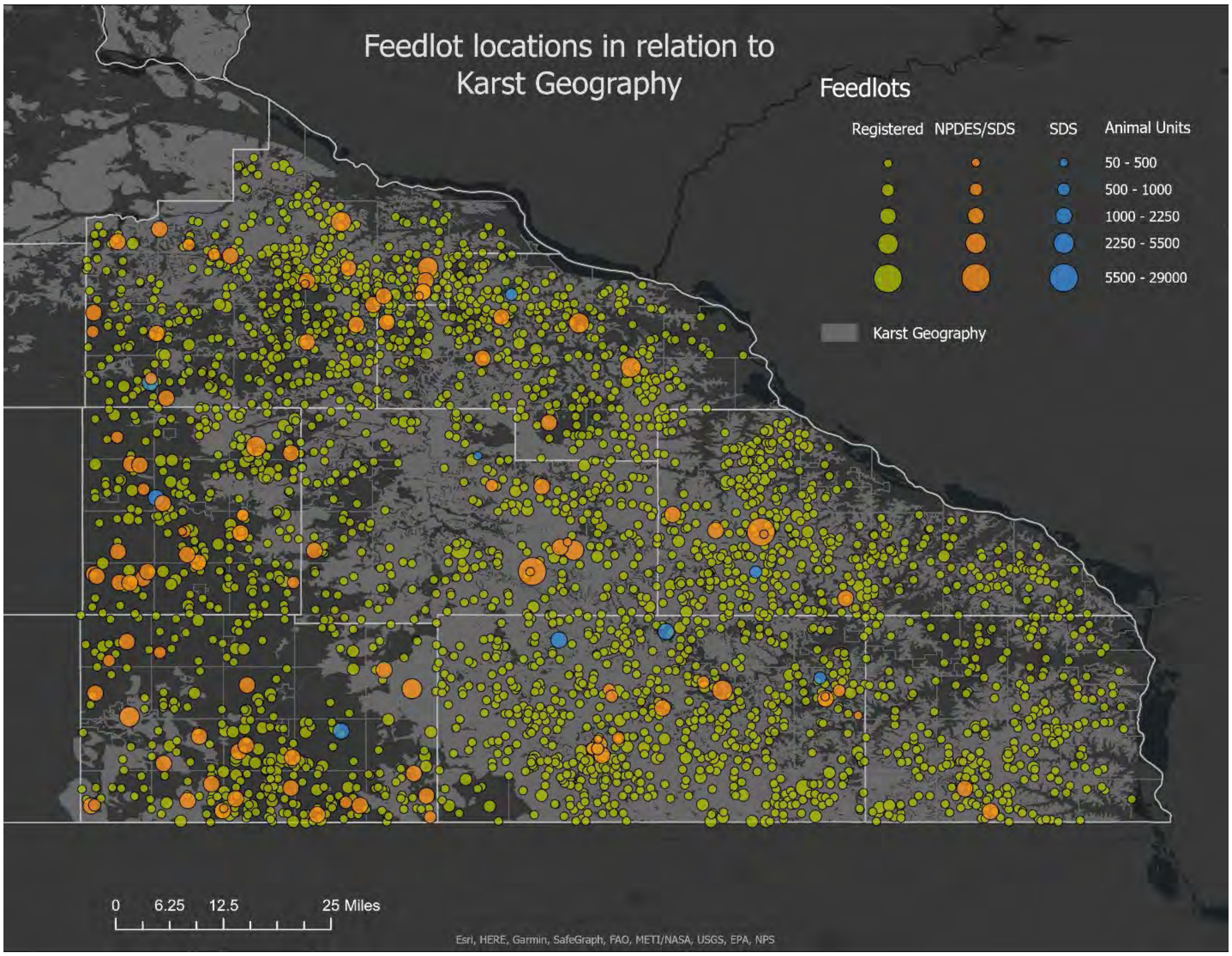

Figure 8: Karst Region Feedlots
Data from MPCA's Feedlots in Minnesota Database

[73] *Counties Delegated to Administer the MPCA Feedlot Program*, MINN. POLLUTION CONTROL AGENCY (Apr. 2022), https://www.pca.state.mn.us/sites/default/files/wq-f1-12.pdf.

The counties that are subject to this Petition house approximately 500,000 dairy cow and cattle animal units and another 260,000 swine units.[74] And the number of feeding operations statewide is on the rise.[75] Current feeding operations also continue to grow: in February 2023, the Fillmore County Board of Commissioners voted unanimously to increase the county's animal unit cap from 2,000 to 4,000 animal units per feedlot.[76] Moreover, almost 65% of the cattle units and over 37% of the swine units are located within landscapes designated as prone to surface karst feature development by MDNR. Those numbers jump to 96% and 69% respectively if we look at facilities within one mile of areas prone to the development of surface karst features.[77]

The storage structures designed to contain millions of gallons of liquid manure, manure piles, and feedlot runoff, can also be significant sources of nitrogen to groundwater in this area.[78] Manure storage structures that are constructed in compliance with National Resource Conservation Service (NRCS) standards are actually designed to leak. According to the NRCS handbook, "properly" constructed lagoons can leak up to 5,000 gallons of manure wastewater per acre per day.[79] In one study conducted by MPCA, "[t]here was evidence of shallow ground water contamination down-gradient of manure storage areas at each [feedlot operation]."[80]

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[74] *Feedlots in Minnesota*, MINN. GEOSPATIAL COMMONS, https://gisdata.mn.gov/dataset/env-feedlots (last visited Apr. 17, 2023).

[75] Sarah Porter & Craig Cox, *Manure Overload: Manure Plus Fertilizer Overwhelms Minnesota's Land and Water*, ENV'T WORKING GRP. (May 28, 2020), https://www.ewg.org/interactive-maps/2020-manure-overload/ [hereinafter *Manure Overload*].

[76] Brian Todd, *Fillmore County doubles its animal unit cap for feedlots*, AGWEEK (Mar. 1, 2023), https://www.agweek.com/news/policy/fillmore-county-doubles-its-animal-unit-cap-for-feedlots.

[77] *Minnesota Regions Prone to Surface Karst Feature Development*, MINN. GEOSPATIAL COMMONS, https://gisdata.mn.gov/dataset/geos-surface-karst-feature-devel (last visited Apr. 17, 2023).

[78] MINN. POLLUTION CONTROL AGENCY, EFFECTS OF LIQUID MANURE STORAGE SYSTEMS ON GROUND WATER QUALITY–SUMMARY REPORT (2001), https://www.pca.state.mn.us/sites/default/files/rpt-liquidmanurestorage-summary.pdf.

[79] U.S. DEP'T OF AGRIC. NAT. RES. CONSERVATION SERV., AGRICULTURAL WASTE MANAGEMENT FIELD HANDBOOK, CHAPTER 10: AGRICULTURAL WASTE MANAGEMENT SYSTEM COMPONENT DESIGN App. 10D-16 (2009), https://directives.sc.egov.usda.gov/OpenNonWebContent.aspx?content=31529.wba ("NRCS guidance considers an acceptable initial seepage rate to be 5,000 gallons per acre per day.").

[80] MINN. POLLUTION CONTROL AGENCY, EFFECTS OF LIQUID MANURE STORAGE SYSTEMS ON GROUND WATER QUALITY–SUMMARY REPORT 2 (2001), https://www.pca.state.mn.us/sites/default/files/rpt-liquidmanurestorage-summary.pdf.

In addition to the manure storage structures themselves, manure from livestock operations in the karst region is commonly used as fertilizer for row crops in the area. When liquified manure storage systems reach capacity, operators must empty them, often by disposing of the liquified manure and process wastewater onto nearby agricultural fields, regardless of the season. These land applications of manure are one of the largest sources of nitrogen from animal feeding operations.[81]

The karst region includes a number of townships, such as Utica and Fremont, that have sandy soils derived from sandstone bedrock. Applications of manure to sandy soils at high agronomic rates leave nitrogen in the soil after the growing season, which then leaches into the groundwater as nitrate, endangering public health.[82] The townships with the highest percentages of private wells exceeding 10 mg/L nitrate concentration have sandy soils or thin soils over karst.

2. Industrial Agriculture

Another major contributor to the nitrate contamination is widespread industrial agriculture in the region. In the eight-county area, 73% of land cover is devoted to agriculture—60% is cropland and 13% is hay or pastureland.[83] This is a high concentration of agriculture for a sensitive karst landscape with a high sensitivity to groundwater contamination. In comparison, only 51% of Minnesota's land cover is devoted to agriculture statewide.[84] A significant portion of this southeastern Minnesota land is related to the animal agriculture in the region: it is used to grow feed crops for

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[81] *Estimated Animal Agriculture Nitrogen and Phosphorus from Manure*, ENV'T PROT. AGENCY (Jan. 11, 2023), https://www.epa.gov/nutrient-policy-data/estimated-animal-agriculture-nitrogen-and-phosphorus-manure.

[82] Michael J. Goss et al., *Chapter Five–A Review of the Use of Organic Amendments and the Risk to Human Health*, 120 ADVANCES IN AGRONOMY 275 (2013), https://doi.org/10.1016/B978-0-12-407686-0.00005-1 ("Spreading manure on the land in fall or winter results in smaller recovery of applied nitrogen by the crops, while the risk of surface runoff, leaching and denitrification is greater.") ("Leaching losses of labeled N from the manure application were considerably greater than those from the original fertilizer application in all years.").

[83] These percentages were calculated using the Multi-Resolution Land Characteristics National Land Cover Database Enhanced Visualization Analysis Tool, *see MRLC NLCD EVA Tool*, MRLC, https://www.mrlc.gov/eva/ (last visited Apr. 17, 2023).

[84] *Agricultural Lands*, MINN. BOARD OF WATER AND SOIL RES., https://bwsr.state.mn.us/agricultural-lands (last visited Apr. 17, 2023).

animals[85] and/or receives the application of manure and waste from the nearby CAFOs as fertilizer.

But much of this fertilizer is over-applied. EWG's modeling found that in 69 of Minnesota's 72 agricultural counties, nitrogen from manure combined with nitrogen in fertilizer exceeded the recommended agronomic rates of MPCA and the University of Minnesota.[86] EWG identified 13 counties in Minnesota where the percent of Nitrogen, from fertilizer and manure combined, was more than 150% of the recommended amount needed to maximize crop yields.[87] Five of these 13 counties are in the karst region.[88] The total estimated nitrogen overload in these five counties is 26,424 tons per year.[89]

The image below shows the coverage of corn and soybeans in the karst region along with average nitrate concentrations at areas near designated trout streams.[90]

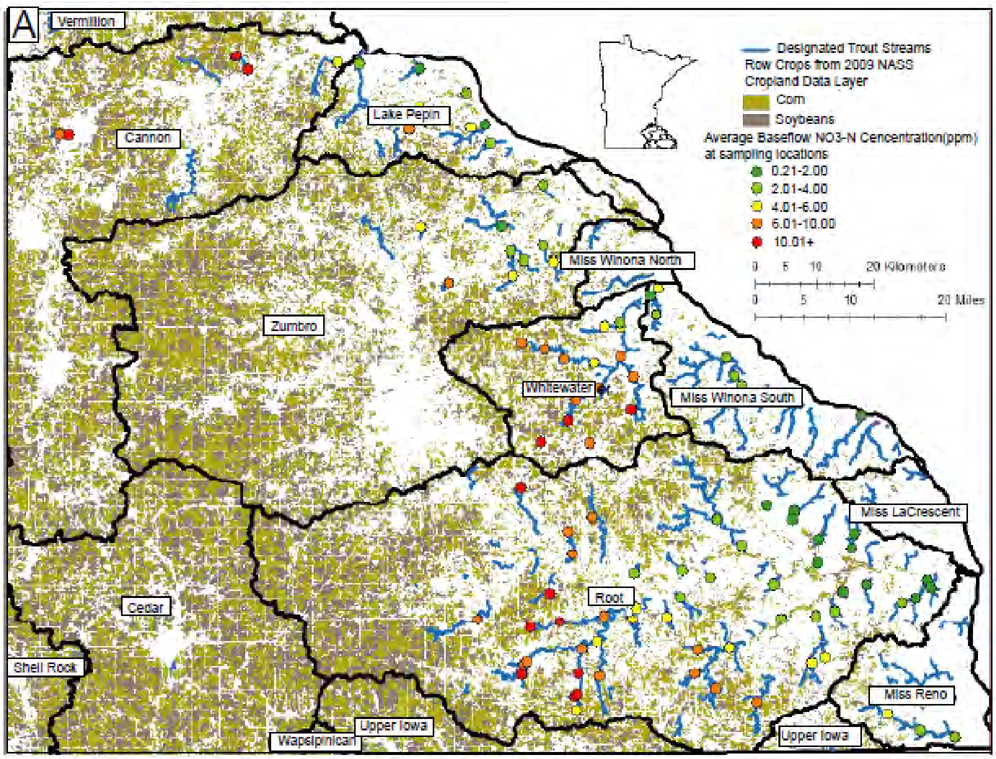

Figure 9: Industrial Agriculture and Nitrate-Contaminated Trout Streams

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[85] Up to 40% of domestic corn use is allocated to livestock feed. *See Feed Grains Sector at a Glance*, U.S. DEP'T OF AGRIC., https://www.ers.usda.gov/topics/crops/corn-and-other-feed-grains/feed-grains-sector-at-a-glance/ (last visited Apr. 17, 2023).
[86] *Manure Overload*, *supra* note 75.
[87] *Id.*
[88] *Id.*
[89] *Id.*
[90] RUNKEL 2013, *supra* note 45, at Fig. 37.

The correlation between land used to grow exclusively corn and soybeans and nitrate pollution is well documented. In a 2020 report, researchers at MDA found that the mean nitrate concentration of lysimeters placed on cropland that was in a constant corn or corn-soybean rotation was 22.3 mg/L.[91] The figure below compares this to other land uses.

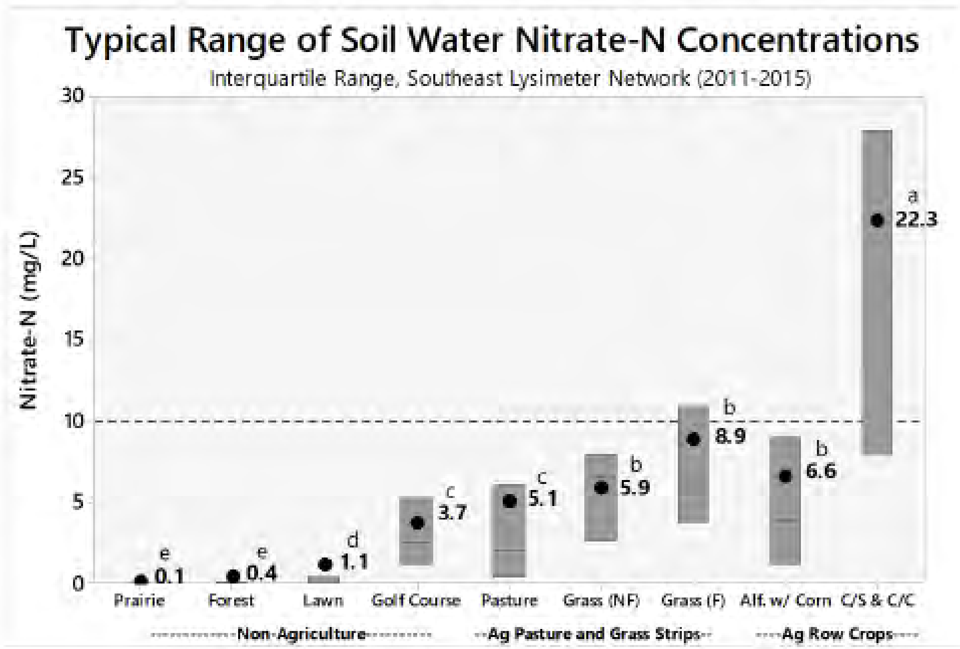

Figure 10: Land Cover and Nitrate Contamination

As Figure 10 demonstrates, industrial agricultural land suffers from significantly more contamination than other types of land uses generating a risk to both surface and groundwater.

D. Conditions in the Karst Region Constitute an Imminent and Substantial Endangerment to Human Health Under the SDWA

The current levels of nitrate in drinking water in the karst region present an imminent and substantial endangerment to human health because consumption of drinking water that is contaminated with nitrate is known to cause serious health risks. Given the thousands of individuals who rely on either contaminated private wells or

[91] KEVIN KUEHNER ET AL., MINN. DEP'T OF AGRIC., EXAMINATION OF SOIL WATER NITRATE-N CONCENTRATIONS FROM COMMON LAND COVERS AND CROPPING SYSTEMS IN SOUTHEAST MINNESOTA KARST 14 (2020), https://wrl.mnpals.net/islandora/object/WRLrepository%3A3654/datastream/PDF/view.

contaminated PWS for drinking water in this region, there is reasonable cause for concern that individuals are, and will be, exposed to this risk at unhealthy concentrations.

Nitrate is plainly an endangerment to public health under the SDWA because EPA not only categorizes it as a "contaminant,"[92] but as an "acute contaminant" known to pose significant health risks. According to EPA, "[n]itrate is an acute contaminant, meaning that one exposure can affect a person's health. Too much nitrate in your body makes it harder for red blood cells to carry oxygen."[93] EPA previously found that nitrate levels above the MCL of 10 mg/L present an imminent and substantial endangerment to human health.[94]

Nitrate is a particularly insidious contaminant because it is colorless, odorless, and tasteless, meaning that people do not have a way of identifying its presence in their drinking water without testing.[95] MNWOO reports that at their testing clinics across the state, many of the people with high nitrate tests were unaware of the contamination and reported that they liked the taste of their well water.

Additionally, boiling nitrate-laden drinking water, as is often done in preparation of baby formula, increases the nitrate concentration of the water because nitrates do not evaporate and become more concentrated in the formula.[96] Shallower aquifers are both more likely to be used for private wells and are more contaminated. For example, in the karst region, the Prairie du Chien aquifer is shallower and much more nitrate contaminated than the deeper Jordan aquifer.[97] But deep wells can also be contaminated. For example, the well on the farm of one of MNWOO's directors is a multi-aquifer well with a total depth of 400 feet, but the water from that well has exceed 13 mg/L nitrates for over 20 years.[98]

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[92] 40 C.F.R. § 141.62(b).

[93] *Frequently Asked Questions About Nitrates & Drinking Water*, ENV'T PROT. AGENCY (Sept. 2012), https://nepis.epa.gov/Exe/ZyPDF.cgi/P10150PM.PDF?Dockey=P10150PM.PDF.

[94] *See, e.g.*, Administrative Order on Consent, *In the Matter of Yakima Valley Dairies*, SDWA-10-2013-0080, at 7 (Mar. 19, 2013) (finding that "above the concentration of 10 mg/L in drinking water, nitrate may present an imminent and substantial endangerment to the health of persons"), https://www.epa.gov/sites/default/files/2017-12/documents/lower-yakima-valley-groundwater-consent-order-2013.pdf.

[95] *Nitrate in Drinking Water*, MINN. DEP'T OF HEALTH (Dec. 8, 2022), https://www.health.state.mn.us/communities/environment/water/contaminants/nitrate.html.

[96] *Frequently Asked Questions About Nitrates and Drinking Water*, ENV'T PROT. AGENCY (Sept. 2012), https://nepis.epa.gov/Exe/ZyPDF.cgi/P10150PM.PDF?Dockey=P10150PM.PDF.

[97] RUNKEL 2013, *supra* note 45, at 45.

[98] Jeffrey S. Broberg, MNWOO founder and board member, personal communication.

Drinking water contaminated with nitrate has well-documented adverse health risks including a variety of cancers, "blue-baby syndrome," and reproductive problems.[99] Childhood brain cancer has been linked to high nitrate levels in drinking water.[100] MDH also reports other potential health effects such as "increased heart rate, nausea, headaches, and abdominal cramps."[101] Nitrate in water supplies has also been linked to spontaneous miscarriages and birth defects.[102]

The numerous studies demonstrating that a contaminant known to cause disease and illness is present at unsafe levels in wells used by tens of thousands of residents proves an unambiguous SDWA "endangerment."

Because the present contamination of the region's drinking water and risk of significant adverse health effects from drinking contaminated water are both thoroughly documented, endangerment is clearly imminent. As explained above, endangerment is "imminent" if conditions that give rise to it are present, even if actual harm has not already been documented in the contaminated area. Unsafe levels of nitrate contamination in the karst region drinking water supply were first identified over 30 years ago,[103] and recent data trends indicate that nitrate contamination is continuing at a persistent—and harmful—level.[104]

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[99] *Nitrate in Drinking Water*, MINN. DEP'T OF HEALTH (DEC. 8, 2022), https://www.health.state.mn.us/communities/environment/water/contaminants/nitrate.html;
N. BEAUDET ET AL., NITRATES, BLUE BABY SYNDROME, AND DRINKING WATER: A FACTSHEET FOR FAMILIES, PEDIATRIC ENV'T HEALTH SPECIALTY UNITS (2014), https://ldh.la.gov/assets/oph/Center-EH/envepi/PWI/Documents/PEHSU_Nitrates_Consumer_1.20.15 FINAL.pdf; Roberto Picetti et al., *Nitrate and Nitrate Contamination in Drinking Water and Cancer Risk: A Systematic Review with Meta-Analysis*, 210 ENV'T RSCH. 112988 (2022), https://www.sciencedirect.com/science/article/pii/S0013935122003152#bib109.

[100] A. Zumel-Marne et al., *Environmental Factors and the Risk of Brain Tumours in Young People: A Systematic Review*, 53 NEUROEPIDEMIOLOGY 121 (2019), https://www.karger.com/Article/Fulltext/500601?utm_source=external&utm_medium=referral&utm_campaign=getFTR; *see also*, Yanqi Xu, *Nebraska's Dirty Water*, THE READER (Oct. 28, 2022), https://thereader.com/2022/10/28/nebraskas-dirty-water/ ("Areas of the state that have higher pediatric cancer rates and birth defect rates also have higher nitrate levels, researchers say.").

[101] *Nitrate in Drinking Water*, MINN. DEP'T OF HEALTH (DEC. 8, 2022), https://www.health.state.mn.us/communities/environment/water/contaminants/nitrate.html.

[102] Allison R. Sherris et al., *Nitrate in Drinking Water during Pregnancy and Spontaneous Preterm Birth: A Retrospective Within-Mother Analysis in California*, 129 ENV'T HEALTH PERSPECTIVES, (2021), https://ehp.niehs.nih.gov/doi/full/10.1289/EHP8205.

[103] ORES 1982, *supra* note 50.

[104] TOWNSHIP TESTING UPDATE 2022, *supra* note 62.

The public health risks associated with nitrate contamination in the karst region constitute a "substantial" endangerment under the SDWA. According to EPA's updated guidance on SDWA emergency authority, an example of substantial endangerment is "a substantial likelihood that contaminants capable of causing adverse health effects will be ingested by consumers if preventative action is not taken."[105] Well sampling has consistently shown elevated nitrate levels in residential drinking water wells across the karst region. Thus, residents of the karst region have been, and continue to be, ingesting this contaminant. This alone demonstrates that the endangerment is substantial.

V. Minnesota Officials Have Failed to Achieve Safe Drinking Water Quality Despite Decades of Attempting to Implement Mitigation Plans

EPA should exercise its emergency authority under Section 1431 of the SDWA because users of USDW and PWSs in the karst region face imminent and substantial endangerment and actions by Minnesota officials have been ineffective. The chronology below describes state agencies' recognition of, and attempts to address, the substantial and imminent endangerment posed by nitrate pollution. The persistent contamination despite these efforts demonstrates their ineffectiveness.

Minnesota enacted the Groundwater Protection Act in 1989. It was based on a growing recognition of the vulnerability of Minnesota's groundwater resources.[106] In part, in was based on groundwater testing in the 1980s that showed nitrate levels exceeding the health limits in 40% of private wells tested and 7% of public wells.[107] It was followed closely by the development of the Nitrogen Fertilizer Management Plan by MDA in 1990.[108] Neither of these initiatives resulted in effective protection of Minnesota's groundwater resources from nitrate pollution, as evidenced by the persistent contamination of private and public water supplies at or above the health risk limit.[109] In 2010, MDA began the process of revising the Nitrogen Fertilizer Management Plan.[110] The updated Nitrogen Fertilizer Management Plan was finalized by MDA in 2015 and led to the Township Testing Program discussed above. One of the objectives for the Township Testing Program was to better grasp the extent and severity of the nitrate

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105 EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at 11 (explaining that an endangerment is substantial "if there is a reasonable cause of concern that someone may be exposed to a risk of harm").
106 JOHN HELLAND, MINN. H.R. RSCH. DEP'T, A SURVEY OF THE GROUNDWATER ACT OF 1989, (2001), https://www.house.mn.gov/hrd/pubs/gdwtract.pdf.
107 *Id*.
108 MINN. DEP'T OF AGRIC., NITROGEN FERTILIZER MANAGEMENT PLAN (2015, addended July 2019), https://www.mda.state.mn.us/sites/default/files/2019-08/nfmp2015addendedada_0.pdf [hereinafter NITROGEN FERTILIZER MANAGEMENT PLAN].
109 JOHN HELLAND, MINN. H.R. RSCH. DEP'T, A SURVEY OF THE GROUNDWATER ACT OF 1989, (2001), https://www.house.mn.gov/hrd/pubs/gdwtract.pdf.
110 NITROGEN FERTILIZER MANAGEMENT PLAN, *supra* note 108, at ix.

contamination problem—which it did. These data were used to inform the development of the Groundwater Protection Rule, which was passed in 2019 but falls short of the regulatory response needed to address the issue for the reasons documented below.

Also in 2010, the Minnesota Legislature approved funds for MPCA to develop aquatic life water quality standards for nitrate, in recognition of the need to protect Minnesota's aquatic life from the toxic effects of high nitrate. In response, MPCA issued its Aquatic Life Water Quality Standards Technical Support Document for Nitrate, which recommended a chronic nitrate standard of 3.1 mg/L to be protective of aquatic life.[111] The MPCA did not adopt water quality standards for nitrate, however, and has continued to defer to that 2010 legislative mandate to this day.

In 2013, MPCA published a report titled "Nitrogen in Minnesota Surface Waters." The report documents the widespread extent of nitrate contamination in Minnesota's waters, noting that in southeastern Minnesota, there are several streams where "groundwater baseflow provides a continuous supply of high nitrate water to streams throughout the year."[112] In other words, MPCA recognized that the groundwater in this area is so polluted, it is polluting the surface water.

In 2014, eleven Minnesota organizations jointly published a Nutrient Reduction Strategy for nitrogen and phosphorous pollution, led by MPCA.[113] The goal was to ultimately reach Minnesota's state water quality goals and downstream impacts like eutrophication in the Gulf of Mexico. In 2020, MPCA issued its 5-year progress report, considering whether the 2014 Nutrient Reduction Strategy was successful. The progress report shows that while phosphorous concentration trends in Minnesota waterways have generally decreased over the past 10-20 years, nitrate concentration trends have increased—in some major rivers by 20-60%. The Progress Report identifies row crop agriculture as the largest source of nitrogen.

Even with overwhelming data and analysis showing the trends and the reasons for concern, more recent strategies have been similarly ineffective. In 2019, MDA finalized

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111 Phil Monson, Minn. Pollution Control Agency, Aquatic Life Water Quality Standards Technical Support Document for Nitrate (2010), https://wrl.mnpals.net/islandora/object/WRLrepository%3A77. Although MPCA's regulatory focus has been on surface water, in the karst region the connection between surface and groundwater is so immediate, that surface water quality standards are highly relevant to protecting groundwater quality.

112 Minn. Pollution Control Agency, Nitrogen in Minnesota Surface Waters 3 (2013), https://www.pca.state.mn.us/sites/default/files/wq-s6-26a.pdf.

113 Minn. Pollution Control Agency, The Minnesota Nutrient Reduction Strategy (2014), https://www.pca.state.mn.us/sites/default/files/wq-s1-80.pdf.

the Groundwater Protection Rule, which has several deficiencies.[114] For example, although fall application of commercial fertilizer is restricted in the karst region, as well as in identified DWSMAs, fall application of manure is not. There are other significant flaws in the rule that fail to adequately protect USDWs. First, the regulatory scope of the rule is limited to DWSMAs for community wells and provides no direct assessment or protection of private wells that fall inside a DWSMA and no assessment or protection for those outside of a DWSMA (see Figure 5 above). As both MCEA and MDH noted in comments on the Groundwater Protection Rule, the Rule should include a mitigation process for private wells and non-community public water supply wells that is equivalent to what it establishes for public water supplies.[115] Without this equitable approach, MDH notes that the rule "does not serve the public health needs of rural Minnesotans, many of whom already suffer inequities relative to public health outcomes."[116] Second, there can be a significant lag time from days to years from the initial contamination of groundwater or surface water from sources of nitrogen and the necessary action taken by the state agencies to address the source. The MDA has the general authority to issue penalties for violations of its rules through Minnesota Statutes 18D, but the Groundwater Protection Rule requires a monitoring period that can last decades before enforcement actions are taken.[117] Lastly, the rule only requires best management practices to be used once a water source reaches mitigation level 3 or 4 contamination and even then, MDA cannot require application rates below that recommended by the University of Minnesota's Extension Services. Since the Groundwater Protection Rule went into effect, none of the DWSMAs with elevated nitrates have been classified at mitigation level 3 or 4, and thirteen mitigation level decisions have been "delayed for good cause."[118] This means that thus far, the Rule continues to rely on voluntary approaches that have not remedied the problem over the last several decades.

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[114] Attached to this Petition as Exhibit A is Petitioner MCEA's Comment to MDA, which explains the deficiencies of the rule in greater detail.

[115] Ex. A; *see also* Minn. Dep't of Health Comment Letter on Proposed Minnesota Department of Agriculture Rules Governing Groundwater Protection, Add. 1 (Aug. 14, 2018), https://speakup-us-production.s3.amazonaws.com/uploads/attachment/file/5b746f627d79656b8800e3cb/MDH_GW_ProtRuleComments.pdf.

[116] Minn. Dep't of Health Comment Letter on Proposed Minnesota Department of Agriculture Rules Governing Groundwater Protection, at 2 (Aug. 14, 2018), https://speakup-us-production.s3.amazonaws.com/uploads/attachment/file/5b746f627d79656b8800e3cb/MDH_GW_ProtRuleComments.pdf.

[117] MINN. DEP'T OF AGRIC., STATEMENT OF NEED AND REASONABLENESS IN THE MATTER OF PROPOSED PERMANENT RULES RELATING TO GROUNDWATER PROTECTION 131-133 (2018).

[118] *Delayed for Good Cause: Drinking Water Supply Management Area Mitigation Level Determination*, MINN. DEP'T OF AGRIC., https://www.mda.state.mn.us/delayed-good-cause (last visited Apr. 21, 2023).

In 2021, MPCA released the final General NPDES Permit for CAFOs, which also has several deficiencies.[119] First, there is no monitoring required to ensure that nitrate is not leaching from storage lagoons into groundwater or whether the land application practices are causing or contributing to water quality problems. Both of these practices are known to contribute nitrate to Minnesota's waters, and all NPDES permits are required to have conditions that assure compliance with applicable limitations.[120] Second, there is no prohibition on fall application of manure, and winter application of solid manure is allowed in December and January. There are also no controls on summertime application of manure on hayfields without incorporation into the sensitive soils of the karst region. Third, there is no required pre-plant testing for nitrate to ensure that farmers properly account for residual nitrates that remain from manure applied in previous years when they calculate expected crop nitrogen needs.[121]

The Minnesota Department of Health is charged with insuring that public water supplies meet drinking water standards and implementing wellhead protection measures.[122] In a March 2021 report, MDH stated that "currently, there are approximately 400,000 acres in vulnerable groundwater Drinking Water Supply Management Areas," and that MDH's Source Water Protection Program "has a goal to protect vulnerable land in DWSMAs statewide by 2034."[123] However, the implementation of land use changes in Source Water Protection Plans is largely voluntary and does not protect underground sources of drinking water supply for private well owners who live outside of DWSMA boundaries. Finally, under the Minnesota Well Code MDH regulates private well construction and initial testing for nitrate and other pollutants like total coliform. However, "private drinking water testing and monitoring are otherwise unregulated and voluntary, with no formal tracking of water quality over time."[124]

Most recently, in 2022, MPCA stated that it was still not going to develop water quality standards for nitrate pollution in surface waters used for recreation and aquatic

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[119] Attached to this Petition as Exhibit B is Petitioner MCEA's Comment to MPCA, which explains the deficiencies of the CAFO General Permit in greater detail.
[120] 33 U.S.C. § 1342(a)(2); *see also* 40 C.F.R. § 122.48(b), Minn. R. 7001.0150 subp.2B.
[121] Ex. B at 22-23.
[122] James Lundy et al., *Minnesota's 1989 Ground Water Protection Act: Legacy and Future Directions*, 5 MINN. GROUNDWATER ASSOC. (2022).
[123] *Protecting Vulnerable Drinking Water Sources*, MINN. DEP'T OF HEALTH (March 23, 2021), https://www.health.state.mn.us/communities/environment/water/docs/cwf/vulnacres.pdf.
[124] James Lundy et al., *Minnesota's 1989 Ground Water Protection Act: Legacy and Future Directions*, 5 MINN. GROUNDWATER ASSOC. 34 (2022).

life, despite the recognition that such a standard is necessary.[125] The State's repeated failures to mitigate nitrate levels in drinking water put more and more people at risk of drinking contaminated water. Allowing agricultural practices to continue in the karst region without meaningful changes to commercial fertilizer application, manure management, and manure disposal practices, will perpetuate the imminent and substantial endangerment to residents' health in direct violation of the SDWA. Although Minnesota officials have clear authority to adopt the mandatory regulations necessary to resolve the imminent and substantial endangerment, they have consistently refused to act. EPA must not let Minnesota officials continue to sit on the sidelines for another decade as the threat to the health of Minnesota citizens grows ever more severe.

VI. Requested Emergency Action to Abate Ongoing and Ever-Increasing Endangerment to Human Health from Nitrate Contamination

As discussed in detail above, the statutory prerequisites for emergency action under 42 U.S.C. § 300i are satisfied here. First, nitrate, which is a "contaminant" under the SDWA, is present in and continues to leach into USDW in the karst region. Second, the presence of nitrate contamination in groundwater is causing an imminent and substantial endangerment to public health; an alarming number of karst region residents rely on USDW that have been identified as carrying substantial nitrate risks for users. Finally, the State of Minnesota has not taken timely or effective action to abate the public health endangerment.

EPA has broad authority to investigate and remediate threats to public health under the SDWA. "Once EPA determines that action under Section 1431 is needed, a very broad range of options is available" as necessary to protect users of USDW.[126] The tools available to EPA include conducting studies, halting the disposal of contaminants that may be contributing to the endangerment, and issuing orders such as mandatory changes to manure generation, handling, and land application practices. In fact, "EPA may take such actions notwithstanding any exemption, variance, permit, license, regulation, order, or other requirement that would otherwise apply."[127]

EPA should prioritize investigating and abating nitrate contamination in the karst region. Specifically, Petitioners respectfully request EPA take at least the following measures under its SDWA Section 1431 emergency powers, either by administrative order or through civil action:

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[125] PHIL MONSON, MINN. POLLUTION CONTROL AGENCY, AQUATIC LIFE WATER QUALITY STANDARDS TECHNICAL SUPPORT DOCUMENT FOR NITRATE (2010), https://www.pca.state.mn.us/sites/default/files/wq-s6-13.pdf.

[126] EMERGENCY AUTHORITY GUIDANCE, *supra* note 12, at 14.

[127] *Id.* at 9.

Investigation and Risk Assessment:

- Conduct investigation and monitoring throughout the karst region to more accurately trace the sources and quantities of nitrogen pollution, and to identify which sources are causing nitrate contamination;
- Investigate MPCA's CAFO permit requirements and MDA's and MPCA's best management practices for nutrient management to determine why they have been unsuccessful at protecting groundwater in the karst region;

Engagement and Communication:

- Work with MDH to notify the public of the existing nitrate hazards and provide public updates throughout the process of returning drinking water to a safe condition;

Planning:

- Determine what enforcement measures should be implemented to effectively reduce nitrogen pollution from CAFO and industrial agriculture sources;
- Provide a timetable for implementing a remedy to abate nitrate contamination from identified contaminators;

Assistance:

- Order the parties responsible for the nitrate contamination to supply free water testing and ensure a free source of clean drinking water to residents of the karst region whose private wells or PWSs exceed safe limits for nitrate to prevent blue-baby syndrome, cancer, and other adverse health effects;
- Provide assistance to private well owners to engage in effective private well management practices;

Regulation:

- Prohibit CAFOs from opening, expanding, or modifying operations in the karst region unless and until nitrate concentrations in wells with historically high levels of nitrate consistently fall below the MCL of 10 mg/L;
- Require CAFOs and agricultural operators land-applying CAFO waste or other nitrogen fertilizers to modify their practices so that these operations will cease overburdening the area with nitrogen pollution via lagoon leakage, land application of manure, and/or spills and leaks.

The threat to public health in the karst region from nitrate pollution of groundwater is present and pervasive, and all signs indicate a continuation and exacerbation of dangerous contamination levels absent EPA action. Therefore, the undersigned Petitioners respectfully request that EPA use its emergency powers under the SDWA to take the actions necessary to abate the sources of contamination that increasingly place the public at substantial risk and provide other forms of relief within its authority as long as the endangerment persists.

VII. Conclusion

In conclusion, for the reasons stated above, the undersigned Petitioners respectfully request that EPA invoke its emergency authority under Section 1431 of the Safe Drinking Water Act to urgently address the imminent and substantial endangerment to public health within the karst region of Minnesota caused by ongoing and increasing nitrate contamination. Please contact the undersigned for more information regarding this Petition.

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Exhibit A

Minnesota Center for Environmental Advocacy

Using law, science, and research to protect Minnesota's environment, its natural resources, and the health of its people.

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August 15, 2018

VIA ELECTRONIC SUBMISSION

Administrative Law Judge Jessica Palmer-Denig

Re: *Proposed Rules Governing Groundwater Protection, Minnesota Rules, 1573;*
Revisor's ID Number RD4337,
OAH Docket No. 71-9024-35205

Dear Administrative Law Judge Palmer-Denig:

This letter includes the comments of the Minnesota Center for Environmental Advocacy (MCEA) on the Minnesota Department of Agriculture's (MDA) Proposed Rules Relating to Water Resource Protection Requirements. MCEA is a Minnesota nonprofit environmental organization whose mission is to use law, science and research to preserve and protect Minnesota's wildlife, natural resources, and the health of its people. MCEA has statewide membership. MCEA is concerned about the impacts of agricultural pollution on Minnesota's waters and has been engaged with MDA on issues related to nitrogen fertilizer management for a number of years, including commenting on MDA's 2017 rule proposal.[1]

MCEA has supported its comments with references with numerous published documents which are provided as exhibits to this letter.

I. INTRODUCTION AND SUMMARY OF POSITION

MCEA agrees that a rule to protect groundwater from nitrate contamination is needed: indeed, it is long overdue.[2] Documentation in the record supplied by the Minnesota Department of Agriculture ("MDA") establishes that the voluntary best management practices ("BMPs") have failed to reduce or stabilize nitrate concentrations in groundwater in many areas of the state, and that those

[1] A copy of MCEA's 2017 comment letter is included as an attachment.
[2] Chapter 103H was enacted in 1989, with the goal of preventing degradation of the groundwater by human activities. Where prevention is practicable, it is intended that it be achieved. Minn. Laws 1989, ch. 326, art. 1, section 1.

concentrations continue to grow. Even where fully adopted, the BMPs are not enough to reduce excessive nitrate levels where they already exist.[3] More is needed.

MCEA supports the proposed fall and frozen soils application ban in “vulnerable areas” and in drinking water system management areas (“DWSMAs”) where N has exceeded 5.4 mg/L at any time in the past 10 years. However, the fall application ban part of the rule as proposed is riddled with convoluted and unsupported exclusions and exceptions which will make the fall application ban difficult to implement. Most importantly, the record shows that simply restricting the timing of nitrogen fertilizer application will not meet the statutory goals in those areas that are vulnerable to contamination. In fact, restricting the timing of application is one of the least effective of the University of Minnesota nitrogen fertilizer application recommendations.[4] At the very minimum, the record shows that in these vulnerable areas of the state, all the University of Minnesota “recommended” practices, including rate, timing, source, and placement, must be mandated to have a significant impact on excessive nitrate levels, with a particular focus on the “right rate” of nitrogen fertilizer.[5] And likely more actions must be required in order to prevent exceedances of the nitrate Health Risk Limit (“HRL”) in these areas.[6]

MCEA supports the issuance of Water Resource Protection Requirement orders (“WRPRs”) by the commissioner, but believes that the proposed rule too narrowly restricts the use of such WRPRs to public water supply system protection areas. Protection is also needed for people who drink well water. MCEA also believes that the proposed rule fails to provide adequate due process when a WRPR is issued: both “responsible parties,” and people who drink groundwater, must have the right to challenge the order.

Below, MCEA has provided alternatives that are supported by the record and that will not result in a substantially different rule within the meaning of Minn. Stat. § 14.05, subd. 2, but which will result in a rule that is in compliance with Minn. Stat. § 103H.275, subd. 1(c), by requiring water resources protection requirements that are “designed to prevent and minimize the pollution to the extent practicable” and, most importantly, are “designed to prevent the pollution from exceeding the health risk limits.”

The main issues with the rule are as follows.

A. The Proposed Rule Fails To Comply With Statutory Authority And Is Arbitrary Because It Does Not Protect People Who Drink From Private Wells

Persons who use water supplied by municipal or rural water supply providers are protected against drinking high nitrate levels by existing regulations requiring testing and which ensure a

[3] This is not surprising because, while helpful in controlling nitrogen fertilizer-related pollution, the BMPs were developed from research based on yield optimization and the production economics of corn and not specifically on water quality indices. Randall, *Nitrogen BMP’s for Corn in Minnesota* (provided in the exhibits).
[4] Wall, *Nitrogen in Minnesota Surface Waters*, Minnesota Pollution Control Agency (June 2013). *See also* comments filed by Dr. Gyles Randall, August 1, 2018.
[5] *Id.*
[6] *Id.*

healthy water supply.[7] When a community water supply well becomes contaminated, community water supplies typically have various options to deal with it.[8] In contrast, people drinking from a private well may not test on a regular basis[9] and suffer the same costs[10]—but with fewer options—when their water becomes contaminated. Despite these facts, the rule as proposed only protects persons who use water supplied by municipal or rural water supply providers.[11] The proposed rule should be amended to require mandatory requirements and WRPRs in township areas where excessive nitrate levels are present based on available test results. This change is supported by the record. Indeed, the MDA notes that it initially considered implementing regulatory actions "on the township level" in 2017, and further admits that in at least twenty townships more than 10% of the people who voluntarily sampled their wells are drinking water that exceeds the health risk limit for nitrate.[12] The only reason offered as to why townships with significant private well contamination levels were not included in the published rule is the lack of resources and a preference on the part of affected responsible parties to have the program stay voluntary.[13] These reasons do not provide an adequate basis for the decision to abandon private well users and this decision is inconsistent with the MDA's duty under Minn. Stat. § 103H.275, subd. 1(c)(1) and (2). Furthermore, the MDA has undermined its "limited resources" argument by noting that "the MDA will implement the voluntary parts of the 2015 NFMP in townships up to level 2, including forming [Local Advisory Teams] and conducting groundwater

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[7] The federal Safe Drinking Water Act standards apply to community water systems in Minnesota. *See* 42 U.S.C. § 300f et seq. The Safe Drinking Water Act standards are enforced by the Minnesota Department of Health. *See* https://data.web.health.state.mn.us/drinkingwater.

[8] As noted by the Department of Health, community water systems can take a high nitrate well and reclassify it to only be used in case of emergency, remove the well from service, or seal the well so that it cannot be used again. While these strategies may appear to be more economical than adding a treatment process, there are still costs associated with each strategy - locating a new well site, drilling a new well, or treating for a different contaminant. *See* 2017 Annual Report at 15, available at http://www.health.state.mn.us/divs/eh/water/com/dwar/report2016.pdf.

[9] According to the Minnesota MDA of Health, "Twenty-one percent of Minnesotans (1.2 million people) get their drinking water from a private well. Private well users are not afforded the same water quality safeguards as people who get their water from public water systems. While public water systems make sure water is safe for the end-user, private well users are responsible for making sure their water is safe for everyone in the household to drink." http://www.health.state.mn.us/divs/eh/cwl/wells/index.html.

[10] In 2008, average remediation costs were \$190 y-1 to buy bottled water, \$800 to buy a NO3 removal system plus \$100 y -1 for maintenance, and \$7,200 to install a new well. Lewandowski, A. M., Montgomery, B. R., Rosen, C. J., & Moncrief, J. F. (2008*). Groundwater nitrate contamination costs: A survey of private well owners*. Compare to increased public water supply costs cited in http://www.house.leg.state.mn.us/comm/docs/CostofNitrateContaminationtoPublicSuppliers2007.pdf.

[11] The attached map demonstrates how little area is *potentially* covered by the proposed rule (the black circled areas), as opposed to the areas where townships have already tested as having more than 5 percent wells above the HRLs.

[12] Statement of Need and Reasonableness dated April 30, 2018 ("SONAR") p. 110.

[13] *Id*.

monitoring."[14] It is unreasonable for MDA to prioritize its limited resources to require action to reduce nitrate contamination for public water supply users who are already guaranteed clean water over private wells owners who do not have such a guarantee. Moreover, if resources are limited, the MDA has non-arbitrary means for deciding how to allocate these resources, such as phasing in a program based on priorities, which this rule already identifies.[15] MDA's decision to abandon private well owners from the protections of the rule is arbitrary for the same reasons that the Minnesota Department of Labor and Industry was found to have acted arbitrarily in *Builders Ass'n of Twin Cities v. Minnesota Dept. of Labor and Industry*, 872, N.W.2d 263 (Minn. App. 2015). In that case, the Court of Appeals concluded that it was unreasonable for the Minnesota Department of Labor to adopt a building code that failed to require smaller homes to be protected by sprinkler systems where the record supported the potential for a phase-in of sprinkler requirement. MDA has not provided a reasonable basis for making WRPR protection available to only some of the millions of Minnesota residents who use drinking water as their major source of water – the nearly 30% of those residents excluded from these protections are those most in need. Private well users must be included; fundamental fairness compels nothing less.[16]

The following chart reflects a reasonable system to protect private well users from nitrogen fertilizer-related pollution which could be adopted as part of this rule in addition to the current provision protecting those who consume water from community drinking water sources:

Mitigation Level ("ML")	Criteria	Required actions for the commissioner and responsible parties[17]	Transition to higher level
1	At least 3 to less than 5% of private wells tested exceed the HRL within a township	Commissioner provides education and compliance resource information to all responsible parties within the township; Commissioner provides notice of opportunity to form a local advisory team ("LAT"). All responsible parties required to maintain and produce (on request) nitrogen fertilizer application records.	Exceed criteria for ML1.

[14] SONAR p. 111.
[15] *See* proposed 1573.0050, subp. 1, Item D (prioritization criteria for WRPRs).
[16] MCEA refers MDA to the petition filed as a separate comment today, signed by close to 200 individuals, that asks MDA to protect the drinking water of individual well owners contaminated by nitrates, not just city water supplies.
[17] MCEA also proposes, as discussed below, that the designation of a mitigation level area include certain reasonable actions that can be taken by responsible parties prior to the issuance of a WRPR. The actions shown in this chart are the same as those proposed by MCEA for the equivalent DWSMA mitigation level areas, creating a level playing field for responsible parties in DWSMA areas and township areas.

2	At least 5 to less than 10% of private wells tested exceed the HRL within a township	ML 1 actions; Responsible parties: • comply with no-risk nitrogen BMPs; • obtain yearly subsoil nitrogen samples (Nebraska program) and produce upon request.	Exceed criteria for ML2.
3	Greater than 10% of private wells tested exceed the HRL within a township	ML2 actions; Responsible parties: • develop and implement a nutrient management plan; • comply with all other actions required by the commissioner in a WRPR.	Exceed criteria for ML3.
4	Greater than 15% of private wells tested exceed the HRL within a township	ML3 actions; Responsible parties comply with all other actions required by the commissioner in a WRPR.	

B. The Rule Arbitrarily Prolongs Reliance On Voluntary Best Management Practices To Reduce Nitrates In Groundwater Despite Evidence That The Best Management Practices Have Not Succeeded In Controlling Nitrate Levels. Further, The Rule Allows The Voluntary Compliance To Continue For An Indeterminate Period Of Time

Initially, the rule allows the commissioner to establish only Mitigation Level ("ML") 1 and 2 areas. In these areas, there are no mandatory requirements and WRPRs cannot be issued, despite the fact that in ML2 areas the water is predicted to exceed the health risk limit ("HRL") in 10 years or has already had a reading in excess of the HRL. In the ML1 and 2 areas, MDA proposes only to try—again—to get responsible parties to use the nitrogen BMPs to control nitrate levels. This is manifestly unreasonable because the MDA has admitted in the SONAR that the existing nitrogen use BMPs have not proven to be a successful means for reducing nitrate levels, particularly due to adoption failure.[18] Worse, the proposed rule prohibits the commissioner from evaluating the impact of the nitrogen use BMPs for "at least three growing seasons" or the "lag time," whichever is longer. Lag times can be decades. The phrase "at least" is not limiting. As a result, the proposed rule unreasonably and arbitrarily allows the commissioner to prolong this monitoring period, potentially for decades, regardless of whether the nitrogen use BMPs have been implemented and regardless of whether nitrate levels continue to increase in the subsoil.[19] Thus, voluntary activities can be continued for an endless period of time, regardless of result.

[18] SONAR part IV, pp. 49-59.
[19] Proposed rule 1575.0040, subp. 7, Items G and H allow the commissioner, with unfettered discretion, to postpone mandatory actions for an additional 3 or more growing seasons if the commissioner determines that the "responsible parties…have demonstrated progress in

The MDA cannot have it both ways. The MDA cannot continue to rely on voluntary BMP compliance while admitting that voluntary compliance has not been effective. If the MDA believes one last voluntary period is justified, then that period must be carefully limited by the rule and not be subject to extension. The commissioner should react to the data—not BMP compliance—to determine when more action is needed.[20]

Further, MCEA believes that the record supports a decision to require responsible persons in all areas where elevated nitrate levels are detected (both for public and private wells) to require compliance with certain reasonable requirements such as recordkeeping *before* a site specific WRPR is issued, in particular in areas where exceedance of the health risk limit is statistically likely to occur.

The following table shows reasonable criteria for establishing mitigation levels for areas served by public wells and private wells. This table also shows reasonable actions that MDA could require responsible parties to take prior to WRPR issuance. MCEA believes these actions are needed and reasonable to ensure that the goal of the Groundwater Protection Act—to *prevent* groundwater from exceeding HRLs—is met.

Mitigation Level ("ML")	Criteria	Required Actions for Commissioner and Responsible Parties	Transition to higher level
1	One reading of 3.0 mg/L or greater in a public water supply well(s) At least 3 to less than 5% of private wells tested exceed the HRL within a township	• Commissioner provides education and compliance resource information.[21] • Commissioner provides notice of opportunity to form a local advisory team ("LAT"). • All responsible parties required to maintain and produce (on request) nitrogen fertilizer application records.	ML 1 stays a ML1 so long as it does not meet the criteria for a ML2.

addressing nitrates…" or if there is a "significant change in land use in a drinking water supply management area." Neither "demonstrated progress" nor "significant change" are defined in any manner that would allow a party to determine with any certainty what these statements mean. The lack of enforceability of these rule provisions contravenes the statutory goals and is unsupported by the record.

[20] Although MDA suggests that is it is required by statute to "evaluate" BMP adoption before it can issue a WRPR, Minn. Stat. § 103H.275 says nothing about evaluation of BMP adoption before a WRPR can be issued. Instead, the statute indicates that the contents of a WRPR—the requirements in the WRPR—must be based on "the use and effectiveness of best management practices." The BMPs already exist. If the BMPs have been effective, they can be included in the WRPR. If they have not been effective, they should not be included in the WRPR. But in any event, BMP adoption levels are not mandated as a pre-condition for issuance of a WRPR.

[21] This would include providing the recommended BMPs for the area.

2	One reading of 5.4 mg/L or greater in a public water supply well(s). At least 5 to less than 10% of private wells tested exceed the HRL within a township	All ML1 activities plus: • All responsible persons required to obtain yearly subsoil nitrogen samples (Nebraska program) and produce upon request.	ML2 becomes a ML3 if statistics show HRL will be exceeded in 10 years.
3	One reading of 7.0 mg/L or greater in a public water supply well(s). Greater than 10% of private wells tested exceed the HRL within a township	All ML 2 activities plus: • The No-risk Nitrogen Fertilizer Use BMPs. • Compliance with a Nutrient Management Plan. • [Commissioner issues WRPR based on priority criteria.]	ML3 becomes an ML4 if the health risk limit is exceeded.
4	One reading of 8.0 mg/L or greater in a public water supply well(s). Greater than 15% of private wells tested exceed the HRL within a township	All ML 3 activities plus: Commissioner issues a WRPR based on priority criteria that must include AMTs.	

Neither recordkeeping[22] nor subsoil sampling are presently included in the rule as actions that responsible parties should take at lower mitigation levels, yet these actions would provide the commissioner information that the commissioner could use to determine whether BMPs are being complied with and are being effective, and would not be costly.[23] The sampling is reasonable because it is currently conducted by Nebraska producers and others.[24] Recordkeeping is reasonable because compliance with the BMPs requires recordkeeping, and any producer applying nitrogen fertilizer (or their agent or consultant) would be required to have such records.[25] The requirement for responsible parties in ML3 areas to comply with nitrogen fertilizer BMPs and nutrient management plans immediately upon triggering the ML3 designation is reasonable because these actions will not significantly increase costs for the

[22] Recordkeeping is only required after a WRPR is issued. *See* 1573.0060, Item A(1).
[23] In fact, many Minnesota producers are already keeping such records and taking such samples. *See* testimony of Zach Johnson and Richard Syverson, July 25, 2018.
[24] *See Id.*; SONAR p. 122.
[25] *See* http://www.mda.state.mn.us/nitrogenbmps.

responsible parties,[26] and it may take some time for the commissioner to develop and issue a WRPR. In the interim, because the health risk limit may shortly be exceeded, it is reasonable to require the responsible parties to take immediate actions to better document and control nitrogen fertilizer use.

C. The Rule Lacks Adequate Due Process When The Commissioner Issues A WRPR Order, And Limits The Commissioner's Discretion To Include Effective Conditions

Although the rule requires notice to be given to affected persons prior to issuance of a WRPR as required by statute, only "responsible persons" subject to the order can seek review, which is unfair to the affected persons drinking the water. All persons impacted by the WRPR must be provided an opportunity for administrative and judicial review. Further, no standard is stated in the rule against which the commissioner's decision will be judged to determine whether it meets the standards of the statute. The rule should—at a minimum—require that a WRPR "prevent and minimize the pollution to the extent practicable" and be "designed to prevent the pollution from exceeding the health risk limits."[27] Finally, the review process lacks basic standards necessary to limit frivolous appeals, and appears to confuse "contested case hearings" with "public hearings."

D. The Rule Unreasonably Limits The Commissioner's Discretion To Require Actions That Would Reduce Nitrogen Concentrations Where Necessary To Ensure That The Health Risk Limit For Nitrate Is Not Exceeded

The proposed rule fails to require the commissioner to include certain basic content that should be required in the WRPR, including monitoring, record-keeping, reporting, and the like. But more importantly, the proposed rule limits the commissioner's authority to require certain actions in a WRPR that are immediately effective to reduce nitrogen—alternative management tools—just because the alternative management tool might cost money to implement. Similarly, the proposed rule limits the commissioner's authority to require any changes to the "primary crop" and limits the use of nitrogen fertilizer to levels below rates the University of Minnesota has identified as the most profitable. Although undefined, it would appear that this provision would limit the commissioner's ability to require, for any area for any time, a different crop to be grown (say alfalfa as part of a rotation on a particular field), as part of a WRPR. These limitations are unreasonable and unsupported by the record and do not meet the goals stated in Minn. Stat. § 103H.275. Instead, if there is a particular requirement that would cause hardship for a responsible party to implement, the commissioner should have the authority to enter into a two-year schedule of compliance that would allow a regulated party to make the necessary adjustments to come into compliance.

E. The Rule Contains Many Provisions That Provide The commissioner Too Much Discretion, As Further Described Below

The rule uses the phrase "as determined by the commissioner" in four places and the phrase "if the commissioner determines" in seven places. This language does not meet the standard for a

[26] Throughout this record it is noted that compliance with nitrogen BMPs may save producers money.
[27] Minn. Stat. § 103H.275, subd. 1(c)(1)(2).

rule, because it vests the decision in the commissioner without establishing a standard or a process. For example, all areas where "exclusions" can be established from the ban on fall nitrate fertilizer application are "as determined by the commissioner." This fails to meet the standard for administrative rules, which cannot allow excessive and unfettered discretion such that a party is unable to determine how the rule will be applied. The Administrative Law Judge must reject a rule if it "is not a "rule" as defined in Minnesota Statutes, section 14.02, subdivision 4, or by its own terms cannot have the force and effect of law."[28] This rule cannot be determined by its own terms, because it relies on decisions by the commissioner based on unstated criteria in many provisions. In fact, this lack of standards for WRPRs makes it extremely difficult to determine whether the rule will have any positive impact – the commissioner could rely on exclusions and issue WRPRs that include very minimal requirements (there is no stated standard for the commissioner's WRPR order, just a list of potential options that could be included in a WRPR), and implement the rule in a manner that contradicts the goals of the Groundwater Protection Act.

F. The Rule Contains Many Provisions That Are Fatally Vague, As Further Described Below

For example, the proposed rule does not establish a deadline in part 1573.0040, subpart 2, for the commissioner to designate a DWSMA as a mitigation level 1 or 2 following receipt of information from the Department of Health ("MDH") that a public well has exceeded a trigger level as set forth in subpart 3. To be enforceable, the rule must establish a deadline for the commissioner to act, i.e., within 60 days of receipt of information from MDH.

In addition to the above, the rule contains numerous provisions that are poorly drafted and should be fixed to ensure that the rule can be enforced.

II. FACTUAL AND LEGAL BACKGROUND

A. Nitrogen Fertilizer Use And Nitrate Contamination In Minnesota

The following are the underlying facts pertaining to these proposed rules that must be taken into consideration in evaluating whether the proposed rule meets the statutory standard.

Despite MDA's years of promoting compliance with the University of Minnesota nitrogen fertilizer use recommendations, nitrogen fertilizer sales in Minnesota skyrocketed by nearly 200,000 tons/year from 1990 to 2016, including a 15% increase over the past 5 years.[29] In addition the acreage of crops that "leak" nitrogen fertilizer into groundwater, corn and soybeans, are consistently expanding, with over 4 million more leaky acres today than in 1990.[30]

The result is widespread nitrate contamination of groundwater in Minnesota's agricultural landscapes. Nearly half of the wells in MDA's shallow groundwater monitoring network exceed

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[28] Minn. R. 1400.2100 (g).
[29] MDA Draft Nitrogen Fertilizer Rule Presentation, at slide 24, found at http://www.mda.state.mn.us/~/media/Files/chemicals/nfmp/nfrpresentation.pdf (last visited Aug. 14, 2018).
[30] *Id.* at slide 25.

the nitrate Health Risk Limit ("HRL") of 10 mg/L.[31] Where shallow wells are contaminated, deeper wells also are likely contaminated.[32]

The Minnesota Department of Health reviewed data for 2014 – 2015 from Minnesota's public water supply wells across the state and found that 537 of 10,519 (5.11 percent) had nitrate levels above 3 mg/L. These include wells for both communities and for businesses, schools, and organizations that provide water to the public.[33]

The Minnesota Department of Agriculture's Township Testing Program ("TTP") provides testing for nitrate to homeowners who have wells in vulnerable areas of the state where groundwater used for drinking water can be affected by agricultural production. As of March 2018, 242 vulnerable townships from 24 counties participated in the TTP from 2013 to 2017. In the 242 townships tested, 113 (47%) have 10% or more of the wells over the HRL for Nitrate-N. Overall, 10.1% (2,583) of the 25,652 wells voluntarily tested exceeded the HRL for Nitrate-N.[34]

And these numbers are expected to rise: changes to cropping practices can be expected to result in an increased risk of nitrogen loading.[35]

B. Statutory Requirement For WRPRs

The Groundwater Protection Act of 1989 has the goal of preventing groundwater degradation.[36] For agricultural chemicals and practices, including the use of nitrogen fertilizer, the statute is implemented by the MDA, and requires MDA to evaluate the detection of agricultural pollutants in the state's groundwater;[37] monitor groundwater for pollutants found to be of "common detection" as the result of normal use of a product or practice;[13] develop voluntary, practicable measures that are capable of preventing and minimizing degradation of groundwater from agricultural chemicals and practices, called BMPs;[38] and promote and evaluate the use and effectiveness of these BMPs.[39]

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[31] *Id.* at 2-83.
[32] In 2010, MDA installed eight new wells in the Central Sands Region, approximately 10-15 feet deeper than existing shallow well sites. Id. at 2-75. 75% of these wells exceeded the Health Risk Limit. *Id.* at 2-83.
[33] *Minnesota Drinking Water 2017*, Annual Report for 2016, Minnesota Department of Health Environmental Health Division Section of Drinking Water Protection, available at http://www.health.state.mn.us/divs/eh/water/com/dwar/report2016.pdf
[34] http://www.mda.state.mn.us/~/media/Files/chemicals/nfmp/ttpudate201806.pdf
[35] Keeler and Gourevitch et al, *The Social Costs of Nitrogen*, Sci. Adv. 2016, at 6. The mechanisms are graphically explained at http://www.bwsr.state.mn.us/practices/farm-bill/FBAP_Winter_Meeting/2015/Estimating_the_External_Costs_of_Nitrogen_Fertilizer_in_MN.pdf.
[36] Minn. Stat. § 103H.001.
[37] Minn. Stat. § 103H.251, subd. 1. 13 Minn. Stat. §§ 103H.251, subd. 1(b) and 103H.005, subd. 5.
[38] Minn. Stat. §§ 103H.151, subd. 2 and 103H.005, subd. 4.
[39] Minn. Stat. §§ 103H.151, subd. 3 and 103H.275, subd. 1.

If implementation of BMPs proves ineffective, the Act provides MDA with the authority to adopt mandatory water resource protection requirements (WRPRs) that include "design criteria, standards, operation and maintenance procedures, practices to prevent releases, spills, leaks, and incidents, restrictions on use and practices, and treatment requirements."[40] WRPRs may be statewide or targeted, but those that are not statewide become effective only in areas designated by order of the MDA Commissioner.[41] WRPRs must be intended to prevent and minimize groundwater pollution to the extent practicable; be designed to "prevent the pollution from exceeding the health risk limits;"[42] and be based on "the use and effectiveness of best management practices, the product use and practices contributing to the pollution detected, economic factors, availability, technical feasibility, implementability, and effectiveness."[43] Although economic factors can be considered in decisions, these factors do not trump the overall goals established for the Act and cannot be paramount in view of overarching state policy in support of maintaining the resources of the state for the use of future generations.[44] Further, economic considerations cannot be limited to just those related to the cost to the responsible party; MDA must consider the cost of not acting on the affected public, who must pay to replace contaminated water supplies, as noted above.

Where this rule does not meet the intent of Groundwater Protection Act, MCEA requests that the Administrative Law Judge recommend changes to the rule that will ensure that it meets the minimum goals of the Groundwater Protection Act, in particular that the actions "prevent the pollution from exceeding the health risk limits" rather than allowing the status quo to continue, as that status quo has not succeeded in reducing impacts from nitrogen fertilizer to the groundwater as required by law.

III. MDA'S PROPOSED RULE: DETAILED PART BY PART ANALYSIS

MCEA provides detailed comments on the proposed rule below. In addition, MCEA has prepared a separate redline document of the proposed rule (attached). The proposed redline language addresses the problems identified in the proposed rule language and includes MCEA's proposed language.

A. DEFINITIONS (1573.0010):

1573.0010, subp. 2. Alternative management tools ("AMTs") are "specific practices and solutions described in part 1573.0090, subpart 1. . .that are approved by the commissioner to address groundwater nitrate problems," but in fact no specific practices are described in the referenced part. Instead, the referenced subpart merely indicates that the commissioner will post a list. Based on the SONAR, the AMTs are intended to "go beyond the nitrogen fertilizer BMPs" and could be identified by the local advisory teams, and could include a variety of management

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40 Minn. Stat. § 103H.005, subd. 15.
41 Minn. Stat. § 103H.275, subd. 2(c).
42 Minn. Stat. § 103H.275, subds. 1-2.
43 Minn. Stat. § 103H.275, subd. 2(a).
44 In addition to the Act, Minn. Stat. § 116D.02 makes clear that economic impacts are not more important than the value of preserving natural resources for future generations.

practices. Because the commissioner may allow these practices to substitute for nitrogen fertilizer best management practices,[45] the rule must define all the practices that would be approvable AMTs and establish a standard for new practices that might not be currently known. As currently drafted, the rule is too vague and provides too much unfettered discretion to the commissioner in allowing the unknown AMTs to substitute for mandated best management practices.

Needed definition: Health Risk Limit or HRL. The definitions should reference the particular health risk limit at Minn. Stat. § 103H.201 because this term is used throughout the rule and has a particular meaning.

Needed definition: Interested Person. To simplify references to public notice procedures, MCEA recommends that the commissioner define "interested persons" as those who have registered with the department to receive public notices concerning actions of the commissioner under the rule.

1573.0010, subp. 12. The definition of lag time is limited to areas "being monitored." The definition is too restrictive. Areas that have been monitored in the past will have an established lag time. It is unclear who is performing the monitoring referenced in this definition. Lag time should be defined to include all areas where data is adequate to support a determination of how long it takes for nitrogen fertilizer applied at the surface to enter the groundwater.

1573.0010, subp. 14. The rule must establish a process by which members of a "local advisory team" ("LAT") are "approved" by the commissioner and the definition should reference that process, or the rule should establish that the LAT must have a certain constitution, but does not require "approval" by the commissioner. The rule must better define the role of the LAT.

1573.0010, subp. 17. For the purpose of this rule, it does not make sense to use additional concepts from Minn. Stat. § 18C.215, which is a chapter designed for the regulation and control of the manufacture, distribution, and sale of fertilizer in this state. The intent of this rule is to ensure that the MDA can regulate agricultural practices that are leading to excess nitrate levels, and the definition of nitrogen fertilizer must reflect all fertilizers that are applied to supply nitrogen. The MDA should amend this definition to simply reference the statutory definition.

1573.0010, subp. 18. Subpart 18 defines a "public well" as a "community water system" which includes permanent (but not necessarily municipal) water supplies. MCEA supports this definition, but notes that the definitions of municipal public water supply well, and public well, as used in the rule, create confusion. The rule should cover all drinking water supply management areas that have been established to protect public water supplies, whether municipal or non-municipal. There is no basis under this rule for a distinction.

1573.0010, subp. 19. It is unclear why this definition restricts soil tests to those conducted by or under the direction of the commissioner within a drinking water supply management area. Residual soil nitrate tests should include any tests conducted under appropriate controls in any area by any person. MCEA recommends striking the phrase "conducted by or under the direction

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[45] *See* Minn. R. 7040.0040, subp. 6 (evaluation of BMP adoption as part of determination of whether a "level 2" mitigation area continued).

of the commissioner" from this definition. The phrase "that are representative" will prevent non-standard test results from being considered. MCEA recommends that MDA reference a standard method of obtaining results from soil testing.

B. FALL AND FROZEN SOILS VULNERABLE AREAS BAN (1573.0030):

This part of the rule establishes a ban on application of nitrogen in areas with vulnerable soils in the fall and when there are frozen soil conditions. However, part 1573.0030, subp. 2 and subp. 3 establishes numerous exclusions and exceptions that undermine the intent of the ban. MCEA supports the ban, but does not agree with the language that allows the commissioner excessive discretion.

1573.0030, subp. 1. The proposed provision contains an odd wording. A DWSMA is not "from" a municipal public water supply well. The rule should state that the water supply management area is "established for" a public water supply well. Item A (3)(b) needs to worded in a similar fashion, i.e., reference that it is a drinking water supply management area established for a public water supply well with (or "which has had") nitrate-nitrogen levels greater than or equal to 5.4 mg/L at any point in the previous ten years. DWSMAs are established for public wells that are not municipal. MCEA believes that all public wells should be included.

1573.0030, subp. 1, Item C. Item C indicates that a responsible party in charge of cropland depicted on the commissioner's map is subject to the prohibition on fall application that is stated in part A. This sets up a potential conflict between the criteria in part A and duty to comply with the map in part C. It is important that the map not undermine the prohibition in part 1573.0030, Subp. 1, Item A. If Item A says "a responsible person shall not," then Item C, which states that "any responsible person is subject to Item A," is not needed.

1573.0030, subp. 2. Exclusions.

In general, this section of the proposed rule is drafted in a convoluted manner that makes it difficult to understand. However, closely read, the "exclusion" section appears to remove a significant portion of the vulnerable and DWSMA areas[46] subject to the prohibition on fall application based on certain broad soil ("leaching index") and climactic ("frost-free") assumptions. In Item G, the proposed rule also authorizes the commissioner to allow, based on unstated criteria and without any process whatsoever, fall applications in areas within a high-reading DWSMA if the commissioner believes "that the area is not contributing significantly to the contamination of the well" in the drinking water supply management area. Thus, the overall impact of Subpart 2 is to undermine the protection provided by prohibiting fall application of nitrogen fertilizer in vulnerable areas and threatened drinking water supply management areas.

The "exclusions" allow fall application of nitrogen fertilizer based on frost-free dates "in the county or a portion of the county" and a "leaching index" of various levels.[47] Later, however, the proposed rules state (Item B) that the exclusion applies to the entire county if a condition is represented on 50 percent or more of the land area of the county, but (Item C) commissioner can

[46] MCEA notes that MDA has proposed to correct this section to include DWSMA areas.
[47] The proposed rule states that the "leaching index" is "determined by the commissioner," but the definition of "leaching index" references the gridMet dataset for 1981-2010.

also subdivide a county by geographical boundary “if there is a clear change in conditions represented in a specific area of the county,” but there is no description of what this “clear change in conditions” might be, or how the commissioner will make this determination or announce this determination. Finally, as noted above, the proposed rule appears to limit the exclusions to areas that are not drinking water supply management areas “with nitrate-nitrogen levels greater than or equal to 5.4 mg/L.”[48] It is unclear whether these areas are the same as the areas subject to the fall application prohibition, which are stated to be those with a well having “nitrate-nitrogen levels greater than or equal to 5.4 mg/L at any point in the previous 10 years.” Even so, as previously noted, this “exception to the exclusion” is undermined by Item G, which broadly allows the commissioner to exclude high-reading DWSMAs without any particular criteria for such an exclusion being set forth, nor any process by which the commissioner will exercise this authority.

The SONAR demonstrates that the MDA has proposed these exclusions based on the notion that cooler spring soils, combined with lower leaching indices, would result in reduced risk of groundwater contamination. However, although the MDA documents that it “heard many concerns from farmers in the western and northern parts of the state about the importance of fall nitrogen applications because of the short application window in the spring,”[49] there is little evidence of *scientific support* for the theory advanced by the MDA cited in the SONAR. No peer-reviewed or published articles are cited as support for the two-factor theory. One can only conclude that the MDA put the exclusions into this rule not on the basis of science, but instead because “there are logistical problems such as with an insufficient numbers (sic) tender trucks and spreaders to complete all fertilizer applications in this compressed spring period.”[50]

If the MDA’s theory that cooler spring temperatures and a reduced leaching index is scientifically based, MCEA would support removing areas that have these characteristics from the fall application ban area. However, the language creating the exclusion areas must be clear and not subject to the discretion of the commissioner, as detailed below.

1573.0030, subp. 2, item E. This Item appears intended to exclude non-agricultural counties, but references the wrong “Item A.” The exclusion should be for *subpart 1,* Item A.[51]

1573.0030, subp. 2, items F and G. These are both problematic because they are vague. In Item F, what does it mean for a point source to be “a significant source” of N contamination? In Item G, the rule fails to specify the criteria that the commissioner will use to determine that the area is “not contributing significantly” to the N problem. Both of these exclusions are too vague to be enforceable unless amended. They both allow the commissioner free-rein to determine that an area will not be subject to the fall nitrogen prohibition, without any possibility of review. And

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[48] As above, it is assumed that this reference is to the wells in the drinking water supply management areas.
[49] SONAR p. 97.
[50] SONAR p. 98.
[51] MDA has identified this as a needed change in an errata document published on the MDA website.

such discretion is unnecessary: state law already provides a variance procedure that a person needing relief can use if the application of the rule is unreasonable as applied to the person.[52]

1573.0030, subp. 3. Exceptions.

The MDA asserts that these exceptions are needed because they are a "necessary agricultural practice."[53] MCEA supports the requirement that the fall application allowed by the rules must be consistent with the BMPs or the rates in the Fertilizer Guidelines published by the University of Minnesota Extension.[54] However, in a number of cases, the information presented in the SONAR undermines the assertion that the exceptions are needed as a necessary agricultural practice.

For example, for item 2, the SONAR states that, for pasture fertilization, "an early spring nitrogen application is the recommended timing." The fall application exception is only necessary, apparently, if the producer is seeking a "high yield system," and then only ¼ of the application is to occur in the fall, a limit which is not reflected in the exception.[55] As a result, a reasonable "exception" would be "when nitrogen fertilizer is required for a high yield pasture, provided that only ¼ of the yearly application is made in the fall." Similarly, for item 4, grass seed production, the cited reference indicates that "either a fall application or very early spring application is recommended."[56] As a result, fall application is not a necessary practice. Where fall application is a necessary practice, it should be done by October 1 to get plant root uptake of the nitrogen.

Item C is arbitrary as drafted. The SONAR notes that when farmers are adding phosphorus to fields, it generally is formulated with up to 40 pounds per acre of nitrogen and applied in the fall for use over two seasons. The Item states that "notwithstanding subpart 1" and "in addition to item A" (it is assumed that rule intended to reference *Subpart 2*, Item A), fall application is allowed so long as the applied N rate does not exceed an average of 40 pounds per acre in a field. However, without explanation, the rule then allows more than 40 pounds per acre (without any upper limit whatsoever), if a soil analysis demonstrates that the fields have "low to very low phosphorus levels." Although the SONAR argues that this exception will be temporary, the language in the rule does not reflect any temporal limit. No scientific information is provided to explain what the impact of this exception would be on soil nitrate levels. Because (as noted in the SONAR), there are other methods to increase P where needed, this exception is arbitrary and

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[52] *See* Minn. Stat. §§14.055-.056. For example, a farmer who applies nitrogen in the fall using techniques and equipment that ensure that leaching does not occur might be able to apply to the commissioner for a variance from the fall application ban, on the ground that it is unreasonable under the unique site conditions and techniques being used. The commissioner, in granting such a variance, could agree so long as the farmer continued to use the techniques and documents the results.
[53] SONAR p. 102.
[54] Proposed rule, 1573.0030, Subpart 3, Item B. It would appear that this document is no longer available on the internet, making it difficult to check the references.
[55] SONAR p. 103.
[56] *Id.*

undermines the intent of the rule. Only the first part of the phosphorus-related exception is justified.

C. DRINKING WATER SUPPLY MANAGEMENT AREA; MITIGATION LEVEL DESIGNATION (1573.0040).

This part of the rule establishes the preconditions for the issuance of "water resource protection orders" or "WRPRs." This part provides various duties for the commissioner: establishing mitigation level areas ("MLs"); "determining" BMPs; monitoring; and evaluating. The rule requires no actions by responsible parties until WRPRs are issued. The rule is unreasonable and will not meet the goals of the Groundwater Protection Act where it continues voluntary actions in areas where nitrate levels threaten to exceed the HRL. The rule is defective because it fails to establish a clear deadline for an ML2 to move to a ML3, a level at which the commissioner could issue a WRPR. In particular, MCEA believes that the current rule language, which allows unlimited "evaluation time" for a ML2, is unreasonable and not supported by the record.

MDA has the authority to require, by rule, statewide actions applicable to areas where specific evidence exists of the threat of public (and private) well contamination and should use this authority to establish reasonable conditions, such as recordkeeping, sampling, and nutrient management planning, that apply where a threat has been documented and a "mitigation area" established, *prior to a WRPR being issued.*

It is not reasonable for all sites—even sites where statistical evidence suggests that the HRL will be exceeded—to be classified in the "voluntary" ML1 and ML2 categories. More serious sites—where the HRL has been exceeded or is statistically likely to be exceeded or where a significant number of private wells already exceed the HRLs—must immediately be prioritized for WRPRs. Under Minn. Stat. § 103H.275, the commissioner is required to ensure that the water source protection requirements are "designed to prevent the pollution from exceeding the health risk limits." As currently drafted, this rule fails to meet this standard.

1573.0040, subp. 1.[57] Although subpart 1 notes that the application of the part is "to responsible parties in drinking water supply management area," it would be more accurate to state that this part establishes the procedures that the *commissioner* will use to establish and evaluate mitigation level areas prior to issuance of a water resource protection requirement order. MCEA proposes that requirements for responsible parties in designated mitigation areas prior to the issuance of a WRPR also be included in this section of the rule.

1573.0040, subp. 2. This states that the commissioner will use public well nitrate-nitrogen concentration data provided by the commissioner of health to designate a DWSMA with a "mitigation level." While there is no problem with using data provided by the Department of

[57] As noted above, MCEA finds no support in the record for the commissioner's decision to limit the designation of mitigation levels to DWSMAs, because the decision arbitrarily leaves persons depending on private wells—persons who are more vulnerable to health impacts from nitrate levels with fewer options for addressing the exceedance—without regulatory protection.

Health (and indeed, the MDA should defer to the Minnesota Department of Health), this rule subpart cannot be enforced because it does not provide a deadline for the commissioner to act on the data provided. To address this issue, the rule must provide an action deadline, i.e., 60 days from the date that the Department of Health provides the necessary data.

1573.0040, subp. 3. This section establishes the criteria for "being designated" by the commissioner at a particular "mitigation level."

A ML2 is where, within a rolling 10-year period, (a) based on a "statistical analysis[58]. . .the groundwater. . .is projected to exceed the health risk limit in the next ten years; or (b) a reading has been 8 mg/L or greater. It is unreasonable to classify an area as an ML2 if it is statistically likely to exceed the HRL, or has in fact documented an exceedance of an HRL. Immediate mandatory actions are needed for such sites, i.e., a WRPR, if the statutory goal of Minn. Stat. § 103H.175 to prevent exceedance of the health risk limit is to be achieved. Under the rule as currently proposed, a public well could have had a reading of 12 mg/L nitrate, but still have its associated DWSMA characterized as a "voluntary only" mitigation level 2. This approach is not supported by the record, and does not comply with the Minn. Stat. § 103H.275.

Having established these "voluntary only" mitigation levels, the rule provides that the commissioner can, nevertheless, exclude portions of the affected DWSMA from the ML area. Subpart 3, item B provides that the commissioner "may make exceptions for increasing a mitigation level" for a "nonmunicipal" public supply well based on "significant change" in land use, and "the severity of nitrate" in "other wells" and the "population affected" and "other factors."[59] Item C provides that the commissioner "may exclude" an area if there is a point source "that is…significant" and item D provides that the commissioner "may exclude" a part of a DWSMA from the mitigation level if the commissioner determines that the area is not contributing "significantly" to the contamination. These exclusions are all purely subject to the discretion of the commissioner and fatally vague, and must be eliminated from the proposed rule or amended to remove the vague language and excessive discretion.

1573.0040, subp. 4. Subpart 4 requires the commissioner to "determine" the nitrogen fertilizer BMPs for the affected DWSMA, but this is unnecessary because the BMPs for various areas of the state are well-established.

1573.0040, subp. 5. In subpart 5, the commissioner is required to conduct some form of monitoring, but that monitoring may only be to obtain data from the public well. As the commissioner is already obtaining data from the public well, this part fails to define any new mandated monitoring activities and therefore fails to protect the public. To the extent that this provision was written because of limited resources for monitoring, MCEA proposes that the monitoring criteria include priorities for monitoring.

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[58] The method should be described in the rule.

[59] This provision suffers from the same "substantive due process" defect as the decision to abandon private wells from protection under the rule: it provides lesser protection to smaller public well user groups based on the argument that MDA needs to prioritize work in other areas.

1573.0040, subp. 6. In subpart 6, the commissioner is required to conduct an evaluation of the ML2 to determine whether the BMPs have been implemented. There is no time limit on the commissioner to conclude this evaluation, but only a minimum time (3 years) that the commissioner must allow for evaluation. In general, voluntary implementation of BMPs has not protected the groundwater from nitrate contamination, and should not be continued under this rule. MCEA believes that BMP implementation is not a valid criterion on which to base continuous voluntary action, particularly when a significant percentage (20 percent) of responsible parties are not counted, the criteria for determining BMP compliance are not clearly stated, and the time and resources needed to accomplish this survey has not been justified. At any rate, it is manifestly unreasonable for the rule to allow evaluation of compliance for an unlimited period of time. The rule must establish a firm limit for the time that the commissioner can take to evaluate BMP compliance. Given the prolonged period of time that BMPs have been the subject of outreach to agricultural communities, this time should be short.

1573.0040, subp. 7. Subpart 7 is important, because it describes how the commissioner can redesignate a ML2 (where nothing is required) to a ML3 (where a WRPR can be issued).

Item A. This item suffers from the same defect as subpart 6: no limit is put on the time during which the commissioner will evaluate ML2 designation. The length of the allowed evaluation period is "no fewer than three growing seasons" or "the lag time"—whichever is longer.[60] This means that the commissioner could "evaluate" for an unlimited amount of time. If BMP compliance is maintained as part of this rule, it must be changed to provide a firm end-date for the evaluation period, such as 3 years. This period should be adequate for the commissioner to determine whether the BMPs have been implemented, and whether they are having an impact.

Item B. MCEA does not support item B, which allows a ML2 to become and ML1. Once the criteria for an ML2 have been met, the ML2 should not be redesignated as a lower-priority ML1, as that may allow the conditions under which the nitrate contamination developed to re-occur. MCEA supports adding mandated actions for responsible parties once a ML has been designated. For example, at a ML2, MCEA believes that responsible parties should conduct soil sampling. This soil testing requirement is reasonable because it has been implemented in Nebraska for many years, is not burdensome and is likely in use where a crop consultant is employed, and (where manure is used) can be combined with required testing under MPCA's rules. It is reasonable for the responsible parties and the commissioner to collect this data to ensure that actions that are being taken are having a positive effect, and to be able to better determine where additional resources and actions may be necessary.[61] The SONAR also notes that "Canadian researchers have used nationwide residual soil nitrate information from shallow sampling over time to make policy decision related to fertilizer use efficiencies and groundwater implications (Yang et al., 2007; Drur et al., 2007)." *Id.* The SONAR rejects the idea of requiring testing on the basis of unstated "cost" and because "this testing requires access to a large number of acres."[62]

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[60] MCEA notes that the proposed rule also states that, "however," if residual soil nitrate testing is conduced, the review period shall not be less than three growing seasons. As the word "however" seems to be wrong in this context because nothing is changed, MCEA wonders if MDA meant to propose that the review period would "not be *more* than three growing seasons."
[61] *See* SONAR pp. 122-4
[62] *Id.*

However, if the producers are doing the testing themselves, no access is needed. The unstated cost cannot be unreasonable given that the requirement is one of longstanding in Nebraska. Other state rules require regular soil testing without compensation.[63] The BMPs recommend use of soil nitrate tests in a number of cases.[64] Testimony at the St. Cloud rulemaking hearing supports that producers are testing their soils voluntarily. Similarly, responsible parties in an ML3 area should prepare nutrient management plans in accordance with National Resources Conservation Service Practice Nutrient Management guidelines.[65]

Items C-E. Items C-E establish criteria for moving a well from a ML2 to a ML3. MCEA does not support item C, which appears to allow the area to remain a ML2 so long as 80 percent of the responsible parties are in compliance with the BMPs, even if the statistical analysis still demonstrates that exceedance of the HRL is probable. Item D provides that the commissioner "shall" move to a ML3 if the net residual nitrate in soil below the root zone is increasing "after not less than 3 growing seasons." MCEA cannot support this criterion, because there is no limit on the number of growing seasons that could be considered, but could support this criterion if the evaluation was required after 3 years. Item E provides that the commissioner "shall" move to a ML3 "if the statistical analysis indicates the nitrate-nitrogen concentration is increasing for the public well or groundwater monitoring network." MCEA supports this criterion, provided this evaluation is not viewed as being limited by the time criterion stated in Item A.

Item G. This item allows the commissioner to "grant a onetime exemption" from the move to ML3 on the vague criteria that "responsible parties...have demonstrated progress." Because there are no criteria for "demonstrating progress," MCEA does not support granting the commissioner this authority.

Item H. MCEA does support item H, which allows the commissioner to "make exceptions for increasing a mitigation level designation if there has been a significant change in land use." Because what is "significant" is not defined, this criterion is fatally vague and should be eliminated.

1573.0040, subp. 8. Subpart 8 suffers from many of the same defects as subpart 7, in particular the language allowing the commissioner an unlimited period in which to evaluate whether a ML3 should be redesignated as a ML4. MCEA refers the ALJ to its comments on subpart 7.

1573.0040, subp. 9. Subpart 9 describes how ML4 area can be redesignated as a ML 3 area, if the water will not exceed the HRL in 10 years based on statistical analysis, and no three samples have reached or exceeded 9.0 mg/L. As noted above, MCEA does not believe that it is appropriate for an area that has demonstrated the potential to exceed the HRL to "drop back" to a level of lessor protection that may allow the prior conditions to re-occur.

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[63] *See* Minn. R. 7020.2225, Subp. 3, Item C (phosphorus).
[64] *See*, e.g., sugarbeet production.
[65] Available at https://www.nrcs.usda.gov/wps/portal/nrcs/detail/mn/technical/ecoscience/nutrient/?cid=nrcseprd1369002 (last visited August 14, 2018).

1573.0040, subp. 10. MCEA does not support the artificial and unsupported limit stated in Subpart 10, which limits the move to one ML. If an area should suffer a sudden increase in nitrate levels, there is no reason for the rule to limit the authority of the commissioner not to take action as required by the Groundwater Protection Act.

D. WATER RESOURCE PROTECTION ORDER PROCESS (1573.0050):

Part 1573.0050 establishes the requirement for the commissioner to issue a WRPR, but does not provide adequate due process or standards for WRPR development.

1573.0050, subp. 1 requires the commissioner to issue a WRPR to responsible parties in ML3 and ML4 areas, but does not establish any deadline or any standard that must be met. As a result, there is no stated basis on which the order can be challenged or reviewed, except broadly as not meeting the requirements of the statute.

Item A. Item A notes that the commissioner will issue WRPRs based on the monitoring in part 1573.0040, subp. 5, but, as discussed above, this provision does not require the commissioner to do any monitoring as currently drafted.

Item B. Item B requires the WRPR to apply to the "entire" DWSMA—but only if a groundwater monitoring well network is installed or residual soil nitrate testing is conducted. As noted above, such testing is not mandated. As a result, the commissioner's authority to issue a WRPR to the entire DWSMA is likely quite limited and will not achieve the statutory mandate of Minn. Stat. § 103H.275 to prevent exceedances of the health risk limit.

Item C. This item includes another unnecessary and complicating limitation on the scope of the WRPR that can be issued. If the commissioner has not installed a groundwater monitoring network,[66] subpart 1, item C, limits the scope of the WRPR based on estimated lag time and travel time.[67] Again, the WRPR will not necessarily apply even to the whole DWSMA established by the Commissioner of Health. MCEA objects to this unreasonable limitation on the commissioner's authority.

Item D. This item prioritizes the issuance of WRPRs. It is reasonable for the commissioner to establish criteria for prioritization, but these criteria could be expanded.

Item E. Item D states what must be included in a WRPR, but isn't specific other than including "the water resource protection requirements."[68] For a meaningful order, there needs to be

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[66] Although the commissioner is required by part 1573.0040, Subp. 5 to monitor a DWSMA, the commissioner is **not** required to install a groundwater monitoring network. Thus, it is impossible to predict how many DWSMAs will be fully subject to the WRPR, once issued.

[67] The process by which the commissioner will make the determination is vaguely described in part 1573.0050, Subp. 1, Item C. As a DWSMA is generally based on the 10-year travel time to the protected well, it is unclear why the commissioner here would choose a different area to protect, and this provision therefore introduces unnecessary complication into the process. *See* Minn. R. 4720.5510.

[68] These requirements are evidently intended to be the requirements in part 1573.0060, but those requirements are only to maintain and provide upon request the field-specific records documenting nitrogen fertilizer use, to comply with the already applicable fall application and

language (at a minimum) such as "the commissioner's order must include water resource protection requirements that are necessary to ensure that pollution is minimized to the extent practicable and to prevent the pollution from exceeding the health risk limits." Even better, MDA should establish that each WRPR must include basic items, such as mandated practices, monitoring, recordkeeping, and reporting requirements, to be adequate.

Item F. Item F is unnecessary and redundant with Item A.

Item G. Item G is vague and cannot be enforced because no standards are established under which the commissioner will determine than an "area is not contributing significantly to the contamination in the well or that it is not practicable to include that part." As a result, it should simply be eliminated from the proposed rule.

1573.0050, subp. 2. This subpart addresses notice that will be given regarding the WRPR, but lacks properly articulated due process.

Item A. This item requires the commissioner to hold "at least one" public information meeting in the county affected by the proposed MRPR before it is published. Normally, a proposed permit, environmental review document, or other administrative action would first be published so that the public attending the meeting have an opportunity to review and raise questions that are meaningful. Subpart 2 should be amended to require the public informational meeting(s) to be held during the public comment period following publication of the proposed WRPR notice. The rules should specify how the commissioner will conduct the public informational meeting, particularly if the commissioner decides to use the public informational meeting as a forum for receipt of comments on the rule in lieu of or in addition to the right to request a contested case hearing under the Minnesota Administrative Procedures Act. The rule should provide that the commissioner must include a record of comments and responses to all substantive comments received during the public informational meeting when the final WRPR is issued as part of the findings on the WPRP.

Item B. This item deals with notice. It should be amended to specify that the commissioner must provide a copy of the proposed order, proposed findings, and a technical support document explaining its terms and conditions, to the "affected parties" who must include persons who are drinking the water that is threatened with nitrate contamination. This is reasonable because other agencies (i.e., the MPCA) typically provide fact sheets or technical support documents in support of their proposed actions.[69]

1573.0050, subp. 3 addresses contested case hearings.

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frozen soils prohibitions, and "comply with any water resource management requirements orders that apply to the drinking water supply management area governing the cropland over which the responsible party has control" which adds nothing and is circular in the extreme. In proposed part 1573.0070, the rule lists only content that the commissioner "shall consider." Alternative management practices can only be mandated if they are "funded" meaning that a responsible party does not bear the cost of compliance.

[69] *See, e.g.,* Minn. R. 7001.0100.

Item A. This item should be amended to provide that "any person or entity subject to the water resources protection requirements order *or affected by the water resource protection requirements order*" can petition for a contested case hearing. It is necessary to include affected persons (i.e., persons who depend on the water supply) to ensure that the persons who are supposed to be protected by the rule can exercise their rights if the commissioner's order is deficient.

Item C. This item requires the commissioner to order a "public hearing" if one is requested. A "public hearing" is not the same as a "contested case hearing." In the SONAR, MDA states that the process that it intends to follow was based on that used to create the "public waters inventory." It is unlikely that MDA has correctly selected the necessary due process, because the public waters inventory did not create any new requirements on the owners of the listed waters. The public waters inventory simply created a record of which waters were or were not public waters based on existing statutory criteria, and did not impose new requirements.[70] Furthermore, the proposed rule does not, in fact, set forth or follow the procedures that were used to adopt the public waters inventory, which involved county review and approval and special hearing teams.[71]

MCEA recommends that the commissioner create a "two option" process for receiving comments and recommendations on the proposed WRPR. The first process would be informal: holding a public informational meeting where members of the public could testify before department representatives who would then have to draft a formal "response to comments" document as part of the WRPR findings. The second process would be formal: holding a contesting case hearing under chapter 14 rules if the criteria for requesting a formal hearing are met.[72] Minn. Stat. § 14.57 provides that, unless otherwise provided by law, "an agency shall decide a contested case only in accordance with the contested case procedures of the Administrative Procedure Act." As there is no other law establishing a separate procedure, MDA must order any "contested cases" as provided under Chapter 14.

1573.0050, subp. 5. This subpart appears to allow amendments to the WRPR just with notice and comment. MCEA does not object to this process, provided that the final amended order is subject to judicial review as a final agency order. MCEA proposes that the commissioner have the duty to review and amend issued WRPRs on a 5 year basis to ensure that the terms are having the desired impact on nitrogen levels.

1573.0050, subp. 6. This subpart allows "any person subject to a final . . .order or amended order to seek judicial review." This provision suffers from the defect that it limits review only to those persons "subject to" orders, which (MCEA assumes) means that only the responsible person can appeal. Minn. Stat. § 103H.275 does not limit rights to persons "subject to" orders,

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[70] *See* Minn. Laws 1979, ch. 199, § 7 (required DNR publication, county board review, DNR notice to counties of accepting or rejecting county recommendations, publication of final listings, process by which "any person" or county could challenge the designation of specific waters as public waters, publication of final listing).

[71] *Id.*

[72] MCEA recommends that MDA use the criteria employed by other state agencies for ordering contested case hearings. *See, e.g.,* Minn. R. 7000.1900 (MPCA); Minn. Stat. § 93.483, subd. 3 (DNR mining permit).

but instead refers to "persons affected by the rule and order of the commissioner."[73] The rule must be clarified to ensure that any affected party (i.e., party that can establish standing and who has participated in administrative proceedings) can appeal an order. The rule also fails to specify how a party can obtain judicial review. Is the judicial review provided under the Minnesota Administrative Procedures Act for a "contested case" (Minn. Stat. § 14.63), which provides that an appeal must be filed in 30 days, or would review be provided under the "generic" certiorari statute, Minn. Stat. ch. 606, which provides for 60 days in which to seek review? If MDA intends that review be under the Minnesota Administrative Procedures Act, then a hearing under that act must be offered.

1573.0050, subp. 7. This provision requires the commissioner to record all final WRPRs. MCEA respectfully suggests that MDA ascertain whether this is possible, and what the effect of a "blanket" recording would be.

E. REQUIREMENTS FOR RESPONSIBLE PARTIES SUBJECT TO WRPRs (1573.0060-90).

In this part, the proposed rule establishes certain requirements for responsible parties subject to WRPRs, such as recordkeeping. Above, MCEA has proposed to include certain of these requirements (such as recordkeeping) when mitigation levels are established, and does not agree with limiting these requirements to parties that are subject to a WRPR. If MCEA's proposal is accepted, this part is needed only to specify what records must be kept and for how long, and to provide conditions on access consistent with MDA's statutory authority.

1573.0060. This provision requires a responsible party in a mitigation level 3 or 4 area to maintain field-specific records "starting with the effective date of the water resource protection requirements order." As noted above, it is unreasonable to wait to require such record-keeping until a WRPR is issued as this is a low-impact requirement that producers should be using under the BMPs to monitor their nutrient use. Item A(3) requires compliance with the fall application prohibition, but this would already be required for these producers if the DWSMA protected well has had a reading over 5.4 mg/L, which would be the case for ML3 and 4 areas receiving a WRPR, so it adds nothing and could be confusing, causing persons subject to the "part 1" fall application ban to believe that nothing is required until a WRPR is issued.

1573.0070, subp. 1. This section requires the commissioner "to consider" including the listed requirements in a WRPR. As a result, the content of the order is not cabined in any way by this rule. Under these circumstances, only the due process related to the draft order will allow parties to challenge the content of the order, but this due process is deficient as noted above. MCEA supports making certain of these content requirements mandatory with any order, i.e., field testing, monitoring, crediting of all nutrient sources, nutrient management plans, and the use of alternative management tools that the commissioners specifically finds are necessary to reduce soil nitrogen-nitrate levels in the area subject to the WRPR.

[73] Minn. Stat. § 103H.275, subd. 2(d).

MCEA is deeply troubled by the limit posed by subpart 1, item B. Item B limits the commissioner’s ability to impose alternative management tools by stated that such tools can only be mandated as part of an order “provided a source of funding for increased costs related to the implementation of the alternative management tool is available to responsible parties.” This is arbitrary and will thwart achievement of the goals of the Groundwater Protection Act. Other parties required to protect public resources (for example, those who are regulated under air, water or solid waste permits issued by the MPCA) must internalize the cost of compliance, and are not allowed to avoid compliance unless government money pays for it. In other regulatory programs, if a regulated party finds that the cost of compliance is unreasonable, the regulated party has the burden of seeking relief.[74] The same process should be applied to agricultural producers, especially where there are numerous sources of public funds available to defray the cost of compliance.[75] Compliance should not be limited to funded activities unless the cost of compliance would present a hardship, and then only if reasonable conditions are established in a schedule of compliance to ensure that any damage caused by the delay is limited. The proposed rule does not require any showing of hardship, and therefore is unreasonable. The prohibition on requiring AMTs, the very practices that the MDA has acknowledged will be necessary to achieve the HRL in vulnerable areas, unless funding is provided, must be removed from the rule because it is contrary to the goals of the Groundwater Protection Act. If MDA wants to provide some limited time for a responsible party to obtain funding necessary to comply with the AMTs, a schedule of compliance process could be included in the part of the rule addressing WPRPs, limited to agreements with the commissioner lasting no longer than two years. This should be adequate to address temporary situations resulting from weather events and temporary financial situations affecting a particular responsible party.[76]

1573.0070, subp. 2. This subpart addresses requirements for mitigation level 4. In the SONAR, the MDA states that in mitigation level 4, “alternative management practices that meet the requirements listed under Minn. Stat. § 103H.275, subd. 2(a) shall be considered for inclusion in a water resource protection requirements order regardless of whether or not funding is available” but this authority is not found in the rule. If ML4 area regulated parties can be mandated in a WRPR to use alternative management tools, it should be expressly stated. The cost of compliance should not be the deciding factor in determining whether a management practice should be imposed. Cost is but one factor that should be considered under the statute.[77] Item B in this section limits the commissioner’s authority to require fertilizer application rates that are less than the recommended rate set by the University of Minnesota. Fertilizer application rates are set to ensure the maximum harvest level, not to protect groundwater. As the purpose of the WRPR is to protect groundwater, the commissioner must have the authority to require application rates

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[74] *See, e.g.,* Minn. R. 7000.7000.
[75] The various funding opportunities are listed on MDA’s website and the website of the Board of Water and Soil Resources.
[76] For other parts of the rule, MCEA notes that state law already contains a variance process, which could be utilized by responsible parties. The proposed rule might be amended to include a reference to that process. See Minn. Stat. §§ 14.055-.056.
[77] Minn. Stat. § 103H.275, subd. 2.

that are less than recommended if the particular circumstances of the situation make such reduced rates reasonable.

1573.0070, subp. 2, item C. Subpart 2, item C, prohibits the commissioner from restricting the selection of the “primary crop.” The term “primary crop” is undefined. It is unclear whether this term means that the commissioner is prohibited from requiring, as an alternative management tool, the inclusion of a nitrogen-reducing crop in a rotation, and thus is fatally vague. To achieve the goal of the Act, the commissioner must have the authority to require, if circumstances demand, that extremely vulnerable acres not be planted with crops that contaminate drinking water supplies, or that a different crop be added into a crop rotation, such as alfalfa or grasses, that would quickly reduce soil nitrate levels.[78] To eliminate the commissioner’s authority to require a technique that is well-established as a method to reduce soil nitrogen-nitrate levels is arbitrary.

1573.0070, subp. 3. Subpart 3 provides the commissioner, with unlimited discretion, the authority to provide exemptions to a WRPR “on a site-specific basis.” There is no description whatsoever of how this process would be made public or controlled. As a result, this provision is fatally vague. Instead, the commissioner should establish a fair temporary schedule of compliance process whereby particular conditions that create hardship, on a site-specific basis, can be fairly evaluated and addressed in a controlled fashion.

1573.0080. This rule provides that a responsible party who is certified through the Minnesota Agricultural Water Quality Certification Program (“MAWQCP”) is “deemed to be in compliance” with this chapter. MDA’s rule proposal requires the Department to presume that land certified under the MAWQCP is cropland where the nitrogen fertilizer use recommendations have been fully implemented. However, the MAWQCP does not require certified farms to either meet these recommendations, or implement any other practices that reduce nitrate contamination in groundwater.[79] Unless MDA provides evidence that a certified farm has implemented the nitrogen fertilizer use recommendations, this presumption is not justified.

1573.0090. subp. 1. This subpart requires the commissioner to maintain a list of alternative management tools (“AMT”) on the MDA website, and to note if the tool can be substituted for a nitrogen fertilizer best management practice. No standard is provided for when this substitution is to be authorized, making this rule fatally vague. The commissioner should, *in this rule*, list the alternative management tools and which AMTs can be substituted for specific BMPs or amend the rule to provide a more functional definition of AMT.

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[78] *See* De Haan et al, *Residual soil nitrate content and profitability of five cropping systems in northwest Iowa*, PLOS One, March 1, 2017; 12(3); e0171994, available at https://www.ncbi.nlm.nih.gov/pmc/articles/PMC5332022/. *See also* Comment of Dr. Gyles Randall, August 1, 2018.

[79] *See Minnesota Agricultural Certainty Program: Is It Working for Water Quality, An Assessment of Minnesota’s Agricultural Water Quality Certification Program*, MCEA, December 2015.

Item C. Item C allows a responsible party subject to a WRPR to implement an AMT if the commissioner's list allows it, subject only to keeping records of all AMTs used "and the specific water resource protection requirements order that allows the alternative management tool to be used." This is reasonable if the only time an AMT is allowed to substitute for a BMP is under the control of a WRPR, but the rule is not clear.

1573.0090, subp. 2. This subpart allows a person who is subject to a WRPR to apply to the commissioner for an alternative protection requirement pursuant to statute. However, the rule fails to establish any due process concerning how such a substitution will be approved, and is therefore deficient. MCEA suggests requiring such alternative protection requirements to be proposed during the comment period on the WRPR.

IV. CONCLUSION

MCEA supports the need for a rule to prevent and mitigate nitrate pollution in groundwater. The instant rule falls short of what is needed and what Minn. Stat. § 103H.275 demands, in particular because it offers little protection to persons who get their drinking water from private wells, and because it continues to lean on BMPs to reduce nitrate levels despite the fact that BMPs have not succeeded in reducing nitrate levels to date. In order to be approved, the rule must be amended to eliminate vague and unenforceable language and the rule must ensure that groundwater is protected and that the HRL is not exceeded. Finally, where the rule is to be used as the basis for issuance of an order, it must include adequate standards and procedures to ensure that all affected parties have an opportunity to seek meaningful relief, and should not prevent the commissioner from requiring reasonable agricultural practices that reduce soil nitrate/nitrogen levels.

Sincerely,

/s/Ann Cohen
Ann Cohen
Staff Attorney
Minnesota Center for Environmental Advocacy

Exhibit B

Clean Water Organizations'
Comments on the Proposed 2021 NPDES General Permit
for Concentrated Animal Feeding Operations

July 23, 2020

INTRODUCTION

Nitrate pollution from manure and commercial fertilizer is a serious problem in Minnesota. Despite laws intended to limit manure application, nitrate pollution from excess manure continues to contaminate drinking water and degrade aquatic habitats. Minnesota Center for Environmental Advocacy,[1] Friends of the Mississippi River,[2] Minnesota Well Owners Organization,[3] and Sierra Club North Star Chapter[4] (collectively, "Clean Water Organizations") have concluded that the

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[1] Minnesota Center for Environmental Advocacy ("MCEA") is a Minnesota non-profit organization that defends every aspect of Minnesota's environment, relying upon facts, science, and the law. For nearly half a century, MCEA has worked with community members, decision makers, and other partners to protect Minnesota's natural resources and the health and wellbeing of all the state's citizens. As a public interest organization, MCEA works to ensure that Minnesota's bedrock environmental laws are enforced and defended. It has a particular interest in water quality, and it has engaged in legislative and administrative advocacy, rulemaking and permitting proceedings, and litigation to protect Minnesota's water quality.

[2] Friends of the Mississippi River ("FMR") is a nonprofit established in 1993 to engage Minnesotans to protect, restore, and enhance the Mississippi River and its watershed in the Twin Cities Metro area. As part of its efforts to protect and preserve a clean Mississippi River, FMR works with 2,500 members, 2,000 advocates, and over 5,000 volunteers yearly. A major part of FMR's work is focused on watershed protection for the Mississippi River, including preserving water quality by advocating for land use policies and practices that will lead to cleaner water throughout the entire watershed.

[3] Minnesota Well Owners Organization ("MNWOO") is a nonprofit organization for private well owners that works to preserve, protect, and restore Minnesota's water resources and to ensure the safety of those who use private wells for drinking water. MNWOO also provides education, technical and legal services, and advocacy for private well owners. MNWOO works to protect the water quality of the 1.2 million private wells in Minnesota, more than 10% of which are contaminated at levels above allowed health risk limits. This includes many private wells with elevated levels of nitrates.

[4] The Sierra Club North Star Chapter ("SCNS") is a nonprofit organization that is the Minnesota branch of the national Sierra Club, America's oldest, largest, and most influential grassroots environmental organization. SCNS works through grassroots political action, including its 80,000 members, to strategically address Minnesotans' most pressing environmental issues. One of SCNS's priorities in its water program is fighting agricultural pollution in Minnesota, including nitrate pollution.

newly proposed National Pollutant Discharge Elimination System ("NPDES") General Permit for Concentrated Animal Feeding Operations ("Proposed General Permit") drafted by the Minnesota Pollution Control Agency ("MPCA") does not effectively address this problem or follow Minnesota's laws regarding land application of manure. Unless MPCA revises the Proposed General Permit to better reflect the protective standards of the law, Minnesota's water quality is likely to worsen during the permit's tenure.

Since the MPCA issued the 2016 NPDES General Permit for Concentrated Animal Feeding Operations ("2016 General Permit"), Minnesota's nitrate pollution problem has intensified. The drinking water for nearly half a million Minnesotans is now tainted with elevated levels of nitrates, which can cause cancers and other diseases. Now, MPCA has an opportunity to provide better protections for Minnesota's waters, while ensuring farmers can meet their crops' nitrogen needs, through the Proposed General Permit. Yet, the Proposed General Permit perpetuates the same problems that exist in the 2016 General Permit, which will lead to continued contamination of water needed for drinking, recreation, wildlife, and aquatic habitat. Accordingly, the Clean Water Organizations suggest changes to the Proposed General Permit to ensure the protection of water quality and compliance with Minnesota laws regarding manure application.

Most importantly, the Clean Water Organizations propose that the MPCA revise the Proposed General Permit to limit manure application rates to truly reflect expected crop nitrogen needs. As it did in the 2016 General Permit, the MPCA has referenced recommendations for manure application based on maximizing the economic return for farmers, not on the actual plant needs for nitrogen. These recommendations are inconsistent with the governing rules for land application of manure and have led to over-application by many farmers. MPCA must amend the Proposed General Permit to ensure that the referenced recommendations are consistent with the rule's requirements. In addition, the Clean Water Organizations request that the MPCA revise Proposed General Permit to restore the section from the 2016 General Permit regarding pre-plant testing for nitrates, provide clearer requirements to farmers about determining soil temperatures prior to manure application, strengthen October restrictions on manure application, prohibit application of solid manure in December and January, and require geographic-information-system ("GIS") identification of fields in manure management plans. The Clean Water Organizations ask that MPCA revise the permit to make these changes or grant a contested case hearing so that material issues of fact can be heard by a neutral administrative law judge who can develop the record and present a recommendation to the MPCA.

I. MINNESOTA'S DRINKING WATER AND AQUATIC HABITATS ARE ALREADY POLLUTED WITH DANGEROUS LEVELS OF NITRATES

Minnesota takes great pride in its water. Minnesotans depend on their lakes, rivers, and groundwater as sources of clean, drinkable water and habitats for wildlife. While the "Land of 10,000 Lakes" claims the headwaters of the Mississippi River and other historical, cultural, and economically significant waterways, increasing levels of nitrates, which have profound impacts on aquatic and human life, are threatening the health of many of Minnesota's great waters.

A. Minnesota's Nitrate Pollution Is Worsening.

Nitrate contamination in Minnesota's drinking water systems is getting worse. Data collected by the U.S. Environmental Protection Agency ("EPA") showed that between 1995 and 2018, 63% of Minnesota's 115 community wells experienced growing nitrate contamination, with the southern part of the state experiencing the largest increases.[5] As one example, in the Rock County Rural Water System, located in southwestern corner of the state, 24 of the 107 tests

[5] Envtl. Working Grp., *Nitrate Trends in Minnesota Drinking Water*, https://www.ewg.org/interactive-maps/2020-in-minnesotas-farm-country-nitrate-pollution-of-drinking-water-getting-worse/map/ (last visited July 17, 2020).

collected during this time frame revealed nitrate levels exceeding 10 milligrams per liter ("mg/l"), the "safe for consumption" threshold set by the EPA in 1962.[6] Across the state in Winona County, nitrates in the Utica water system surged between 2016 and 2018.[7] Nitrates also threaten metropolitan area community water supplies. EPA tests collected from the Kjellberg system in Wright County, which serves approximately 1,000 people, revealed nitrate levels greater than 3 m/l in more than half of the 204 tests obtained during the study period.[8] In Hastings, 217 out of 313 tests of its groundwater supply, which serves over 22,000 Minnesotans, showed nitrate concentrations exceeding 5 mg/l.[9]

The Minnesota Department of Health's ("MDH") testing also shows troubling trends for private wells. Prior to 2011, less than 1% of MDH private well tests showed nitrate contamination exceeding 10 mg/l.[10] However, with the exception of 2016, beginning in 2011 and every year thereafter, more than 1% of tested private wells were contaminated with nitrate levels exceeding the federal safe consumption limit.[11]

MPCA data confirms that nitrate levels in Minnesota's surface waters are also increasing. Data collected between 1976 and 2010 reveal that 22 of Minnesota's 32 major rivers shows a statistically significant upward trend in overall nitrate concentrations.[12] These rivers showed increases in nitrate concentrations as much as an astonishing 268% during the 30 to 35 year study

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[6] *Id.*
[7] *Id.*
[8] *Id.*
[9] *Id.*
[10] Minn. Dep't of Health, *Nitrate in Private Wells*, https://data.web.health.state.mn.us/nitrate_wells (last visited July 17, 2020), attached as Ex. 3.
[11] *Id.*
[12] Minn. Pollution Control Agency, *Nitrogen in Minnesota Surface Waters* 150 (2013), *available at* https://www.pca.state.mn.us/sites/default/files/wq-s6-26a.pdf, [hereinafter "Nitrogen in Surface Waters].

period.[13] Most of MPCA's regularly monitored testing sites along the Mississippi River have recorded an explosive growth of nitrate concentrations, with MPCA noting that, except for two specific sites, "nitrate concentrations [in the Mississippi River] have been increasing everywhere downstream of Clearwater at a rate of 1% to 4% per year" in recent years.[14] MPCA monitoring sites on the St. Croix River reflected a 49% growth in nitrate concentration between 1976 and 2004.[15] MPCA data collected from major tributaries similarly shows nitrate concentrations increased in the majority of sampled waterways during the study period, with the greatest recorded growth reaching 207%.[16] And the contaminated Rock County Rural Water System discussed above is a surface water source of drinking water.[17]

B. Nitrate Pollution Poses Dangers For People And Aquatic Life.

This increase in nitrate pollution is a serious problem for Minnesotans, as elevated nitrate levels are hazardous to human health and wreak havoc on aquatic life. Increasing nitrate contamination threatens the health of the nearly 75% of Minnesotans who rely on groundwater for their drinking water.[18] Consuming water contaminated with nitrates is associated with adverse birth outcomes, thyroid disease, neural tube defects, and several cancers.[19] Elevated nitrate levels in drinking water are especially dangerous for infants, pregnant women, and people with certain

[13] *Id.* at 151.
[14] *Id.* at 398.
[15] *Id.* at 177.
[16] *Id.* at 150-51, 53.
[17] Envtl. Working Grp., *supra* note 2.
[18] Minn. Dep't of Agric., *Minnesota Nitrogen Fertilizer Management Plan* 20 (2019), *available at* https://www.mda.state.mn.us/sites/default/files/2019-08/nfmp2015addendedada_0.pdf, attached as Ex. 4.
[19] Alexis Temkin et al., *Exposure-Based Assessment and Economic Valuation of Adverse Birth Outcomes and Cancer Risk Due to Nitrate in United States Drinking Water*, 176 ENVIRONMENTAL RESEARCH 1-2 (2019), *available at* https://www.sciencedirect.com/science/article/pii/S001393511930218X, attached as Ex. 5.

blood disorders, who are at risk of methemoglobinemia, or "blue-baby syndrome," which causes severe oxygen deficiency that, without medical treatment, can lead to death.[20]

The EPA set the current health standard for nitrate in water at 10 mg/l in 1962 largely to protect against blue-baby syndrome. New studies strongly suggest that the current standard does not reflect the present understanding of nitrate associated health risks.[21] According to a recent study by Environmental Working Group ("EWG"), lower levels, even below 5 mg/l, are associated with higher risks of certain cancers and adverse birth outcomes.[22] EWG concluded that nitrate pollution of drinking water at levels far below the legal limit may cause up to 12,594 cases of cancer each year in the United States.[23] This tracks large-scale studies in Spain and Italy, published in 2016, and Denmark, published in 2018, which found statistically significant increases of colorectal cancer risks associated with nitrate levels below 2 mg/l.[24] Minnesota regulators should be exceedingly concerned by these new studies because hundreds of thousands of Minnesotans currently access public water systems contaminated with nitrates exceeding 3 mg/l.[25] Even worse, the data shows that over 150,000 Minnesotans accessed public water systems with nitrate contamination levels exceeding Minnesota's health standard of 10 mg/l.[26] Nitrates also plague private water supplies. Minnesota Department of Agriculture data collected pursuant to its Nitrate Clinic Outreach Program shows that 7.7% of 2,063 private well tests reported nitrate levels

[20] Minn. Dep't of Agric., *supra* note 15, at 7-8.
[21] Minn. Dep't of Health, *Nitrate in Well Water*, https://www.health.state.mn.us/communities /environment/water/wells/waterquality/nitrate.html#:~:text=Safe%20Level,water%20for%20public%20water%20supplies (last visited July 17, 2020), attached as Ex. 6; Sarah Porter & Anne Weir Schechinger, Envtl. Working Grp., *Tap Water for 500,000 Minnesotans Contaminated with Elevated Levels of Nitrate* (Jan. 14, 2020), attached as Ex. 7.
[22] Temkin et al., *supra* note 16, at 11; Porter & Schechinger, *supra* note 18.
[23] Porter & Schechinger, *supra* note 18.
[24] *Id.*
[25] Minn. Dep't of Health, *supra* note 18.
[26] Porter & Schechinger, *supra* note 18.

exceeding 10 mg/l.[27] The 2012 data shows an increase in the percentage of private wells exceeding the current standard from samples tested in 2011, suggesting nitrate infiltration into well water supplies throughout Minnesota is an increasing problem.[28] In fact, due to a lack of testing, the number of contaminated wells in Minnesota may actually be much greater.[29]

In addition to impairing drinking water, elevated nitrate concentrations in Minnesota's waterways are significant contributors to aquatic habitat destruction. High nitrate levels in surface waters directly contribute to eutrophication, which stimulates excessive plant growth and depletes oxygen levels in the water, causing harm or death to fish.[30] Nitrate also is directly toxic to fish and other aquatic organisms, causing heart and liver problems, electrolyte imbalance, and increased vulnerability to bacterial and parasitic diseases.[31] Due to nitrate's solubility in water, its ultimate intrusion into the Mississippi River is in part to blame for the hypoxic "dead zone" in the Gulf of Mexico.[32] One study estimates that the 158 million pounds of nitrate that leave Minnesota annually via the Mississippi River has caused nearly $2.4 billion in annual damages to fish stocks and habitat for more than 30 years.[33]

C. Much Of Minnesota's Nitrate Problem Is Caused By Agriculture.

Agriculture is Minnesota's largest contributor to nitrate pollution—specifically, nitrate runoff or leaching from farmland from commercial nitrogen fertilizer or manure. According to the

[27] Minn. Dep't of Agric., *2012 Nitrate Clinic Outreach Summary Report* 2 (2012), *available at* https://www.mda.state.mn.us/sites/default/files/inline-files/2012nitrateclinic.ashx.pdf.
[28] *Id.*
[29] Jennifer Bjorhus, *One in Eight Minnesotans Drink Nitrate-Tainted Tap Water, Report Says*, STAR TRIBUNE (Jan. 14, 2020), *available at* https://www.startribune.com/one-in-eight-minnesotans-drink-nitrate-tainted-water/566960262/.]
[30] Nitrogen in Surface Waters, *supra* note 9, at 43.
[31] *Id.*
[32] *Id.* at 36, 46.
[33] Rebecca Boehm, Union of Concerned Scientists, *Reviving the Dead Zone* 3 (2020), *available at* https://www.ucsusa.org/sites/default/files/2020-06/reviving-the-dead-zone.pdf.

Minnesota Department of Agriculture, approximately 2.7 million tons of inorganic nitrogen are added to Minnesota soils each year, and 80% of that nitrogen is attributable to agriculture.[34] Unfortunately, a significant portion of that nitrogen reaches state waters. In its 2013 study, MPCA estimated that cropland sources account for almost 73% of the statewide nitrate load to streams and lakes in an average year.[35] A "significant" part of this comes from applied manure.[36] Notably, MPCA found that the largest increases in nitrate pollution are clustered in the southern third of the state, where most of Minnesota's confined animal feeding operations are located.[37]

This is unsurprising. Domestic and international studies have long confirmed an association between livestock concentration and a documented degradation in water quality. For example, Iowa watersheds with the highest livestock density had some of the highest stream concentrations of nitrates in the state.[38] In the Chesapeake Bay watershed, for example, land application of manure contributes to elevated ground water nitrate concentrations and suffocating algae blooms.[39] This connection is not new. In the 1960s, nutrient runoff from the Danube River seriously degraded the northwestern Black Sea.[40] Conditions rapidly improved after the fall of communist regimes in the late 1980s precipitated the closure of many large animal farms.[41]

The ease with which nitrate escapes the fields is largely to blame. A significant amount of nitrogen from applied manure is lost through volatilization, runoff, and leaching. The University of Minnesota Extension Service ("Extension Service") estimates that up to 50% of the nitrogen

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[34] Minn. Dep't of Agric., *supra* note 15, at 33-34.
[35] Nitrogen in Surface Waters, *supra* note 9, at 205.
[36] *Id.* at 219.
[37] *Id*. at 295; Minn. Pollution Control Agency, https://resources.gisdata.mn.gov/pub/gdrs/data /pub/us_mn_state_pca/env_feedlots/preview/preview.jpg (last visited July 17, 2020).
[38] Dr. Christopher Jones, *Expert Report* 6 (2020), attached as Ex. 1.
[39] *Id.*
[40] *Id.*
[41] *Id.*

from manure may be lost through these processes.[42] University of Minnesota research indicates that applications of nitrate above the economically optimum nitrogen rate for a specific crop significantly increase the potential for nitrate losses.[43]

Partly to blame for the nitrogen losses is the way manure is applied by farmers and how it is used by plants. Manure contains both organic and inorganic forms of nitrogen.[44] While inorganic nitrogen—in the form of nitrate or ammonium—is available to be used by plants for growth immediately, the organic form is not.[45] Before plants can take up organic nitrogen, it must first be mineralized by microorganisms in the soil to inorganic forms.[46] After this conversion process, however, the inorganic form ammonium can be easily converted into gas and lost into the atmosphere through volatilization, only to cause water pollution when it dissolves in rain and returns to earth.[47] But more significantly, since inorganic nitrates are soluble, they are prone to leaching.[48] Thus, the converted nitrate is highly susceptible to filtering through the soil profile and into the groundwater.[49]

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[42] Univ. of Minn. Extension, *Manure Application Methods and Nitrogen Losses*, (2018), https://extension.umn.edu/manure-land-application/manure-application-methods-and-nitrogen-losses, [hereinafter "Manure Application Methods"], attached as Ex. 8.
[43] Melissa Wilson, Univ. of Minn. Extension, *Guidelines for Manure Application Rates*, https://extension.umn.edu/manure-land-application/manure-application-rates (last visited July 17, 2020), [hereinafter "Guidelines for Manure Application"], attached as Ex. 9.
[44] Melissa Wilson, Univ. of Minn. Extension, *Manure Characteristics*, https://extension.umn.edu/manure-land-application/manure-characteristics (last visited July 17, 2020), [hereinafter "Manure Characteristics"], attached as Ex. 10.
[45] *Id.*; Manure Application Methods, *supra* note 39.
[46] Manure Characteristics, *supra* note 41; Ron Wiederholt, N.D. State Univ. Extension Serv., *Environmental Implications of Excess Fertilizer and Manure on Water Quality* (2017) https://www.ag.ndsu.edu/publications/ environment-natural-resources/environmental-implications-of-excess-fertilizer-and-manure-on-water-quality, attached as Ex. 11.
[47] *Id.*
[48] Scott C. Killpack & Daryl Bucholz, Univ. of Mo. Extension, *Nitrogen in the Environment: Leaching*, https://extension2.missouri.edu/wq262 (last visited July 17, 2020).
[49] Wiederholt, *supra* note 43.

In addition, if a farmer applies manure incorrectly—in too large of quantities, on vulnerable soils, or at improper times—leaching or runoff is more likely. If too much manure is applied, plants do not take it up, allowing nitrates to leach away.[50] If manure is applied to coarse-textured soils, nitrates can sink past plant roots and into groundwater.[51] If manure is applied early in the fall on ground that is too warm, it will quickly convert to into nitrate and likely be lost before spring planting; but if manure is applied in the winter on frozen soils, it is unlikely to be incorporated into the soil and instead runs off during melts or spring rains.[52]

In addition, multiple factors make manure challenging to manage as fertilizer and encourage over-application. First, the nutrient concentration in manure is far lower and much more uncertain than commercial fertilizer.[53] Time windows for effective manure application are narrower than with commercial fertilizer, and farm implements designed to distribute manure to fields can apply material non-uniformly.[54] Nitrogen loss to the atmosphere through volatilization can be significant and difficult to predict.[55] And insufficient storage capacity for manure may lead to farmers applying manure at ineffective times, when it is more likely that nutrients will run off or leach into the water and be lost to plants.[56] These uncertainties may lead farmers to over-apply manure in their eagerness to ensure that plants have abundant sources of nitrogen to use as they grow—or may even cause them to apply manure in the fall followed by commercial fertilizer in

[50] Guidelines for Manure Application, *supra* note 43.
[51] *Id.*
[52] Melissa Wilson, Univ. of Minn. Extension, *Manure Timing*, https://extension.umn.edu/ manure-land-application/manure-timing (last visited July 17, 2020), [hereinafter "Manure Timing'], attached as Exhibit 12.
[53] Jones, *supra* note 35, at 6.
[54] *Id.*
[55] *Id.*
[56] *Id.*

the spring.[57] These factors "frequently result in manured land receiving larger amounts of nutrient than those that receive only commercial N [fertilizer]."[58]

This is not necessarily a problem for the farmer, however. Unlike commercial fertilizer, which must be purchased, farmers with large livestock operations have access to free, always available manure in ample quantities. In some scenarios, research has found maximizing nitrogen loss to the environment is more profitable than attempting to use all of the nutrients from the manure.[59] For these farmers, manure is a waste product, and squandering its nutrients is not necessarily economically wasteful.[60] In fact, because of the costs of hauling manure, farmers may find it more profitable to concentrate manure applications on the fields closest to the animal confinements and buy commercial fertilizer—with its higher, uniform, and known nitrogen content—for the remaining fields.[61]

Overall, for farmers, the economic risk of under-applying manure is far greater than that of over-applying.[62] When a farmer under-applies nitrogen, the farmer takes on a considerable economic risk: that crop growth will not be maximized, leading to lower yields and less product to sell.[63] But when a farmer over-applies nitrogen, the farmer is only taking on the risk of the cost of the additional manure—which in many cases costs nothing at all—while increasing the opportunity to maximize crop yields and product for sale.[64] While the economic risk to the *farmer* of over-application is small, however, the *environmental* risk of over-application is severe.[65] Any

[57] *Id.*
[58] *Id.*
[59] *Id.* at 6.
[60] *Id.*
[61] *Id.*
[62] *Id.* at 8.
[63] *Id.*
[64] *Id.*
[65] *Id.*

excess nitrate not taken up by crops is vulnerable to loss to the atmosphere, aquifers, lakes, and streams.[66] This increases the costs to the public, which takes on the burden of addressing pollution, but does not increase costs to the farmer.[67] Accordingly, over-application of nitrogen "transfers the economic and natural risks associated with nitrogen application from the individual farmer to the public." [68]

Preventing nitrate from reaching water is vital to successfully addressing the growing nitrate pollution problem. Prevention is far less costly than treatment of contaminated water—when treatment is even possible.[69] Accordingly, controlling manure application to prevent nitrate runoff and leaching is critical to protecting public health from still worse increases in nitrate pollution. MPCA must ensure that the Proposed General Permit imposes restrictions that will adequately limit nitrate pollution to protect the people and aquatic habitats of Minnesota.

II. MINNESOTA LAW PLACES LIMITS ON LAND APPLICATION OF MANURE

Because of the harm posed by the threat of nitrate pollution, and the economic incentive of farmers to over-apply nitrogen, MPCA adopted a rule—Minn. R. 7020.2225, subp. 3 ("Land Application Rule")—that imposes limits on the amount of manure that can be applied to fields as fertilizer. The Proposed General Permit must include those limitations.[70]

The Land Application Rule requires that manure application be "limited" so that "the estimated plant available nitrogen from all nitrogen sources does not exceed *expected crop nitrogen needs* for nonlegume crops and *expected nitrogen removal* for legumes."[71] In other words,

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[66] *Id.* at 2.
[67] *Id.* at 8.
[68] *Id.*
[69] Minn. Dep't of Agric., *supra* note 15, at 18, 68.
[70] Minn. R. 7001.1080, subp. 1 (stating that any NPDES permit issued by the MPCA must "contain conditions necessary for the permittee to achieve compliance with all Minnesota or federal statutes or rules").
[71] Minn. R. 7020.2225, subp. 3(A) (emphasis added).

farmers must determine how much nitrogen their crops are expected to need or remove from the soil, how much nitrogen is available to their crops from all sources, and how much manure is needed to make up the difference between the needed nitrogen and available nitrogen. Then farmers must limit their manure application to ensure the application does not provide more nitrogen than the crops "need" or "remove."

To perform this calculation, farmers must first determine "expected crop nitrogen needs," "crop nitrogen removal rates," and "estimated plant available nitrogen." According to the rule, these variables "must be based on the most recent published recommendations of the University of Minnesota Extension Service or of another land grant college in a contiguous state."[72] Farmers must also identify all sources of nitrogen available to their crops, including "commercial fertilizer nitrogen, soil organic matter, irrigation water, legumes grown during previous years, biosolids, process wastewater, and manure applied for the current year and previous years."[73]

The rule provides some flexibility for farmers, however. Once the manure application calculation has been performed, farmers may deviate up to 20% from the Extension Service recommendations "where site nutrient management history, soil conditions, or cool weather warrant additional nitrogen application."[74] And if crop nitrogen deficiencies are "visible" or "measured," farmers may be able to apply even more nitrogen than the extra 20%.[75]

III. THE PROPOSED GENERAL PERMIT SHOULD BE REVISED TO PROTECT WATER QUALITY AND COMPLY WITH MINNESOTA RULES

While the Proposed General Permit includes some positive changes, the draft does not go far enough to protect Minnesota's water quality or comply with the Land Application Rule. Unless

[72] *Id.*, subp. 3(A)(1).
[73] *Id.*, subp. 3(A)(3).
[74] *Id.*, subp. 3(A)(2).
[75] *Id.*

MPCA makes changes, nitrate pollution in Minnesota is likely to worsen during the five-year tenure of the Proposed General Permit. Accordingly, the Clean Water Organizations request MPCA make the following changes to the Proposed General Permit.

A. Section 13.3: Limitation Of Manure Application Rates

First, MPCA must revise the Proposed General Permit to limit rates of manure application so that application is truly restricted to the amount of nitrogen the crop needs, as required by the Land Application Rule. As written, the Proposed General Permit references recommendations from the Extension Service and the MPCA for plant nitrogen needs that are based on economic risk and cost factors that are unrelated to the amount of nitrogen a typical crop will actually need or remove. This is called the Maximum Return to Nitrogen, or MRTN, system. Based on analysis by experts Dr. Gyles Randall, professor emeritus at the University of Minnesota's Department of Soil, Water, and Climate, who has conducted numerous studies relating to plant nitrogen needs and removal; and Dr. Christopher Jones, research engineer at Iowa State University, the MPCA's referenced recommendations are not consistent with the standard established by the Land Application Rule.

1. MRTN is not a measure of expected crop nitrogen needs or expected nitrogen removal.

Under the Land Application Rule, farmers must "limit[]" manure application so that the plant available nitrogen in the soil from all nitrogen sources is no more than "expected crop nitrogen needs" for nonlegumes and "expected nitrogen removal" for legumes.[76] The Land Application Rule states that the "expected crop nitrogen needs" and "expected nitrogen removal" must be based on the most recent published recommendations from the Extension Service (or of

[76] *Id.*, subp. 3(A).

another land grant college in a contiguous state).[77] The Proposed General Permit, accordingly, identifies recommendations from the Extension Service and specifically two fact sheets from MPCA to use in determining "expected crop nitrogen needs" and "expected nitrogen removal."[78] These fact sheets direct users to an Extension Service website, entitled "Calculating Manure Application Rates," which directs users to first "find the nutrient needs of the crop."[79] To do so, users are directed to another Extension Service website, called "Guidelines for Manure Application Rates." This website provides recommendations based on the MRTN system, for example, 195 pounds of nitrogen per acre for corn following corn and 150 pounds of nitrogen per acre for corn following soybeans.[80]

The MRTN referred to in these documents is based on three variables: expected crop price, expected nitrogen source cost, and expected crop production in response to the amount of fertilizer

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[77] *Id.*, subp. (3)(A)(1).

[78] Minn. Pollution Control Agency, Proposed General Permit § 13.3 (2020) [hereinafter "Proposed General Permit"] (directing permit holders to "the most recent recommendations of the Extension Service and the MPCA fact sheets '*Manure Nitrogen Rates For Corn Production (wq-f8-18)*' and '*Manure Management For Corn On Irrigated Sandy Soils (wq-f8-52)*'" (emphasis added)); *see also* Minn. Pollution Control Agency, *Manure Nitrogen Rates for Corn Production (wq-f8-18)* (2019) [hereinafter "Manure Nitrogen Rates for Corn"], attached as Ex. 13; Minn. Pollution Control Agency, *Manure Management For Corn On Irrigated Sandy Soils (wq-f8-52)* (2016), attached as Ex. 14.

[79] Melissa Wilson, Univ. of Minn. Extension, *Calculating Manure Application Rates* (2019), https://extension.umn.edu/manure-land-application/calculating-manure-application-rates, attached as Ex. 15.

[80] Guidelines for Manure Application, *supra* note 40. Concerningly, the MRTN recommendations under the current Extension Service documents are much higher than under previous versions of the recommendations. For example, the 2011 recommendations from Extension Service identify the MRTN at the 0.05 ratio as 155 lb. N/acre for corn after corn, and 120 lb. N/acre for corn after soybeans (and are even lower for less productive soils). It is unclear to MCEA why the recommendations have risen by 25% in both cases: 40 lb. N/acre for corn after corn and 30 lb. N/acre for corn after soybeans. This is a substantial and unexplained change that is almost certain to have significant environmental effects. *See* Univ. of Minn. Extension, *Fertilizer Guidelines for Agronomic Crops in Minnesota* 15 (2011), available at https://conservancy.umn.edu/bitstream/handle/11299/198924/Fertilizer%20Guidelines%20for%20Agronomic%20Crops%20in%20Minnesota.pdf?sequence=1&isAllowed=y.

applied.[81] While the expected crop production is based on research into plant nitrogen needs, the other variables can significantly change the recommended amount of nitrogen farmers should apply.[82] Accordingly, recommendations based on the MRTN system are intended to maximize economic performance for farmers, not simply to provide the crop with the nitrogen it needs to grow.[83]

Specifically, the MRTN calculates a ratio of the cost of commercial nitrogen fertilizer to the expected sale price for that crop. For example, if anhydrous ammonia fertilizer is being sold for $0.30/lb.-N, and the price of corn is $3.00 per bushel, the ratio will be 0.10.[84] This ratio is then used to determine how much nitrogen should be applied to a field to achieve the most *cost-effective* outcome.[85] Plants can only use a certain amount of nitrogen—at some point, plants stop taking in nitrogen from the soil and further application will produce no additional plant growth. However, at a certain point before plants reach this maximum growth, the incremental increase of nitrogen applied to the crop will produce a diminishing return in terms of crop yield.[86] Thus, the cost of adding that extra fertilizer to achieve the smaller potential growth becomes less cost-effective for the farmer.[87] The MRTN identifies the crucial point that produces the *maximum economic return for the farmer*. Beyond that point, the revenue generated from the additional bushels produced by additional fertilizer will (in theory) be less than the cost of the extra fertilizer applied to produce those bushels.[88] But if fertilizer is cheap, the MRTN system recommends additional applications

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[81] *See* Iowa State Univ. Agronomy Extension & Outreach, *Corn Nitrogen Rate Calculator* (2020) [hereinafter "Corn Nitrogen Calculator"], available at http://cnrc.agron.iastate.edu/.
[82] *See id.*
[83] *See* Manure Nitrogen Rates for Corn, *supra* note 75, at 1.
[84] Jones, *supra* note 35, at 5-6.
[85] Corn Nitrogen Rate Calculator, *supra* note 78.
[86] Jones, *supra* note 35, at 5.
[87] *Id.*
[88] *Id.* at 6.

in the hope that additional grain yields will occur, even if plants are unlikely to need the additional nitrogen and nitrogen loss to groundwater is highly likely. For this reason, the MRTN does not strictly focus on the magnitude of the grain yield or the crop needs for nitrogen, but instead on the economic return to the farmer.[89]

The recommendations generated by the MRTN system do not align with the Land Application Rule's requirement that manure application be limited to "expected crop nitrogen needs" for nonlegumes or "expected nitrogen removal" for legumes.[90] Contrary to the rule's language, the recommendations identified by the Proposed General Permit do not in fact define "expected crop nitrogen needs" or "expected nitrogen removal." Instead, they define the *maximum economic return to nitrogen* for farmers. The growth needs of a plant are not the same as a farmer's desire to maximize economic return. Actual crop nitrogen needs are dependent on a number of variables, including the timing, intensity, and total amount of precipitation; amount of sunshine; insect, weed, and disease pressures; other nutrient deficiencies (such as phosphorus, potassium, and sulfur); the amount of soil organic matter (which breaks organic nitrogen down into a form plants can use); and soil type and texture.[91] The MRTN system includes no variables for these factors. Instead the MRTN recommendations are explicitly based on *fertilizer and crop price*, not *crop needs,* and accordingly these recommendations allow manure applications that likely exceed crop needs if it appears the farmer may economically profit.

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[89] *Id.* at 6.
[90] Minn. R. 7020.2225, subp. 3(A).
[91] Jones, *supra* note 35, at 3. Notably, the MPCA fact sheet recognizes that some fields can be highly productive without applying the maximum MRTN, based on different conditions. *See* Manure Nitrogen Rates for Corn, *supra* note 75, at 1. For example, the fact sheet acknowledges that fields in southeastern Minnesota with loess soils need less nitrogen to maximize yields. *Id.* But MPCA provides no recommendation for what the nitrogen level should be in these situations.

Because the section of the Proposed General Permit that identifies the MPCA fact sheets and Extension Service websites is based on the MRTN, the Proposed General Permit is inconsistent with the requirements of the Land Application Rule. The MPCA must adjust the recommendation to reflect the rule's requirement that the application rate must be strictly based on expected crop nitrogen needs and expected nitrogen removal. The Clean Water Organizations therefore propose that Section 13.3 be revised as follows:

> The Permittee shall ~~control~~ limit manure application rates so the estimated nitrogen available to crops from all nitrogen sources (including commercial fertilizer) does not exceed expected annual crop nitrogen needs for non-legumes and expected nitrogen removal for legumes. Expected crop nitrogen needs, crop nitrogen removal rates, and estimated plant available nitrogen from manure and legumes must be based on the most recent published recommendations of the University of Minnesota Extension Service, but must not be based on recommendations incorporating cost-factors for nitrogen fertilizer (i.e., MRTN system).. ~~based on the most recent recommendations of the MES and the MPCA fact sheets "*Manure Nitrogen Rates For Corn Production (wq-f8-18)*" and "*Manure Management For Corn On Irrigated Sandy Soils (wq-f8-52)*".~~ The Permittee may use recommendations for annual crop nitrogen needs from another land grant college in a contiguous state ~~may be utilized~~ in the MMP provided the field and climate conditions at the land application site are similar to those within the contiguous state, and do not incorporate cost-factors as set forth above. [Minn. R. 7020.2225]

2. The MRTN for manure should not be calculated using a lower cost ratio than that used for commercial nitrogen fertilizer.

The manure application rates identified by Extension Service are also improper and inconsistent with the Land Application Rule because the rates are formulated based on the cost of commercial nitrogen fertilizer and often produce excessive results when used for manure. If MPCA uses the MRTN recommendations, at a minimum those recommendations should be the same for manure as for commercial fertilizer. After all, expected crop nitrogen needs should not change based on whether the farmer applies commercial fertilizer or manure.

As explained above, the MRTN is calculated based on the ratio of the cost of *commercial nitrogen fertilizer* to the expected sale price of the crop. Minnesota's recommendations for the

MRTN for commercial fertilizer include calculations that use ratios of 0.05, 0.10, 0.15, and 0.20 to account for price fluctuations in fertilizer and corn.[92] However, because the ratio of the prices of fertilizer to corn has remained approximately the same, the 0.10 ratio usually been used for commercial fertilizer recommendations in Minnesota.[93]

For manure, considerations are different. Often, the farmer owns or manages livestock and may obtain manure without paying for it.[94] Presumably to account for that fact that manure is cheaper than fertilizer, the Extension Service recommendations identified in the Proposed General Permit do not use the 0.10 ratio that would be used for commercial fertilizer. Instead, the recommendations use the 0.05 ratio.[95]

This leads to a significantly larger nitrogen recommendation for manure application than for commercial fertilizer, Dr. Jones explains. As an example, using the 0.10 ratio for corn grown after soybeans produces a recommended MRTN of 131 lb. N/acre, with a profitable range of 118–144 lb. N/acre.[96] Using the 0.05 ratio, by contrast, increases the MRTN Rate to 150 lb. N/acre and the profitable range to 135–169 lb. N/acre.[97] Thus, under the Extension Service recommendations, for the same field, a farmer could add 19 lb. N/acre when applying manure instead of commercial fertilizer. These two examples are shown below in Figure 1:

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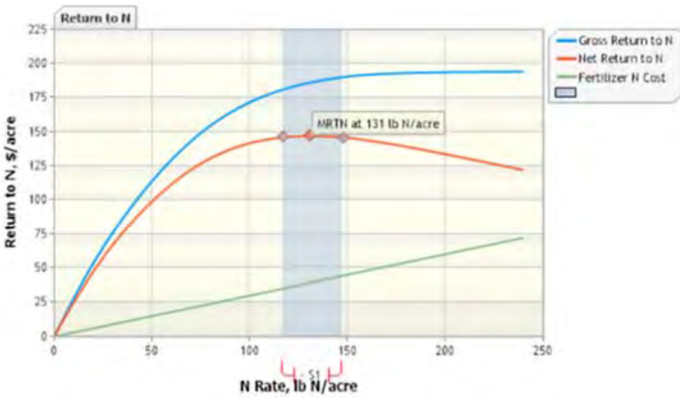
[92] Daniel Kaiser, et al., Univ. of Minn. Extension, *Fertilizing Corn in Minnesota* (2020), https://extension.umn.edu/crop-specific-needs/fertilizing-corn-minnesota#standard-n-guidelines-2237060, attached as Ex. 17.
[93] Dr. Gyles Randall, *Expert Report* 1 (2020), attached as Ex. 2.
[94] Jones, *supra* note 35, at 7.
[95] Manure Nitrogen Rates for Corn, *supra* note 75; Guidelines for Manure Application, *supra* note 40.
[96] Jones, *supra* note 35, at 7.
[97] *Id.*

Rates and Charts

State: Minnesota
Number of sites: 121
Rotation: Corn Following Soybean

Nitrogen Price ($/lb):	0.30
Corn Price ($/bu):	3.00
Price Ratio:	0.10
MRTN Rate (lb N/acre):	131
Profitable N Rate Range (lb N/acre):	116 - 147
Net Return to N at MRTN Rate ($/acre):	$146.86
Percent of Maximum Yield at MRTN Rate:	99%
Anhydrous Ammonia (82% N) at MRTN Rate (lb product/acre):	159
Anhydrous Ammonia (82% N) Cost at MRTN Rate ($/acre):	$39.30



Rates and Charts

State: Minnesota
Number of sites: 121
Rotation: Corn Following Soybean

Nitrogen Price ($/lb):	0.15
Corn Price ($/bu):	3.00
Price Ratio:	0.05
MRTN Rate (lb N/acre):	150
Profitable N Rate Range (lb N/acre):	135 - 169
Net Return to N at MRTN Rate ($/acre):	$168.13
Percent of Maximum Yield at MRTN Rate:	99%
Anhydrous Ammonia (82% N) at MRTN Rate (lb product/acre):	182
Anhydrous Ammonia (82% N) Cost at MRTN Rate ($/acre):	$22.50

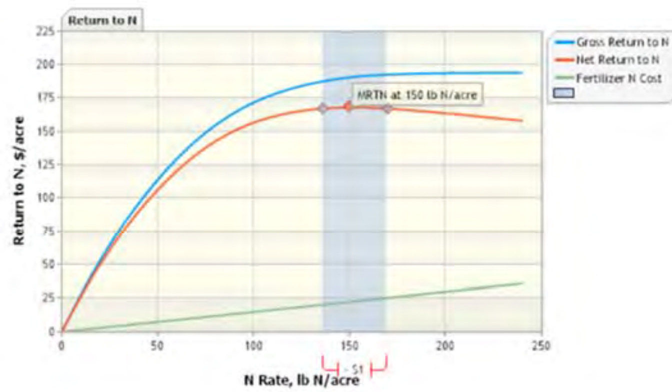


Figure 1. Two scenarios using the MRTN Calculator for commercial anhydrous ammonia (left) and manure (right), at current prices for N fertilizer and corn, using the Proposed General Permit's guidelines for manure application rate at 0.05 MRTN.[98]

Importantly, the orange line's downward slope to the right of the MRTN shows that a farmer who uses commercial fertilizer beyond the MRTN will incur an economic penalty.[99] By contrast, as Dr. Jones explains, "there is almost no economic consequence for the farmer to keep applying manure far beyond the MRTN rate, which is already 19 lbs./acre higher than the recommended rate using commercial nitrogen."[100] In addition, the difference between the total net return to the farmer for commercial fertilizer and manure is notable. When using commercial fertilizer at the 0.10 MRTN rate, the farmer achieves a net return of $146.86/acre. When using

[98] *Id.* at 7.
[99] *Id.*
[100] *Id.*

manure, the farmer could achieve that same net return using an application rate far below the 0.05 MRTN rate—about 80 lbs./acre in this example.[101] Clearly, the farmer using manure can achieve economic parity with the farmer using commercial fertilizer, even while applying manure at a rate far below the Extension Service recommendations.[102] But, according to Dr. Jones, "the Extension Service guidelines do quite the opposite—they encourage application of [nitrogen] far beyond that threshold."[103]

For this additional reason, the Extension Service's recommendations, which are referenced in the Proposed General Permit, do not comply with the Land Application Rule requirement that limits manure application to "expected crop nitrogen needs" or "nitrogen removal rates." The actual crop needs for nitrogen do not change based on whether a farmer applies commercial nitrogen fertilizer or manure, or based on a change in the cost of fertilizer. Accordingly, if the MPCA elects to use the MRTN, it is unreasonable and inconsistent with the Land Application Rule to use a different MRTN for commercial fertilizer than for manure.[104] If the 0.10 MRTN rate provides sufficient nitrogen for plant growth when commercial fertilizer is used, that same rate will provide sufficient nitrogen to meet the expected crop nitrogen needs or nitrogen removal rates when manure is used.[105]

Accordingly, if the MPCA determines that the recommended rate should remain the MRTN, the Clean Water Organizations propose that Section 13.3 be revised as follows:

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[101] *Id.*
[102] *Id.*
[103] *Id.*
[104] Notably, one of the original MRTN developers has stated that the price of commercial nitrogen fertilizer *should* be used to calculate the MRTN ratio for manure, instead of the lower rate indicating that manure is less expensive. Randall, *supra* note 90, at 1.
[105] Maximizing the amount of manure to apply is particularly inappropriate when the Land Application Rule already allows farmers to deviate up to 20% in excess of recommendations when needed under the circumstances. Minn. R. 7020.2225, subp. 3(A)(2).

> The Permittee shall ~~control~~ limit manure application rates so the estimated nitrogen available to crops from all nitrogen sources (including commercial fertilizer) does not exceed expected annual crop nitrogen needs for non-legumes and expected nitrogen removal for legumes. Expected crop nitrogen needs, crop nitrogen removal rates, and estimated plant available nitrogen from manure and legumes must be based on the most recent published recommendations of the University of Minnesota Extension Service, but must not be based on recommendations incorporating cost-factors for nitrogen fertilizer (i.e., MRTN system) unless the MRTN recommendation used is based on a cost factor of at least 0.10. ~~based on the most recent recommendations of the MES and the MPCA fact sheets "*Manure Nitrogen Rates For Corn Production (wq-f8-18)*" and "*Manure Management For Corn On Irrigated Sandy Soils (wq-f8-52)*".~~ The Permittee may use recommendations for annual crop nitrogen needs from another land grant college in a contiguous state ~~may be utilized~~ in the MMP provided the field and climate conditions at the land application site are similar to those within the contiguous state, and if the recommendations are based on the MRTN, they use a cost factor of at least 0.10. [Minn. R. 7020.2225]

B. Section 13.3(a): Pre-Plant Testing For Nitrate.

Next, the Clean Water Organizations request that MPCA add back into the Proposed General Permit a section relating to pre-plant testing for nitrate. MPCA included such a section in the 2016 General Permit, and it is needed to comply with the Minnesota Rules and to ensure that farmers are not over-applying manure that will cause water pollution.

The Land Application Rule requires that manure management plans include "plans for soil nitrate testing in accordance with University of Minnesota Extension Service recommendations."[106] Under the rules, any required testing must be sufficient to yield representative data to determine whether a permittee is complying with the conditions of the permit and state rules.[107] In this case, the Land Application Rule and the Proposed General Permit require farmers to limit manure applications to "expected crop nitrogen needs" or "nitrogen removal rates." The Land Application Rule and the Proposed General Permit also require that in calculating these amounts, farmers consider *all* sources of nitrogen available to their crops, including

[106] *Id.*, subp. 4(D)(12).
[107] Minn. R. 7001.0150, subp. 2(B).

"commercial fertilizer nitrogen, soil organic matter, irrigation water, legumes grown during previous years, biosolids, process wastewater, and manure applied for the current year and previous years."[108] Accordingly, nitrate testing is needed to ensure that farmers properly account for all nitrogen sources, and that farmers do not apply nitrogen in excess of expected crop nitrogen needs. In short, farmers cannot limit their application to the crop's expected nitrogen needs if they do not know how much nitrogen is already in the soil, and they cannot know how much nitrogen is in the soil without testing.

Determining how much nitrogen farmers should credit from previous years is not an easy task without testing. Many factors affect how much residual nitrogen remains in the soil, including the previous crop grown, the soil texture, and historic rainfall.[109] One of the most important factors—with the most difficult-to-predict effects—is the amount of residual nitrates that remain from manure applied in previous years.[110] As the Extension Service explains, microbes require several years to mineralize organic forms of nitrogen in manure into nitrate that can be used by plants, and the length of the process depends on soil moisture and temperature conditions.[111] Accordingly, manure applied in one growing season will continue to provide nitrate to plants for several growing seasons.[112] The amount of residual nitrogen, however, can vary greatly, is difficult to predict, and can have substantial effects on the amount of preplant nitrogen that should be added to the soil.[113] As Dr. Randall explains, a soil test of 13 sites where manure had been applied in the

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[108] Minn. R. 7020.2225, subps. 3(A)(1), (A)(3)
[109] Univ. of Minn. Extension, *Soil Testing for Corn Nitrogen Recommendations* (2018), https://extension.umn.edu/nitrogen/soil-testing-corn-nitrogen-recommendations, [hereinafter "Soil Testing for Corn"], attached as Ex. 18.
[110] *Id.*
[111] Manure Characteristics, *supra* note 41, at 6.
[112] *Id.*
[113] *Id.*; *see also* Randall, *supra* note 90, at 2.

previous five years showed that the amount of nitrogen to be applied should be reduced by an average of 43 lb. N/acre based on the residual nitrogen.[114] For several sites, the recommended rate of nitrogen to be applied was reduced by 70 lb. N/acre, and for others it was reduced by only 19 lb. N/acre, showing the wide range of results that manure application can have at different fields.[115]

Accounting for nitrates released from manure over time can be done using a "credit" for manure from the previous two years.[116] But the crediting system cannot precisely account for the actual amount of nitrates, and in some cases may result in excessive fertilizer recommendations.[117] Measuring nitrates in the soil is more reliable than other methods of estimating the need for additional nitrogen application.[118] As the Iowa State University Extension Service explains, using a late-spring test for soil nitrate "should help corn producers manage N to increase their profits while reducing environmental degradation."[119]

Currently, the Proposed General Permit does not include any requirement for soil testing for nitrogen, although it does require soil testing for phosphorus.[120] The 2016 General Permit, however, *does* require soil nitrate testing "according to the method and frequency recommended by the most recent MES-published guidelines."[121] It is unclear why MPCA removed this requirement in the Proposed General Permit. To comply with the requirements of the Land

[114] Randall, *supra* note 94, at 2.
[115] *Id.*
[116] Manure Characteristics, *supra* note 41, at 4.
[117] Soil Testing for Corn, *supra* note 106, at 4 (explaining that using the standard manure nitrogen crediting system without a soil test when manure was applied in October or November "may result in high fertilizer recommendations if significant residual nitrogen was present before the manure was applied.")
[118] A.M. Blackmer et al., Iowa State Univ. Extension Serv., *Nitrogen Fertilizer Recommendations for Corn in Iowa* 4 (1997), attached as Ex. 19.
[119] *Id.* at 1.
[120] *See* Proposed General Permit, *supra* note 75, § 12.6.
[121] Minn. Pollution Control Agency, *NPDES General Permit for Concentrated Animal Feeding Operations* § 4.5.4 (2016).

Application Rule and ensure farmers are able to accurately determine the proper amount of manure they should apply, the Clean Water Organizations propose that the following language be added to the Proposed General Permit:

> The Permittee shall ensure that fields receiving manure are sampled and tested for soil nitrates according to the method and frequency recommended by the most recent MES-published guidelines. The Permittee shall use the results of the sample in calculating a residual N credit. [Minn. R.7020.2225, subp. 3.A(3)].

C. Section 14.6: October Restrictions On Manure Application.

The Clean Water Organizations also request changes to the section regarding October Restrictions on Manure Application to better guard against nitrate pollution. The Clean Water Organizations appreciate that the Proposed General Permit now requires best management practices ("BMPs") for any manure application in October, but believes that those requirements should be strengthened to further protect water quality.

First, with regard to the soil temperature, the proposed language provides no direction about how to determine soil temperature. This is important, because fall manure application when temperatures exceed 50º F is highly likely to cause nitrate pollution. In such cases, the organic nitrogen will be mineralized to inorganic nitrate at a time when the crops are not growing.[122] Then, the nitrate will remain in the soil until the crop takes it up, possibly not until the following June.[123] The longer the nitrate remains in the soil, the more likely it is to leach into the groundwater—particularly during heavy rains in the fall or early spring.[124] Accordingly, ensuring that soil temperatures prior to manure application are below 50º F, and are likely to *remain* that way until spring, is critical. Allowing farmers to apply manure as soon as their area has one 50º F soil

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[122] Fred Madison et al., Univ. of Wis. Extension Serv., *Guidelines for Applying Manure to Cropland and Pasture in Wisconsin* 11 (2014), https://soilsextension.webhosting.cals.wisc .edu/wp-content/uploads /sites/68/2014/02/A3392.pdf, attached as Ex. 20.
[123] Randall, *supra* note 90, at 2.
[124] *Id.* at 2; Madison, *supra* note 119, at 11.

temperature reading will not prevent nitrate leaching, as mineralization to nitrate will begin again if the soil temperatures rise after manure application. To ensure consistency, Dr. Randall recommends that soil temperature readings be taken at a depth of six inches and be less than 50 degrees for three consecutive days before farmers apply manure.[125]

Second, with regard to cover crops, the Proposed General Permit indicates manure may be applied in October if a cover crop "is established in accordance with the requirements of this Permit for June, July, August, or September applications." But the likelihood that a cover crop can be established drops quickly after the first half of September, particularly in the northern half of the state.[126] After October 1, establishing a cover crop would be very difficult even in southern Minnesota and extremely unlikely in northern Minnesota.[127] To effectively prevent nitrate pollution, a cover crop must not merely be germinated—it must be well-established and sufficiently robust to take up a substantial amount of nitrate from the manure.[128] This means the crop must be well-grown—perhaps six to eight inches tall—by mid-to-late October.[129] A cover crop planted in October is extremely unlikely to fulfill its intended function as a temporary fixer of nitrates.[130] But the Proposed General Permit would allow a farmer to seed a cover crop in October within 10 days of manure application and hope for the best—and there would be no way to remove the manure if the cover crop does not sprout. Any manure applied under these circumstances is very likely to mineralize to nitrate and leach into the groundwater.[131] If, however,

[125] Randall, *supra* note 90, at 3. For the same reason, this standard—three consecutive days of temperatures below 50 degrees, measured at a soil depth of six inches below the surface—also should be added to section 14.4, relating to manure application on coarse-textured soils.
[126] Randall, *supra* note 90, at 3.
[127] *Id.*
[128] *Id.*
[129] *Id.*
[130] *Id.*
[131] *Id.*

a cover crop has already been established prior to October, application of manure through an injector into the growing cover crop could be a potential BMP.[132] Therefore, the Proposed General Permit should be revised to indicate that cover crops may be used as a BMP for October manure application only if the cover crop has been planted in a previous month and already established before the October application.

Third, for the split application of nitrogen, the Proposed General Permit does not indicate when the second half of the nitrogen could be applied. Applying the second half of the manure soon after the first half—in early November, for example—would negate the effectiveness of splitting the nitrogen application. And manure application during the winter months, to frozen or snow-covered soils, is prohibited or subject to strict conditions under the terms of the permit.[133] Even under those conditions, winter manure application is risky and likely to lead to runoff, as explained in the next section. Under no circumstances should applying manure during winter months be considered a BMP. Accordingly, the Proposed General Permit should specify that the second half of the split application of nitrogen should be applied only in the spring, when the ground is no longer frozen.

Finally, the Proposed General Permit does not require implementation of BMPs during an "emergency" manure application, perhaps on the assumption that BMPs would not be feasible. But in some cases, farmers may in fact be able to implement these BMPs despite an emergency. For example, a nitrogen stabilizing agent potentially could be added to the manure before spreading, despite poor weather conditions or equipment failure that prevented an earlier manure

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[132] *Id.*
[133] *See* Proposed General Permit, *supra* note 75, §§ 14.8, 14.10.

application.[134] In such cases, when following the BMPs remains feasible, farmers should not be excused from following the BMPs intended to prevent nitrate pollution.

Accordingly, to better protect water quality, the Clean Water Organizations propose the following revisions to Section 14.6:

> October Restrictions - The Permittee shall not apply manure in October to harvested fields unless at least one of the following nitrogen BMPs are implemented:
>
> a) Soil temperature ~~is~~ has been below 50 degrees for three consecutive days at the time of manure application based on temperatures taken six inches below the soil surface;
>
> b) A nitrogen stabilizing agent/product is added at the recommended inclusion rates;
>
> c) A cover crop ~~is~~ has been established prior to October in accordance with the requirements of this Permit for June, July, August, or September manure applications; or
>
> d) A split application of nitrogen is used where no more than 1/2 of the recommended nitrogen rate is applied before October 31 and the remainder is applied after April 1 or after the soil is no longer frozen or snow-covered, whichever is later.
>
> ~~Alternatives developed by a land grant University can be used if approved by the MPCA and included as part of the approved MMP.~~
>
> Nitrogen BMP implementation is ~~not~~ required for emergency manure application, as defined by this Permit, unless implementation of BMPs is infeasible due to the emergency conditions necessitating the application. [Minn. R. 7001.0150]

D. Section 14.8: Winter Application Of Solid Manure.

Similarly, while the Clean Water Organizations appreciate MPCA's efforts to strengthen the Proposed General Permit's section on winter application of solid manure, a broader prohibition could make this section even stronger. Prohibiting application of solid manure in December and January, along with February and March, will provide even better protection against nitrate pollution.

[134] *See id.* § 30.19 (defining "emergency manure application").

When farmers apply manure to snow-covered or frozen soil, nutrients cannot soak into the soil, and the potential for nitrate loss is "extremely high."[135] When farmers apply manure during the winter months, the majority of the inorganic nitrogen is likely to be lost to the air through volatilization.[136] And winter-applied manure is very likely to be "carried off to lakes and streams during thaws or during winter or early spring rains."[137] For these reasons, the Proposed General Permit contains a prohibition on applying solid manure during February and March. However, these same considerations apply with equal force to December and January, when the ground is also likely to be frozen or snow-covered.[138] Accordingly, the Clean Water Organizations propose the following revision:

> Winter application of solid manure - Winter application of solid manure during the months of December, January, February and March is prohibited. When allowed, winter application must comply with all of the following:
>
> a) Manure is applied on fields identified in the MPCA approved MMP for winter application;
>
> b) Manure is applied more than 300 feet from sensitive features including lakes, streams, open tile inlets, sinkholes, water supply wells, mines and quarries, intermittent streams, un-bermed drainage ditches, or public water wetlands;
>
> c) Air temperatures are less than 40 degrees Fahrenheit during, and for at least 24 hours from the end of, the application process when two or more inches of snow are on the field;
>
> d) Less than a 50% probability of rainfall in excess of 0.25 inches predicted by the National Weather Service within 24 hours of the end of the application period;
>
> e) Slopes are less than or equal to six percent on the entire portion of the field where manure is land applied;

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[135] Manure Timing, *supra* note 52. This Extension Service publication recommends, unless there is an emergency, "Do not apply in winter." *Id*.
[136] Soil Testing for Corn, *supra* note 106, at 4.
[137] Madison et al., *supra* note 119, at 15.
[138] If the ground is not frozen or snow-covered in December or January, then the application would not qualify as a "winter manure application" under the Proposed General Permit definition and therefore would not be prohibited. *See* Proposed General Permit, *supra* note 75, § 30.53.

> f) Water or ice do not occupy tillage furrows to the extent that additional snowmelt or precipitation cannot be contained between furrows or in other depressions within the field; and
>
> g) Fields used for land application meet a total phosphorus loss risk index number of two or less (low to very low relative risk) as calculated according to the Minnesota Phosphorus Index.
>
> In the event of significant snow accumulation within animal holding areas, the Permittee may obtain approval from the MPCA for winter application of the snow and manure-snow mix during December, January, February and March. If approved, the application fields must, at a minimum, meet the requirements above. Additional measures/practices may be required by the MPCA. [Minn. R. 7001.0150]

E. Section 11.4: Review Of Manure Management Plan.

Finally, revising Section 11.4 to require farmers to identify fields in manure management plans ("MMP") using GIS information will assist MPCA staff. Using GIS information will make it easier for MPCA to determine whether any fields receive double applications of manure because they are identified in more than one MMP and receiving manure from more than one farmer.

Pursuant to the Land Application Rule, MMPs "must include acreage available for manure and process wastewater application including maps or aerial photos showing field locations and areas within the fields that are suitable for manure or process wastewater application."[139] The rule, accordingly, requires farmers to specifically identify fields in the MMPs. Identification through GIS information will make descriptions on MMPs more readily comparable for MPCA staff. Under the current system, two applicants could describe the same field using different descriptors, and determining whether there is overlap between two plans is cumbersome for MPCA staff, who must compare different maps or aerial photographs to determine whether the same field has been identified in more than one MMP. Using GIS information would standardize descriptions of fields

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[139] Minn. R. 7020.2226, subp. 4(D)(3).

in the MMPs, making it clear to both MPCA staff and applicants which fields are being referred to in the MMP.

Accordingly, the Clean Water Organizations propose the following revision to Section 11.4:

> The Permittee shall annually review and update the approved MMP to ensure that it meets all applicable requirements. The annual review and update shall include information for each field where manure will be applied during the following growing season. The permittee shall provide an area delineation of each manure application site in a GIS polygon geospatial file format (.kml, .shp, .json, etc.) with detailed coordinate system information, including a description of the site. Annual updates to the MMP do not require a modification of coverage under this Permit provided the updates are consistent with the methodology of the approved MMP. [Minn. R. 7001.0190, Minn. R. 7020.2225]

IV. THE CLEAN WATER ORGANIZATIONS REQUEST A CONTESTED CASE HEARING

The Clean Water Organizations request a contested case hearing on the issue of whether the recommendation MPCA has referenced in Section 13.3 of the Proposed General Permit is consistent with "expected crop nitrogen needs, crop nitrogen removal rates, and estimated plant available nitrogen from manure and legumes" as required by the Land Application Rule.

The information required by Minn. R. 7000.1800 is provided below.

1. Statement of reasons or proposed findings supporting an MPCA decision to hold a contested case hearing.

(A) There is a material issue of fact in dispute concerning this matter.

As noted in the Clean Water Organizations' comments above in section V.A, the Proposed General Permit references recommendations from the University of Minnesota that incorporate economic risk and cost factors unrelated to the amount of nitrogen a typical crop[140] will actually need or remove to support plant growth. As a result, these recommendations are inconsistent with

[140] MCEA notes that Minn. R. 7020.2225, subp. 3 already provides for increased nitrogen application if conditions particular to the crop or field require additional applications to secure the crop.

what the Land Application Rule requires and will allow permittees to apply manure at rates resulting in excess loss of nitrate to the groundwater, exacerbating the issues the Clean Water Organizations describe in section II.B above. Whether the recommendations conform to the objective requirement of the rule is a factual issue that can be resolved with expert testimony.[141] This expert testimony will identify the results of research into "expected crop nitrogen needs, crop nitrogen removal rates, and estimated plant available nitrogen from manure and legumes," why the economic components incorporated into the current recommendation result in applications not supported by the scientific data, and why the recommendations will lead to excess application inconsistent with the text and intent of MPCA's land application rule.

(B) The MPCA has the jurisdiction to make a determination on this issue.

In the proposed general permit, MPCA has referenced a particular recommendation of the Extension Service. If MPCA agrees with the Clean Water Organizations that the recommendation it references is not consistent with the standard established by the Land Application Rule, MPCA could ask the Extension Service to modify its recommendation, or MPCA could modify the Proposed General Permit to ensure that a modified version of the Extension Service's recommendations are referenced in the Proposed General Permit. As a result, this issue is within MPCA's jurisdiction.

(C) There is a reasonable basis underlying the disputed material issue of fact or facts such that the holding of a contested case hearing would allow the introduction of information that would aid the MPCA in resolving the disputed facts in making a final decision on the matter.

[141] *See In re City of Owatonna's NPDES/SDS Proposed Permit Reissuance for the Discharge of Treated Wastewater*, 672 N.W.2d 921, 928 (Minn. Ct. App. 2004) (finding a fact issue supporting a contested case hearing request existed when relator submitted expert affidavits and a report challenging MPCA's interpretation of its modeling and explaining, "When experts disagree, a fact question arises.")

The Clean Water Organizations support this request with two expert reports, by Dr. Christopher Jones, research engineer at Iowa State University (attached as Exhibit 1) and Dr. Gyles Randall, professor emeritus at the University of Minnesota (attached as Exhibit 2).[142] These experts will testify that the recommendation currently included in the Proposed General Permit is not consistent with the standard established by the Land Application Rule.[143] These experts will base their testimony on research conducted in Minnesota and Iowa. These experts will demonstrate that the economic factors incorporated into the current recommendations, particularly as applied to manure, result in excess application inconsistent with "expected crop nitrogen needs, crop nitrogen removal rates, and estimated plant available nitrogen from manure and legumes" and that this excess application can be predicted to lead to enhances nitrogen loss to the groundwater.

2. A statement of the issues proposed to be addressed by a contested case hearing and the specific relief requested or resolution of the matter.

The issue to be addressed by a contested case hearing is whether the recommendation referenced in the Proposed General Permit conforms to the standard established by the Land Application Rule. The relief requested is amendment of the Proposed General Permit to include a recommendation that will result manure application rates consistent with plant needs established by scientific research, as required by the Land Application Rule.

Clean Water Organization has identified two changes that MPCA could make to the Proposed General Permit to address this issue, in section V.A, above. First, MPCA could request the Extension Service to prepare a recommendation that does not include the economic factors on which the current MRTN recommendation is based. Second, MPCA could request the Extension

[142] *See* Jones, *supra* note 35; and Randall, *supra* note 90.
[143] *See City of Owatonna,* 672 N.W.2d at 929 (explaining that relator had sufficiently supported the requested for a contested case hearing when it submitted affidavits of experts who challenged MPCA's methodology and interpretation of the modeling at issue).

Service to prepare a recommendation specific to manure that utilizes the MRTN, but includes a higher cost factor ratio similar to the one used for commercial fertilizer, which is less likely to result in over-application of manure.

3. Witnesses, exhibits, and estimate of time.

At a contested case hearing, the Clean Water Organizations would intend to present the following witnesses: Dr. Christopher Jones and Dr. Gyles Randall. Proposed exhibits would include all exhibits attached to this comment or referenced herein. The estimated time for the contested case hearing would be a half-day. The Clean Water Organizations reserve the right to introduce other witnesses or exhibits in accordance with Minn. R. 7000.1800, subp. 2(C). The Clean Water Organizations note that MCEA has been seeking a meeting with MPCA and Extension Service representatives to discuss the use of MRTN recommendations, which could lead to changes that would resolve this issue without a contested case hearing.

CONCLUSION

While the Clean Water Organizations appreciate that the Proposed General Permit makes some incremental changes that are likely to help improve water quality, the Clean Water Organizations' position is that the Proposed General Permit will allow the continued pollution of Minnesota's water, endangering drinking water and aquatic life. Already, hundreds of thousands of Minnesotans are drinking water with elevated levels of nitrates, which will increase their risks of cancers and other health problems. If farmers are allowed to continue to apply manure to their fields in excess of crop nitrogen needs, and at times and using methods that pose high risk of nitrate leaching and runoff, dangerous nitrate pollution will continue to increase across Minnesota. Accordingly, the Clean Water Organizations respectfully request that MPCA revise the Proposed General Permit as follows:

(1) revise Section 13.3 to limit manure application rates to "expected crop nitrogen needs" or "expected nitrogen removal"; or in the alternative, to ensure that the MRTN uses a cost factor of at least 0.10;

(2) add Section 13.3(a) to require pre-plant testing for nitrate according to Extension Service recommendations;

(3) revise Section 14.4 to require soil temperature measurements below 50 degrees for three consecutive days, measured at a soil depth of six inches below the surface;

(4) revise Section 14.6 to strengthen October restrictions on manure application;

(5) revise Section 14.8 to prohibit application of solid manure in December and January; and

(6) revise Section 11.4 to require GIS field identification in MMPs.

Respectfully submitted,

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Cover crops | UMN Extension

extension.umn.edu/soil-and-water/cover-crops

Cover crops

Cover crops are grown outside of the cash crop growing season, usually seeded in the fall and killed before spring planting.

Keeping living roots in the ground year-round can improve water management, soil protection and nutrient scavenging, but they need to be given the same attention as a cash crop to ensure success.

Try cover crops on a small scale first, and look into cost-share from state and local governments.

Some of the best opportunities are with early-harvested cash crops like corn silage, small grains, and canning crops like beans and peas, as you'll get more vigorous fall growth if you plant in late summer and early fall.

In fields where wheat was just harvested, simply allowing it to reseed itself without tilling the land would work as a cover crop. But cover crops can work with standard corn-soybean rotations as well.

Benefits of cover crops

Erosion control

Cover crops reduce erosion in a few different ways.

- Aboveground, living cover crops protect the soil from rainfall impact and reduce the effect of wind. Runoff is reduced along the way.
- Belowground, roots hold soil in place during active erosion events and build structure. Better soil structure means the soil is less likely to erode even if left bare later in the season, such as between harvest and cover crop planting.
- Runoff sediment also contains soil phosphorus, so reducing runoff is an important strategy for reducing P loading in surface water.

Infiltration and water management

Cover crop root systems create large channels through the soil to allow increased infiltration. This effect is especially significant for species with large taproots, but other cover crops also increase infiltration.

- Increased infiltration means fields are less likely to stay saturated during Minnesota's rainy springs.
- Many farmers report dry field conditions more quickly after a rain event when they use cover crops.

Cover crops can also help soil store water by building soil structure and creating a network of large and small pores.

- Once water enters the soil through infiltration, this pore network retains water for plants to take up as necessary.
- This increase in soil water holding capacity can be especially beneficial in dry years.

Nitrate reduction

Soil nitrate reduction is well-established in Minnesota for a variety of cover crops.

- Nitrate is often left in the soil after fall harvest of corn.
- A winter cover crop takes up soil nitrogen, so less nitrogen is leached. This is an important benefit for reducing groundwater nitrate contamination.
- Farmers should expect some nitrate drawdown by cover crops and plan the subsequent season's fertility accordingly.
- Soil testing before applying N to cash crops can help with field-specific recommendations.

Minnesota cover crop recipes

For a quick way to get started, Minnesota cover crop recipes provide step-by-step guidance to some of the lowest-risk starting points for cover crops. These recipes don't cover all possibilities, but they can help beginners incorporate cover crops into a farm operation.

- Post corn, going into soybean: Use cereal rye
- Post soybean, going to corn: Use oats
- Post corn silage, going to corn: Use cereal rye
- Post corn silage, going to soybean: Use cereal rye

Learn more about reducing tillage and incorporating cover crops:

Watch Video At: https://youtu.be/videoseries

Getting started with cover crops

- Benefits of cover crops.
- Choosing a cover crop (consider crop rotation, harvest timing, overwintering, etc.).
- Recommended planting dates and seeding rates for cover crops.
- Comparison of cover crop benefits by crop.

Planting green in Minnesota

- Benefits of biomass production
- How termination date affects biomass production

Planting date matters for cover crops, too

- Timing of fall seeding
- Termination timing on biomass production

Reduce risk of fallow or flooded soil syndrome with cover crops

- How and when fallow syndrome occurs.
- How it affects crops.
- How cover crops can help.

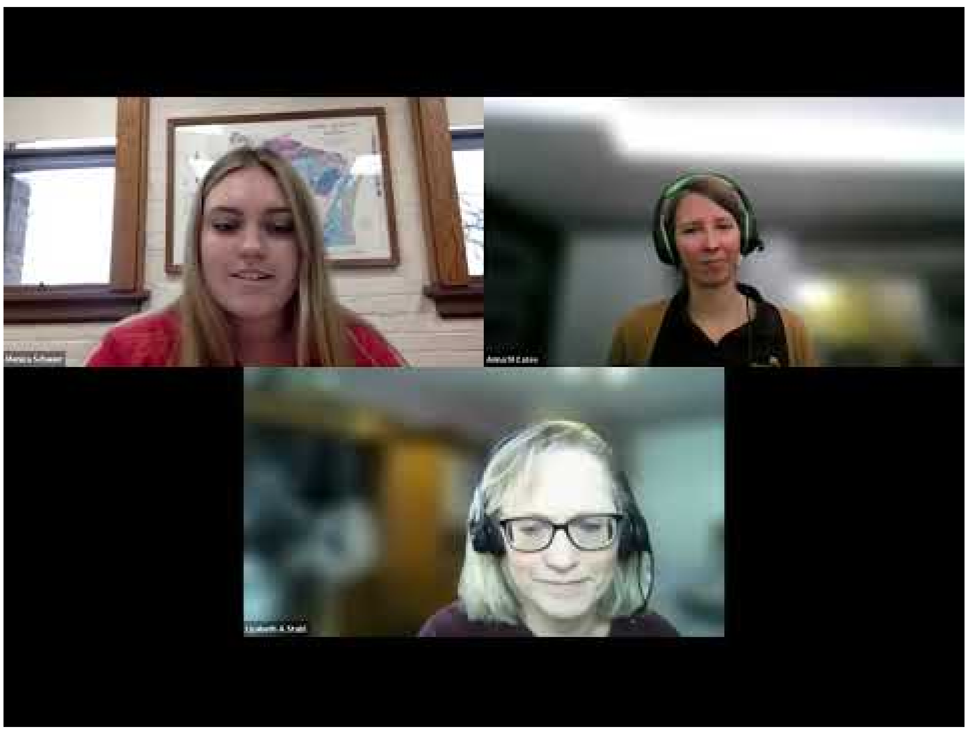
September 3, 2024
Clean Water Organizations Comments Exhibit 36

- How to manage fallow syndrome.

Spring management of cover crops

- Guidance on mechanical and chemical termination, including carbon-to-nitrogen ratios of common crops.
- Factors affecting residue.
- Pest management tips.
- How to time spring termination for cash crop planting.

In this Strategic Farming webinar, researchers Monica Schauer, UW-Madison, and Anna Cates, UMN Extension educator, discuss fertility and crop rotation with cover crops.



Watch Video At: https://youtu.be/XEYbPRzbGN4

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Reductions in soil water nitrate beneath a perennial grain crop compared to an annual crop rotation on sandy soil

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Nitrate (NO_3^--N) leaching into groundwater as a result of high nitrogen (N) fertilizer rates to annual crops presents human health risks and high costs associated with water treatment. Leaching is a particularly serious concern on sandy soils overlying porous bedrock. Intermediate wheatgrass (IWG) [*Thinopyrum intermedium* (Host.) Barkw. & D.R. Dewey], is a perennial grass that is being bred to produce agronomically and economically viable grain, which is commercially available as Kernza®. Intermediate wheatgrass is a low-input crop has the potential to produce profitable grain and biomass yields while reducing NO_3^--N leaching on sandy soils compared with common annual row crop rotations in the Upper Midwest. We compared grain yields, biomass yields, soil solution NO_3^--N concentration, soil extractable NO_3^--N, soil water content, and root biomass under IWG and a conventionally managed corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] rotation for 3 years on a Verndale sandy loam in Central Minnesota. Mean soil solution NO_3^--N was 77–96% lower under IWG than the annual crop rotation. Soil water content was greater under annuals compared to IWG early in the growing season, suggesting greater water use by IWG during this time. Interactions between crop treatments and depth were observed for soil water content in Year 3. Root biomass from 0 to 60 cm below the soil surface was five times greater beneath IWG compared to soybean, which may explain differences in soil extractable and solution NO_3^--N among crops. With irrigation on coarse structured soils, IWG grain yields were 854, 434, and 222 kg ha^{-1} for Years 1–3 and vegetative biomass averaged 4.65 Mg ha^{-1} yr^{-1}; comparable to other reports on heavier soils in the region. Annual crop grain yields were consistent with local averages. These results confirm that IWG effectively reduces soil solution NO_3^--N concentrations even on sandy soils, supporting its potential for broader adoption on land vulnerable to NO_3^--N leaching.

KEYWORDS

intermediate wheatgrass, nitrate, leaching, Kernza, groundwater, perennial grains

Introduction

Water quality in the Upper Midwest is threatened by the intensive management practices used in annual cropping systems, including tillage and fertilizer application that lead to nutrient losses and water contamination through leaching and runoff (Randall and Mulla, 2001; Dinnes et al., 2002; Feyereisen et al., 2006; Erisman et al., 2013). While annual commodity crops like corn provide the potential for high economic return, nutrient losses cause eutrophication and hypoxia in surface waters and contamination of groundwater, posing significant risks to human health (Ward et al., 2010, 2018; Brender et al., 2013). Impacts are often high where shallow aquifers and sandy soils make drinking water sources vulnerable to contamination. This leads to additional water treatment costs of over $5 million for some counties (Keeler et al., 2016). In Southeast Minnesota, for example, conversion of grassland to agriculture is expected to cause a 45% increase in private wells exceeding 10 ppm NO_3^--N, resulting in between $700,000 and $12,000,000 in associated costs over a 20-year period (Keeler and Polasky, 2014). New alternative cropping systems that provide economic returns comparable to those of annual systems and which effectively reduce nutrient losses will be essential for protecting drinking water sources in the future.

Replacing annual crops with perennials has the potential to help reduce NO_3^- leaching to groundwater and provide other ecosystem services (Asbjornsen et al., 2014; Ferchaud and Mary, 2016). Cropping systems that include perennial grasses for conservation, forage, and biofuel production have lower NO_3^- leaching losses than corn-soybean systems, largely because perennial grasses have greater root biomass that extends deeper into the soil, increasing N recovery and reducing leaching (Culman et al., 2013b; Pugesgaard et al., 2015; Ferchaud and Mary, 2016). Deep roots may be particularly important in reducing NO_3^- leaching since they can expand the total volume of soil from which NO_3^--N is taken up, and because NO_3^- is highly mobile and more prone to leaching from deep soil horizons (Maeght et al., 2013). NO_3^- losses in the subsurface drainage water for a corn-soybean system were about 37 times higher than from a Conservation Reserve Program (CRP) planting dominated by perennial grasses (Randall et al., 1997). This reduction was attributed to the greater season-long evapotranspiration (ET) that resulted in less drainage and greater uptake and/or immobilization of N. In that study, average NO_3^- concentrations in the water during the flow period were 24 mg/L for the corn-soybean rotation and 2 mg/L for the perennial grass CRP (Randall et al., 1997). Although plantings that include perennial grasses are effective at reducing NO_3^- leaching, a lack of economic return has prevented their large-scale adoption in Midwestern agricultural landscapes.

Intermediate wheatgrass (IWG), [*Thinopyrum intermedium* (Host.) Barkw. & D.R. Dewey] is a perennial cool-season grass being domesticated to produce a grain marketed as Kernza® (DeHaan et al., 2018) with the first commercial variety, "MN-Clearwater," released in 2020 (Bajgain et al., 2020). The crop has potential to provide economic return for producers (Hunter et al., 2020a,b; Law et al., 2022) while reducing NO_3^- leaching compared to corn (Jungers et al., 2019). Intermediate wheatgrass initiates growth earlier in the season than warm-season forage and bioenergy grasses and is thus better able to reduce NO_3^--N losses early in the season (Jungers et al., 2019) when losses are typically the highest in the Upper Midwest (Randall and Mulla, 2001; Crews and Peoples, 2005). Vegetative regrowth following IWG grain harvest helps reduce post-harvest nitrate losses and erosion late into the fall.

One potential mechanism by which IWG can reduce NO_3^- leaching compared to annual crops is related to water demand. Although total growing season ET and drainage were similar between IWG and corn, soil water content was lower under IWG compared to corn and switchgrass at 50 and 100 cm depths (Jungers et al., 2019), suggesting that soil moisture may be stored in other regions of the soil profile. Compared to annual wheat (*Triticum aestivum* L.), IWG had lower soil moisture up to a depth of 70–100 cm, which was associated with NO_3^--N leaching reductions of up to 86% (Culman et al., 2013b). The distribution of IWG root biomass and its effects on soil water content throughout the soil profile are largely unknown.

Reductions in NO_3^- leaching beneath IWG compared to annual crops can also be related to differences in nitrogen fertilization regimes and associated losses of N in the form of soluble NO_3^--N in the soil water. Soil solution NO_3^- increased from 0.1 to 0.3 mg L^{-1} when IWG was fertilized with 120 kg N ha^{-1} compared to an unfertilized control, yet this was still lower than the 24.0 mg L^{-1} measured beneath corn fertilized at 160 kg ha^{-1} (Jungers et al., 2019). Integrating legumes such as soybean into annual crop rotations can limit N fertilizing needs, yet the effects of legume crops in rotation on NO_3^--N leaching compared to IWG are unknown.

Our objective was to assess the potential of IWG grain production to reduce NO_3^--N leaching compared to an annual soybean-corn-soybean rotation on irrigated sandy soil by measuring soil solution NO_3^--N concentration and soil water content. We hypothesized that soil water NO_3^--N concentrations and soil water content would be lower under IWG, and that this would be related to increased root biomass and rooting depth of IWG compared to corn and soybean. Crop yields and vegetative biomass were measured to assess potential profitability.

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TABLE 1 Average air temperature, precipitation, irrigation, and 30-year averages for each month of the growing season in Staples, MN.

	Mean monthly air temperature (°C)				Monthly and season total precipitation (P) and irrigation (I) (mm)						
	2018	2019	2020	30-year avg.	2018		2019		2020		30-year avg.
					P	I	P	I	P	I	P
April	2	5	3	5	4.6	0	25.7	0	22.4	0	36.8
May	17	11	12	12	62.8	0	62.5	0	33.8	0	72.9
June	20	18	21	18	78.3	12.7	68.4	12.7	57.2	25.4	117.3
July	21	21	22	20	62.5	38.1	103.2	63.5	102.7	38.1	99.1
Aug.	19	18	20	19	66.6	38.1	93.8	12.7	158.6	38.1	74.4
Sept.	14	15	14	15	73.7	0	106.3	0	16	0	71.1
Oct.	4	5	3	7	80.0	0	92.3	0	10.9	0	56.6
					428.5	88.9	552.2	88.9	401.6	101.6	528.2

Methods

Site description

Field research was conducted from 2018 to 2020 at the Central Lakes Community College in Staples, MN, USA (lat. 46.38, long. −94.80). The soil type was a Verndale sandy loam (Typic Argiudoll). The soil contains 1–1.7% organic matter, is excessively well-drained, and is considered low fertility potential (USDA-NRCS, 2021). Local climate data are reported in Table 1. Plots had previously been planted to a corn-soybean rotation followed by barley fertilized with 40 kg N ha^{-1} applied in spring prior to IWG planting in 2017. Baseline soil samples from 0 to 30 cm were collected by block in the fall of 2017. Soil extractable nitrogen was 10.0 mg kg soil^{-1} for NO_3^--N and 3.9 mg kg soil^{-1} for ammonium (NH_4^+-N). Soil phosphorus (P) and potassium (K) concentrations were 9.13 and 72.21 ppm, respectively.

Experimental design

Treatments were applied in a randomized block design with two cropping systems replicated once in each of six blocks for a total of twelve plots. Plots were 4.11 by 9.14 m (13.5 by 30 ft.). The annual cropping system was a soybean-corn-soybean rotation. The perennial system was IWG. Soybeans were planted as the first phase of the soybean-corn rotation in May 2018, followed by corn in May 2019 and soybean again as the third phase in June 2020. Corn and soybeans were seeded in 75 cm rows at rates of 346,000 and 84,000 seeds ha^{-1}, respectively, with four rows per plot. The corn variety was Organic Viking O.84-95UP Seed Corn and the soybean was Organic MN0810CN.

An improved population of IWG bred for increased grain yield was used in this study. The population came from the fourth cycle of selection by Land Institute (Salina, KS) and was seeded at a rate of 15 kg ha^{-1}. The IWG was seeded in 15-cm rows with 20 rows per plot on 20 August 2017. Intermediate wheatgrass was fertilized with urea at rates of 80, 100, and 100 kg N ha^{-1} in May 2018, 2019, and 2020, respectively. Urea was split-applied to corn at 140 and 80 kg Nha^{-1} in May and June 2019. Soybean was not fertilized. Weed pressure was low and when present, weeds were manually removed in all plots. The experiment was irrigated with a linear irrigation system with events based on ET estimates and water demand for the annual crop. The fields received 89 mm of irrigation water over five events in 2018, 89 mm over seven events in 2019, and 102 mm over eight events in 2020. Dates of irrigation events are in Figure 1. Each individual irrigation event resulted in an application of 13 mm of water with the exception of 7/16/2018 and 8/22/2018, which received 25 mm.

Soil fertility and extractable N

Soil was sampled at four depth intervals (0–15, 15–30, 30–45, and 45–60 cm) in June 2019 and October 2020 and analyzed for organic matter, K, P, pH, and extractable NO_3^--N and NH_4^+-N. Samples were taken from eight cores in each plot and aggregated by depth, stored in a cooler when transported, and kept refrigerated until analyzed or processed for shipping. All soil analyses except extractable N were conducted by Agvise Laboratories (Benson, MN; www.agvise.com). Agvise samples were oven-dried prior to shipping. Soil extractable N was determined by extraction with a 2 M KCl solution, where 40 ml solution was added to 10 g fresh soil followed by 1 h shaking (Culman et al., 2013a). Extractions were performed within 48 h of field collection. NO_3^--N and NH_4^+ analyses of the extractions were performed at the UMN Research Analytical Lab. Method details can be found at http://ral.cfans.umn.edu/tests-analysis/soil-analysis.

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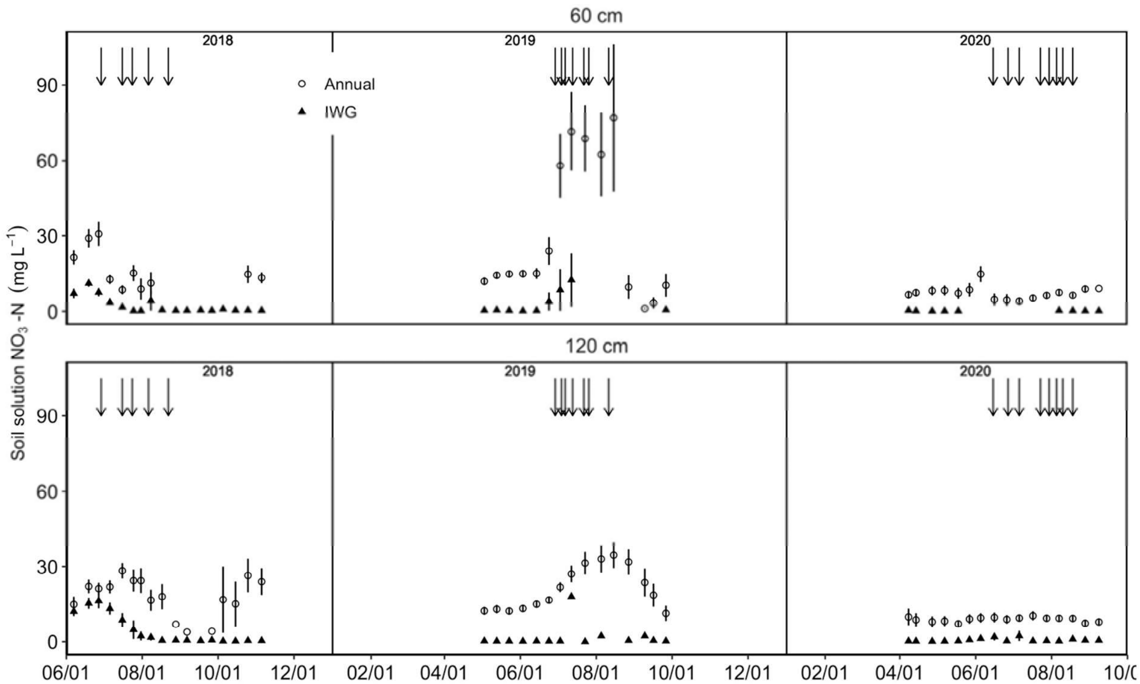

FIGURE 1
Mean soil solution NO_3^--N from lysimeters 60 and 120 cm below annual crops (soybean in 2018 and 2020, corn in 2019) and intermediate wheatgrass (IWG) sampled during the growing season in 2018, 2019, and 2020. Points are means from two lysimeters per treatment replicate and six replicates ($n = 12$), while error bars indicate standard error of the mean. Arrows indicate dates of irrigation events.

Crop yields

Crop yields were estimated each year from 2018 to 2020. Samples were taken in August of each year when the IWG had reached physiological maturity from two 76 by 76 cm quadrats with a total area of 0.58 m^2. Seed heads were removed from all IWG plants within the quadrat by cutting approximately 2 cm below the basal spikelet. After seed heads were removed, all remaining IWG biomass was harvested to an 8 cm stubble height. The remaining biomass was mechanically harvested and removed from the plots following quadrat sampling.

Biomass and seed heads were dried at 35°C for 72 h or until constant mass before being weighed. Grain was removed from spikes using a Wintersteiger LD 350 laboratory thresher (Wintersteiger; www.wintersteiger.com/us/Plant-Breeding-and-Research). Grain was separated from the chaff and other debris by hand-sieving and with a fractionating aspirator (Carter-Day International, Inc.; http://www.carterday.com).

Corn and soybean yields were determined by harvesting a subsection of the middle two rows of each plot. For corn, two 2-m sections of rows were cut from each corn plot. The number of corn stalks cut was recorded for each plot. All ears from the cut stalks were collected, dried (35°C for 72 h), shelled, and both cobs and kernels were weighed. Three stalks from each row section were randomly selected, dried, and weighed to estimate stover mass. Soybean yields were determined by harvesting whole plants from two 1-m sections of rows from each plot, followed by drying, threshing, and weighing. Following harvest for yield measurement, the remaining corn and soybean plants were mechanically harvested and removed from the plots.

Root biomass

Root biomass samples were taken in September 2020 with two 5-cm diameter manual push cores per plot at depths of 0–15, 15–30, 30–45, and 45–60 cm. Roots were separated from soil and debris using a hydropneumatic elutriation system (Smucker et al., 1982), then removed manually from sieves using tweezers. Due to the difficulty of distinguishing live from dead roots, no effort was made to separate them. Roots were dried at 35°C for 72 h. Samples were checked after drying for any remaining sand and debris, which was removed before weighing.

Soil solution NO_3^--N concentration

Soil solution NO_3^--N concentrations were determined by collecting soil solution samples with suction lysimeters. Lysimeters consisted of a porous ceramic end cap, a PVC tube, and an airtight rubber stopper (Jungers et al., 2019). Two pairs of 60 and 120 cm lysimeters were installed in each plot. Samples were collected every 7–10 days from April to October each year and analyzed by depth for soil solution NO_3^--N concentration using a colorimetric assay with a HACH DR 6000 spectrophotometer (Hach, https://www.hach.com).

Soil water content

Soil water content was measured on four dates in 2019 (June 17, July 19, August 21, October 31) and six dates in 2020 (May 12, June 23, July 15, August 5, September 1, and September 25) at 10, 20, 30, 40, 60, and 100 cm using a Delta-T Devices PR2/6 Probe (Delta-T Devices, 2021).

Statistical analysis

Analysis of variance (ANOVA) was conducted using mixed effects models to explain variation in soil water NO_3^--N concentration, soil water content, soil extractable NO_3^--N, root biomass, and crop yields. Predictor variables for the ANOVA were cropping system, depth (for soil variables), and their interaction. Years were analyzed separately because the annual crop varied. Cropping system was treated as a categorical variable; depth was treated as a categorical variable for root biomass and soil extractable NO_3^--N. Soil solution NO_3^--N concentrations from the 60 and 120 cm depths were not statistically different, based on preliminary statistics, and thus were averaged for the analysis. The treatment applied to the nearest neighboring plot was included in the model as a covariate to account for possible lateral movement of N applied to the neighboring plot. Data were analyzed with block as a random effect. For the soil solution NO_3^--N, which included two pairs of lysimeters per plot, plot was nested within block in the random effects structure. An autoregressive 1 correlation structure was fit to the model to account for temporal correlation in sample results within each plot. Analysis of variance was used to explain variation in soil water content for each sampling date, with a model including treatment, depth, and their interaction. Total water content from 0 to 100 cm was calculated for each plot and date using trapezoidal integration (Hupet et al., 2004) and compared among treatments using ANOVA. Mean comparisons using Tukey's adjusted P-value were used to generate estimated means for effects. Statistical analysis was carried out using statistical software program R (Version 3.5.2 GUI 1.70) including *emmeans* and *nlme* packages (R Core Team, 2018; Length, 2019; Pinheiro et al., 2019).

Results

Soil solution NO_3^--N concentration

Annual average soil solution NO_3^--N concentration differed by cropping system treatment in 2018 ($P < 0.001$), 2019 ($P = 0.004$), and 2020 ($P = 0.003$; Table 2; Figure 1), but did not vary by sampling depth or show an interaction effect in any year ($P > 0.05$). The average soil solution NO_3^--N concentration was 77%, 96%, and 96% lower in the perennial system than the annual system in Years 1–3, respectively (Table 2).

Throughout the seasons, both intra- and inter-annual variation was observed (Figure 1). Soil solution NO_3^--N concentrations under IWG initially had mean values between 10 and 20 mg L^{-1} in Year 1 but declined to nearly zero by the end of July 2018 and remained at those levels for all 3 years except for occasional deviations. In 2018, soil solution NO_3^--N concentrations under soybean were initially high at levels above 20 mg L^{-1}, declining to near zero in mid-September, but increasing to early season levels after harvest. In 2019, however, soil solution NO_3^--N concentrations under corn were between 10 and 20 mg L^{-1} but spiked to levels over twice that between late June and late August. Concentrations slowly declined over the remainder of the year. In 2020, mean soil solution NO_3^--N concentrations under soybean were consistently around 10 mg L^{-1}.

Soil extractable NO_3^--N

There was an effect of cropping system treatment, depth, and a depth by treatment interaction (p-values < 0.001) on soil extractable NO_3^--N measured at the end of the study in 2020 (Table 3). Soil NO_3^--N was greater in the annual cropping system compared to IWG at 0–15, 15–30, and 30–45 cm depths at the end of the study ($P < 0.001$), but extractable NO_3^--N levels were similar among treatments at the 45–60 cm depth. Soil extractable NO_3^--N was greatest at the 0–15 cm depth below the annual crops and decreased with each depth interval until 45–60 cm, which was similar to the 30–45 cm depth interval. There was no difference in soil extractable NO_3^--N across depths beneath the IWG.

Root biomass

Root biomass collected at the end of the study in 2020 was affected by treatment ($P = 0.006$), depth ($P < 0.001$), and a treatment by depth interaction ($P < 0.001$). Root biomass was

TABLE 2 Average soil solution NO_3^--N, grain, and biomass yields in the annual and IWG systems in 2018, 2019, and 2020.

	2018		2019		2020	
	Annual	IWG	Annual	IWG	Annual	IWG
Soil solution NO_3^--N (mg L^{-1})	19.0a	4.3b	22.1a	0.8b	7.8a	0.3b
Grain yield (Mg ha^{-1})	3.05a	0.85b	7.33a	0.43b	1.98a	0.22b
Biomass yield (Mg ha^{-1})	2.43b	4.12a	5.85	5.41	2.86b	4.41a

Crops in the annual system were soybean, corn and soybean in 2018, 2019, and 2020, respectively. Soil solution NO_3^--N were averaged across depths. Lower-case letters denote statistical significance between treatments at $P < 0.05$ within each year.

TABLE 3 Mean root biomass and soil extractable nitrate (mg NO_3^--N kg $soil^{-1}$) at four depth intervals from 0 to 60 cm at the end of the study in 2020.

	Root biomass (Mg ha^{-1})		Soil extractable nitrate (mg NO_3^--N kg $soil^{-1}$)	
	Annual	IWG	Annual	IWG
0–15	1.69b	8.57aA	2.77aA	0.17b
15–30	0.42b	2.82aB	1.38aB	0.00b
30–45	0.25	1.30B	0.48aC	0.00b
45–60	0.17	1.03B	0.25C	0.00

Letters denote statistical significance at $P < 0.05$; lower-case indicates difference between treatments; upper-case indicates difference between depths.

greater under IWG compared to the annual cropping system at all depths (Table 3). Soybean root biomass was 80%, 85%, 81%, and 83% lower than IWG root biomass at 0–15, 15–30, 30–45, and 45–60 cm, respectively. Summed over all the depths, total IWG root biomass was 13.73 Mg ha^{-1} while soybean root biomass was 2.54 Mg ha^{-1}, 82% lower ($P < 0.001$).

Crop yield

Grain yield was higher for the annual crops than for IWG in all years ($P < 0.001$, Table 2). Intermediate wheatgrass vegetative biomass yields (Table 2) were higher than soybean in 2018 ($P = 0.001$) and 2020 ($P = 0.009$) but similar to corn in 2019 ($P = 0.322$).

Soil water content

Of the four dates when soil water content was measured in 2019, there were very few effects of treatment, depth, or an interaction. Dates had a significant treatment by depth interaction. There was a main effect of cropping system treatment on soil water content on July 19 and October 31 ($P < 0.001$), in which soil water content was greater beneath the annual cropping system (0.09 m m^{-3}) compared to the perennial (0.03 m m^{-3}) on July 19 but lower in the annual (0.04 m m^{-3}) compared to the perennial (0.05 m m^{-3}) on October 31, 2019. In 2020, soil water content varied by treatment on June 23 ($P < 0.001$), in which soil water content was greater beneath the annual compared to the perennial. There was a significant interaction between treatment and depth on three other dates in 2020. Soil water content by treatment and depth is shown in Figure 2 to illustrate the interaction.

Discussion

Soil solution NO_3^--N concentration

Consistent with previous findings, we observed drastically lower concentrations of soil solution NO_3^--N beneath IWG compared to the annual cropping system (Figure 1). Concentrations under IWG were initially between 10 and 20 mg L^{-1} during June and July of 2018, the first spring after seeding, but approached zero by August and remained very low for the duration of the experiment. A previous study in Minnesota found that soil solution NO_3^--N beneath IWG averaged 0.09–0.3 mg L^{-1} when fertilized with 80 kg N ha^{-1} (Jungers et al., 2019). Despite only receiving 20 kg N ha^{-1} more fertilizer annually in this study, annual average soil solution NO_3^--N concentrations ranged from 4.3 mg L^{-1} in the first-year to 0.3 mg L^{-1} at the end of the study. Higher NO_3^--N concentrations found in this study compared to previous finding in Minnesota could be related to the potentially higher drainage rate associated with coarse structured soil at our study. These relatively higher soil solution NO_3^--N levels observed in the first-year of our study were also likely attributable to lower root biomass during stand establishment and thus reduced ability to capture and assimilate soil solution NO_3^--N. In line with this thinking, our results were similar to a study on sandy soil in

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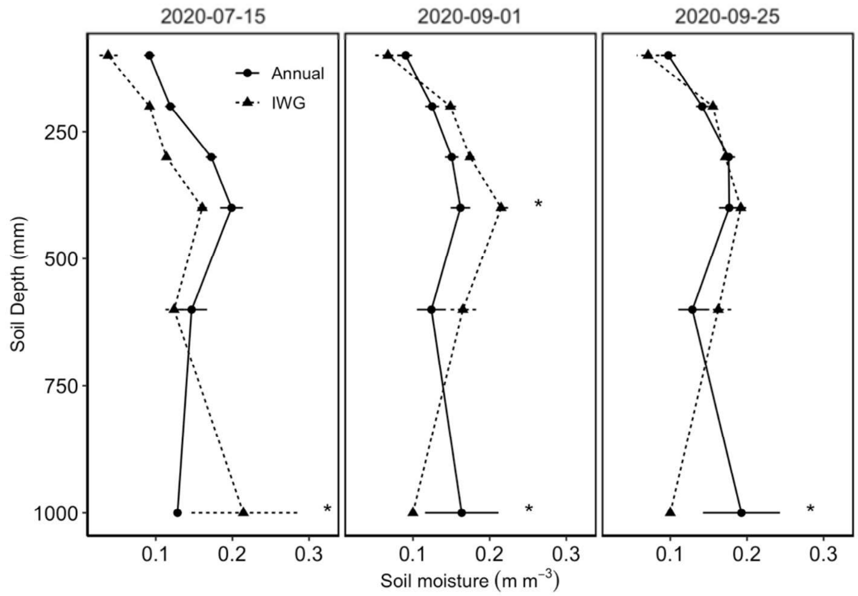

FIGURE 2
Soil water content beneath soybean (annual) and intermediate wheatgrass (IWG) on days when there was a significant interaction between cropping system treatment and depth. Asterisks indicate depths for which soil water content differed significantly between cropping systems.

Michigan during stand establishment (Culman et al., 2013b). Despite the slightly higher soil solution NO_3^--N concentrations observed in Year 1 here and on other sandy soils, values were comparable to mixtures of perennial grasses and forbs found in CRP plantings (Randall et al., 1997) and consistently below the EPA safe drinking standard of 10 mg L^{-1}.

Average annual soil solution NO_3^--N concentrations beneath the annual crops were similar to or slightly lower than those reported by other studies in Minnesota. During the corn phase of the annual rotation, our annual soil solution NO_3^--N of 22.1 mg L^{-1} was similar to findings by Ochsner et al. (2017), who reported an average soil solution NO_3^--N of 21.2 mg L^{-1} beneath a corn-soybean rotation with corn phases fertilized at 146 kg N ha^{-1} as urea annually. In another study also conducted on coarse-structured soils in Minnesota, Struffert et al. (2016) reported an average annual soil solution NO_3^--N concentration of 18.8 mg L^{-1} beneath soybean, and determined that soil solution NO_3^--N during the soybean phase was not affected by N fertilizer rates applied to corn the previous year.

This is also among the first studies to compare soil solution NO_3^--N levels of fertilized IWG to an unfertilized legume crop. Despite applying 100 kg N ha^{-1} of urea annually to the IWG, lower soil solution NO_3^--N concentration were observed in the IWG compared to the unfertilized soybean. Biologically fixed N may have been mineralized after exudation or sloughing of soybean roots, which may have contributed to higher soil solution NO_3^--N levels compared to IWG. The elevated soil solution NO_3^--N in the soybean could also have originated from N fertilizer applied during the previous crops. However, as previously mentioned, N fertilizer rates applied to a previous corn crop did not affect soil solution NO_3^--N beneath subsequent soybean (Struffert et al., 2016). Significant N demand by IWG may have also contributed to the large difference in soil solution NO_3^--N.

Soil extractable NO_3^--N

In addition to lower soil solution NO_3^--N concentration, we also found less extractable NO_3^--N in the soil after 3 years of IWG production compared to the annual rotation system. This suggests that the IWG assimilated NO_3^- more thoroughly from the soil than the annual rotation system, especially because the Year 3 crop was unfertilized soybean. Extractable NO_3^--N remaining in the soil is a major factor determining the concentration of dissolved NO_3^--N in soil solution, which in turn determines total leaching loads (Randall and Mulla, 2001; Culman et al., 2013a; Jungers et al., 2019).

The low levels of extractable NO_3^--N under IWG also suggest that the plants may have been N-limited, despite being fertilized at the high end of optimal rates (Jungers et al., 2017). Nitrogen removal during IWG grain and biomass harvest can exceed 150 kg N ha^{-1} in the first-year of production (Crews et al., 2022; Tautges et al., 2018). Intermediate wheatgrass tissue N concentrations at the time of grain harvest in Minnesota peaked above 10 g N kg^{-1} biomass and declined with stand age (Jungers et al., 2017). If tissue N concentrations were similar to previous studies in MN, removal rates could have been between 46 and 58 kg N ha^{-1} $year^{-1}$, thus less than the N applied as fertilizer (100 kg N ha^{-1}). However, total N demand may have been greater to support root biomass production. If root tissue N was similar to previously reported estimates between 9 and 11 g N kg^{-1} (Dobbratz, 2019), then there would be another pool of nearly 130 kg N ha^{-1} in belowground root tissues. It is not known what fraction of root N is recycled during root death and mineralization of root biomass from year to year in an IWG system, but our results suggest that the N fertilizer applied was needed to support above and belowground IWG biomass and that little N was likely lost *via* leaching or left in the soil.

Root biomass

Root biomass is considered an important trait of perennial crops for providing ecosystem services such as reduced nitrate leaching to groundwater. Intermediate wheatgrass root biomass averaged 13.7 Mg ha^{-1} after the third-year of production, while soybean root biomass was 2.5 Mg ha^{-1} when sampled from 0 to 60 cm. These values are similar to other reported values for these crops. For example, Intermediate wheatgrass fertilized at 80 kg N ha^{-1} had root biomass of 4.10, 7.32, and 9.51 Mg ha^{-1} (0–60 cm depth) in Years 1–3 of a 3-year study, while a soybean-corn-soybean rotation had root biomass of 2.22, 2.93, and 2.30 Mg ha^{-1} in Years 1–3 (Bergquist, 2019). Root biomass accumulation over time allows IWG to more effectively capture NO_3^--N before it reaches depths below the rooting zone where it is subject to leaching to groundwater. Nearly 63% of the IWG root biomass was found in the top 0–15 cm depth. Previous work has reported IWG belowground biomass to be 3.28 Mg ha^{-1} in the first 10 cm, on average, in Minnesota and Wisconsin (Sakiroglu et al., 2020). In an intra-annual study of root biomass beneath IWG, total root biomass from 0 to 20 cm peaked between 3.5 and 4 Mg ha^{-1} in June and July before declining to 1 Mg ha^{-1} at the end of the growing season (Pugliese et al., 2019). This concentration of root biomass at shallow depths also likely increases NO_3^--N capture and consequently reduce soil solution NO_3^--N below the rooting zone.

Soil water content

We found inconsistent differences in soil water content between annual crops and IWG. In the second-year of the study, soil water content was greater beneath the corn compared to the IWG in July, perhaps because IWG biomass would have been approaching peak biomass and thus been demanding more water than corn. A similar early-season pattern was found in Year 3 when soil water content was greater beneath the soybean compared to the IWG when measured in June. By the end of Year 2 (October), soil water content was greater in IWG compared to corn. Only in Year 3 did we observe any differences in soil water content by depth across treatments (Figure 2). In July, soil water content was greater beneath IWG compared to soybean at the deepest measured depth of 1,000 mm. This treatment effect was opposite at the 1,000 mm depth in September, where soil moisture content was greater for the soybean compared to IWG. Our results do match those from previous studies. In one comparison of perennial and annual systems, soil water content beneath *Miscanthus* and switchgrass was lower than a corn-soybean rotation earlier in the season, but the treatment effect flipped later in the season when switchgrass had higher soil water content (McIsaac et al., 2010). It has also been observed that soil water content tended to be higher under annuals than semi-perennials, and that there was less drainage from semi-perennials and perennials than annuals (Ferchaud and Mary, 2016). In studies with IWG, researchers have reported less in soil water content under IWG compared to annual wheat (Culman et al., 2013b) and corn (Jungers et al., 2019).

Soil water content can be used to make inferences on transpiration and drainage, the latter being an important component of nitrate leaching. The timing and frequency of our soil water content measurements precluded us from determining if both treatments had similar ET and drainage rates. Irrigation at our experiment could also have minimized our ability to detect differences in soil water content from plant ET. It is also established that greater root biomass increases water and nutrient uptake, which could reduce soil water content (Ehdaie et al., 2010; Matsunami et al., 2012; Carvalho et al., 2014). In our study, the similar soil moisture contents observed in the perennial and annual treatments may have been a function of the low water holding capacity of the sandy soil, which may have promoted drainage regardless of root biomass.

Grain and biomass yields

Intermediate wheatgrass grain yields at our sandy site were comparable to previous reports from sites with higher soil fertility levels. Under similar fertilizer treatments, reported first-year values range from 763 kg ha^{-1} (Zimbric et al., 2020) to 1,089 kg ha^{-1} at sites in Wisconsin (Favre et al., 2019) and from 893 kg ha^{-1} (Jungers et al., 2017) to 1,150 kg ha^{-1} (Fernandez et al., 2020) in Minnesota. Second- and third-year yields tend to be much lower, typically ranging from 150 kg ha^{-1} (Fernandez et al., 2020) to 630 kg ha^{-1} (Sakiroglu et al., 2020) in Year 2 and from 153 kg ha^{-1} (Jungers et al., 2017) to 371 kg ha^{-1} (Zimbric et al., 2020) in Year 3. Our yields suggest that this soil type and climate is appropriate for IWG grain and biomass production with irrigation.

Forage production is important for profitable IWG systems, since a major challenge of IWG grain production is the substantial yield declines in later years of production (Jungers et al., 2017; Pugliese et al., 2019; Hunter et al., 2020a). Intermediate wheatgrass biomass yields in this study included the stems and leaves that were remaining after grain harvest soon after peak productivity. Biomass harvested at this time, after physiological maturity, is relatively low in terms of forage quality compared to IWG biomass harvested at vegetative stages, but high compared to annual small grain biomass after grain harvest (Hunter et al., 2020b). Intermediate wheatgrass biomass yields were similar to those of other reports in Minnesota, though they were at the lower end of the range. Reported summer aboveground biomass values include 5,130 kg ha^{-1} in the second-year and 5,850 kg ha^{-1} in the third-year for IWG fertilized at 90 and 134 kg ha^{-1} in Wisconsin and 10,600 kg ha^{-1} for third-year stands in Minnesota (Sakiroglu et al., 2020). Similarly, summer yields of approximately 6,200 kg ha^{-1} were reported for first-year monocultures fertilized at 100 kg N ha^{-1} as urea (Favre et al., 2019). Biomass yields averaged 13,400 to 14,320 kg ha^{-1} for control treatments in a management study fertilized at 56 kg ha^{-1} the previous year (Pinto et al., 2021). Our results support that understanding that post-grain harvest biomass yields can be high enough for growers to consider harvesting for used as feed or straw on the farm or marketed for an additional revenue stream.

Conclusion

We found that soil solution NO_3^--N concentrations were 77–96% lower under IWG than the annual corn-soybean rotation, even in the unfertilized soybean phase of rotation, but soil water content was similar. This suggests that the IWG captured and utilized a greater proportion of soil solution NO_3^--N, which is also demonstrated by very low residual soil extractable NO_3^--N levels at the end of the experiment relative to the annual crops. The lower NO_3^--N concentrations in soil solution would be expected to translate to reductions in total leaching load of a similar magnitude. The increased uptake of N by IWG was likely facilitated by its greater root biomass, which was 5.4 times higher than that under the annual system. Despite the challenges associated with production of IWG on low-fertility sandy soils, grain yields were comparable to other locations and the system would likely be profitable in the first-year for grain alone. Biomass yields would support additional revenue streams in subsequent years to improve economic viability, and together our study provides evidence that IWG could be a good option for coarse textured soils that are prone to nitrate pollution.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

ER: collected data, analyzed data, wrote the first draft of the manuscript, and contributed to the final draft of the manuscript. JG: oversaw soil sample processing and contributed to the final draft of the manuscript. CS: designed the field experiment and contributed to the final draft of the manuscript. JJ: acquired funding, designed the field experiment, oversaw field sampling and data collection, data visualization, and contributed to the final draft of the manuscript. All authors contributed to data interpretation, manuscript writing, and revision.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

References

Asbjornsen, H., Hernandez-Santana, V., Liebman, M., Bayala, J., Chen, J., Helmers, M., et al. (2014). Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. *Renew. Agric. Food Syst.* 29, 101–125. doi: 10.1017/S1742170512000385

Bajgain, P., Zhang, X., Jungers, J. M., DeHaan, L. R., Heim, B., Sheaffer, C. C., et al. (2020). 'MN-Clearwater', the first food-grade intermediate wheatgrass (Kernza perennial grain) cultivar. *J. Plant Regist.* 14, 288–297. doi: 10.1002/plr2.20042

Bergquist, G. (2019). Biomass Yield and Soil Microbial Response to Management of Perennial Intermediate Wheatgrass (Thinopyrum *intermedium*) as Grain Crop and Carbon Sink. Master's Theses, University of Minnesota.

Brender, J. D., Weyer, P. J., Romitti, P. A., Mohanty, B. P., Shinde, M. U., Vuong, A. M., et al. (2013). Prenatal nitrate intake from drinking water and selected birth defects in offspring of participants in the national birth defects prevention study. *Environ. Health Perspect.* 121, 1083–1089. doi: 10.1289/ehp.1206249

Carvalho, P., Azam-Ali, S., and Foulkes, M. J. (2014). Quantifying relationships between rooting traits and water uptake under drought in Mediterranean barley and durum wheat: root traits and water uptake. *J. Integr. Plant Biol.* 56, 455–469. doi: 10.1111/jipb.12109

Crews, T. E., Kemp, L., Bowden, J. H., and Murrell, E. G. (2022). How the nitrogen economy of a perennial cereal-legume intercrop affects productivity: can synchrony be achieved? *Front. Sustain. Food Syst.* 6:755548. doi: 10.3389/fsufs.2022.755548

Crews, T. E., and Peoples, M. B. (2005). Can the synchrony of nitrogen supply and crop demand be improved in legume and fertilizer-based agroecosystems? A review. *Nutr. Cycl. Agroecosystems* 72, 101–120. doi: 10.1007/s10705-004-6480-1

Culman, S. W., Snapp, S. S., Green, J. M., and Gentry, L. E. (2013a). Short- and long-term labile soil carbon and nitrogen dynamics reflect management and predict corn agronomic performance. *Agron. J.* 105, 493–502. doi: 10.2134/agronj2012.0382

Culman, S. W., Snapp, S. S., Ollenburger, M., Basso, B., and DeHaan, L. R. (2013b). Soil and water quality rapidly responds to the perennial grain Kernza wheatgrass. *Agron. J.* 105, 735–744. doi: 10.2134/agronj2012.0273

DeHaan, L., Christians, M., Crain, J., and Poland, J. (2018). Development and evolution of an intermediate wheatgrass domestication program. *Sustainability* 10:1499. doi: 10.3390/su10051499

Delta-T Devices (2021). *PR2 Profile Probe - Analogue Version*. Cambridge, UK. Available online at: www.delta-t.co.uk/product/pr2/

Dinnes, D. L., Karlen, D. L., Jaynes, D. B., Kaspar, T. C., Hatfield, J. L., Colvin, T. S., et al. (2002). Review and interpretation: nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. *Agron. J.* 94, 153–171. doi: 10.2134/agronj2002.1530

Dobbratz, M. (2019). *Perennial fuel, feed, and cereal: High diversity perennials for biofuel and intermediate wheatgrass for grain and forage*. PhD Dissertation retrieved from University of Minnesota Digital Conservancy. Available online at: https://hdl.handle.net/11299/211748

Ehdaie, B., Merhaut, D. J., Ahmadian, S., Hoops, A. C., Khuong, T., Layne, A. P., et al. (2010). Root system size influences water-nutrient uptake and nitrate leaching potential in wheat: root system and nutrient uptake in wheat. *J. Agron. Crop Sci.* 196, 455–466. doi: 10.1111/j.1439-037X.2010.00433.x

Erisman, J. W., Galloway, J. N., Seitzinger, S., Bleeker, A., Dise, N. B., Petrescu, A. M. R., et al. (2013). Consequences of human modification of the global nitrogen cycle. *Philos. Trans. R. Soc. B Biol. Sci.* 368, 20130116. doi: 10.1098/rstb.2013.0116

Favre, J. R., Castiblanco, T. M., Combs, D. K., Wattiaux, M. A., and Picasso, V. D. (2019). Forage nutritive value and predicted fiber digestibility of Kernza intermediate wheatgrass in monoculture and in mixture with red clover during the first production year. *Anim. Feed Sci. Technol.* 258, 114298. doi: 10.1016/j.anifeedsci.2019.114298

Ferchaud, F., and Mary, B. (2016). Drainage and nitrate leaching assessed during 7 years under perennial and annual bioenergy crops. *Bioenergy Res.* 9, 656–670. doi: 10.1007/s12155-015-9710-2

Fernandez, C. W., Ehlke, N., Sheaffer, C. C., and Jungers, J. M. (2020). Effects of nitrogen fertilization and planting density on intermediate wheatgrass yield. *Agron. J.* 112, 4159–4170. doi: 10.1002/agj2.20351

Feyereisen, G. W., Wilson, B. N., Sands, G. R., Strock, J. S., and Porter, P. M. (2006). Potential for a rye cover crop to reduce nitrate loss in Southwestern Minnesota. *Agron. J.* 98, 1416–1426. doi: 10.2134/agronj2005.0134

Hunter, M. C., Sheaffer, C. C., Culman, S. W., and Jungers, J. M. (2020a). Effects of defoliation and row spacing on intermediate wheatgrass I: Grain production. *Agron. J.* 112, 1748–1763. doi: 10.1002/agj2.20128

Hunter, M. C., Sheaffer, C. C., Culman, S. W., Lazarus, W. F., and Jungers, J. M. (2020b). Effects of defoliation and row spacing on intermediate wheatgrass II: forage yield and economics. *Agron. J.* 112, 1862–1880. doi: 10.1002/agj2.20124

Hupet, F., Bogaert, P., and Vanclooster, M. (2004). Quantifying the local-scale uncertainty of estimated actual evapotranspiration. *Hydrol. Process.* 18, 3415–3434. doi: 10.1002/hyp.1504

Jungers, J. M., DeHaan, L. H., Mulla, D. J., Sheaffer, C. C., and Wyse, D. L. (2019). Reduced nitrate leaching in a perennial grain crop compared to maize in the Upper Midwest, USA. *Agric. Ecosyst. Environ.* 272, 63–73. doi: 10.1016/j.agee.2018.11.007

Jungers, J. M., DeHaan, L. R., Betts, K. J., Sheaffer, C. C., and Wyse, D. L. (2017). Intermediate wheatgrass grain and forage yield responses to nitrogen fertilization. *Agron. J.* 109, 462–472. doi: 10.2134/agronj2016.07.0438

Keeler, B. L., Gourevitch, J. D., Polasky, S., Isbell, F., Tessum, C. W., Hill, J. D., et al. (2016). The social costs of nitrogen. *Sci. Adv.* 2, e1600219. doi: 10.1126/sciadv.1600219

Keeler, B. L., and Polasky, S. (2014). Land-use change and costs to rural households: a case study in groundwater nitrate contamination. *Environ. Res. Lett.* 9, 074002. doi: 10.1088/1748-9326/9/7/074002

Law, E. P., Wayman, S., Pelzer, C. J., Culman, S. W., Gómez, M. I., DiTommaso, A., et al. (2022). Multi-criteria assessment of the economic and environmental sustainability characteristics of intermediate wheatgrass grown as a dual-purpose grain and forage crop. *Sustainability.* 14, 3548. doi: 10.3390/su14063548

Length, R. (2019). *emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.3.3*. Available online at: https://CRAN.R-project.org/package=emmeans

Maeght, J.-L., Rewald, B., and Pierret, A. (2013). How to study deep roots—and why it matters. *Front. Plant Sci.* 4:299. doi: 10.3389/fpls.2013.00299

Matsunami, M., Matsunami, T., Ogawa, A., Toyofuku, K., Kodama, I., and Kokubun, M. (2012). Genotypic variation in biomass production at the early vegetative stage among rice cultivars subjected to deficient soil moisture regimes and its association with water uptake capacity. *Plant Prod. Sci.* 15, 82–91. doi: 10.1626/pps.15.82

McIsaac, G. F., David, M. B., and Mitchell, C. A. (2010). *Miscanthus* and switchgrass production in Central Illinois: impacts on hydrology and inorganic nitrogen leaching. *J. Environ. Qual.* 39, 1790–1799. doi: 10.2134/jeq2009.0497

Ochsner, T. E., Schumacher, T. W., Venterea, R. T., Feyereisen, G. W., and Baker, J. M. (2017). Soil water dynamics and nitrate leaching under corn–soybean rotation, continuous corn, and kura clover. *Vadose Zone J.* 17, 1–11. doi: 10.2136/vzj2017.01.0028

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., and and, R., Core Team (2019). *nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-139,*. Available online at: https://CRAN.R-project.org/package=nlme

Pinto, P., De Haan, L., and Picasso, V. (2021). Post-harvest management practices impact on light penetration and Kernza intermediate wheatgrass yield components. *Agronomy* 11, 442. doi: 10.3390/agronomy11030442

Pugesgaard, S., Schelde, K., Larsen, S. U., Laerke, P. E., and Jørgensen, U. (2015). Comparing annual and perennial crops for bioenergy production - influence on nitrate leaching and energy balance. *GCB Bioenergy* 7, 1136–1149. doi: 10.1111/gcbb.12215

Pugliese, J. Y., Culman, S. W., and Sprunger, C. D. (2019). Harvesting forage of the perennial grain crop Kernza (*Thinopyrum intermedium*) increases root biomass and soil nitrogen cycling. *Plant Soil* 437, 241–254. doi: 10.1007/s11104-019-03974-6

R Core Team (2018). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing Available online at: https://www.R-project.org/

Randall, G. W., Huggins, D. R., Russelle, M. P., Fuchs, D. J., Nelson, W. W., and Anderson, J. L. (1997). Nitrate losses through subsurface tile drainage in Conservation Reserve Program, Alfalfa, and row crop systems. *J. Environ. Qual.* 26, 1240–1247. doi: 10.2134/jeq1997.00472425002600050007x

Randall, G. W., and Mulla, D. J. (2001). Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices. *J. Environ. Qual.* 30, 337–344. doi: 10.2134/jeq2001.302337x

Sakiroglu, M., Dong, C., Hall, M. B., Jungers, J., and Picasso, V. (2020). How does nitrogen and forage harvest affect belowground biomass and nonstructural carbohydrates in dual-use Kernza intermediate wheatgrass? *Crop Sci.* 60, 2562–2573. doi: 10.1002/csc2.20239

Smucker, A. J. M., McBurney, S. L., and Srivastava, A. K. (1982). Quantitative separation of roots from compacted soil profiles by the hydropneumatic elutriation system. *Agron. J.* 74, 500–503. doi: 10.2134/agronj1982.00021962007400030023x

Struffert, A. M., Rubin, J. C., Fernández, F. G., and Lamb, J. A. (2016). Nitrogen management for corn and groundwater quality in Upper Midwest irrigated sands. *J. Environ. Qual.* 45.1557–1564. doi: 10.2134/jeq2016.03.0105

Tautges, N. E., Jungers, J. M., DeHaan, L. R., Wyse, D. L., and Sheaffer, C. C. (2018). Maintaining grain yields of the perennial cereal intermediate wheatgrass in monoculture v. bi-culture with alfalfa in the Upper Midwestern USA. *J. Agric. Sci.* 156, 758–773. doi: 10.1017/S0021859618000680

USDA–NRCS (2021). *Web Soil Survey. USDA – Natural Resources Conservation Service*. Available online at: https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm

Ward, M., Jones, R., Brender, J., de Kok, T., Weyer, P., Nolan, B., et al. (2018). Drinking water nitrate and human health: an updated review. *Int. J. Environ. Res. Public Health* 15, 1557. doi: 10.3390/ijerph15071557

Ward, M. H., Kilfoy, B. A., Weyer, P. J., Anderson, K. E., Folsom, A. R., and Cerhan, J. R. (2010). Nitrate intake and the risk of thyroid cancer and thyroid disease. *Epidemiology* 21, 389–395. doi: 10.1097/EDE.0b013e3181d6201d

Zimbric, J. W., Stoltenberg, D. E., and Picasso, V. D. (2020). Effective weed suppression in dual-use intermediate wheatgrass systems. *Agron. J.* 112, 2164–2175. doi: 10.1002/agj2.20194

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Financial comparison of seven nitrate reduction strategies for Midwestern agricultural drainage

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ABSTRACT

Much work has been invested in the development of practices and technologies that reduce nitrate losses from agricultural drainage in the US Midwest. While each individual practice can be valuable, the effectiveness will be site specific and the acceptability of each approach will differ between producers. To enhance decision making in terms of water quality practices, this work created average cost effectiveness parameters for seven nitrate management strategies (controlled drainage, wetlands, denitrification bioreactors, nitrogen management rate and timing, cover crops, and crop rotation). For each practice, available published cost information was used to develop a farm-level financial model that assessed establishment and maintenance costs as well as examined financial effects of potential yield impacts. Then, each practice's cost values were combined with literature review of N reduction (% N load reduction), which allowed comparison of these seven practices in terms of cost effectiveness (dollars per kg N removed). At −\$14 and −\$1.60 kg N^{-1} yr^{-1}, springtime nitrogen application and nitrogen application rate reduction were the most cost effective practices. The in-field vegetative practices of cover crop and crop rotation were the least cost effective (means: \$55 and \$43 kg N^{-1} yr^{-1}, respectively). With means of less than \$3 kg N^{-1} yr^{-1}, controlled drainage, wetlands, and bioreactors were fairly comparable with each other. While no individual technology or management approach will be capable of addressing

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drainage water quality concerns in entirety, this analysis provides measures of average cost effectiveness across these seven strategies that allows direct comparison.

1. Introduction

Artificial subsurface drainage systems in the Midwestern "Corn Belt" region have allowed for increased productivity over the past century [1], but nitrate (NO_3^-) losses in drainage have caused significant multi-scale environmental concerns [2,3]. Much work has been done developing and advancing practices to reduce NO_3^- losses in subsurface agricultural drainage. Dinnes et al. [1] provide a comprehensive review of NO_3^- reducing technologies for the Midwest including in-field "preventative" N strategies (e.g., N management, cover crops, diversified rotations) and "remedial" strategies for N removal from drainage (e.g., controlled drainage, bioreactors, wetlands). While each strategy and individual practice can be valuable, the NO_3^- removal effectiveness will be site specific and the acceptability of each individual approach will differ between producers. Nevertheless, no individual technology or management approach will be capable of addressing drainage water quality concerns in entirety [1,4]; as such, a suite of approaches used across these landscapes will be required [5].

On an individual farmer basis, adoption of environmental management practices designed to mitigate or prevent issues such as NO_3^- losses through drainage to surface waters are motivationally different from production innovations largely because short-term economic advantages of adopting a mitigation technology are rare [6,7]. Farm level action involving use of technology is in large part influenced by owner and operator beliefs and attitudes (i.e., regarding environmental and financial risk) in combination with personal environmental goals and knowledge about technology [8]. Perceptions of a technology in turn are shaped by external factors such as cost, overall complexity and effectiveness of the available technology, and available technical/financial support [9,10]. As such, crop producers require comprehensive information about water quality technologies with regard to the context for use, operational parameters, performance efficacy, and the full range of financial parameters (e.g., upfront and long-term costs). Of particular and universal concern for farmers is the financial feasibility of a particular technology in the context of their production system, as well as comparative advantage across technology-based management options. Moreover, comprehensive financial information is needed to calibrate agricultural conservation cost-share programming and targeting and to better guide federal and state technical service provision at county levels [4].

To enhance land-use decision making, this work investigates and makes transparent the financial parameters of seven NO_3^- management strategies; three are remedial N strategies: controlled drainage, wetlands, denitrification bioreactors and four are preventative N strategies: N rate reduction, spring N application, cover crops, and crop rotation. It bears to note early-on; however, that the Midwest is a heterogeneous region where not every abatement strategy will be equally appropriate (i.e., costly or effective) in any given situation. Suitability, in addition to NO_3^- reduction effectiveness, can vary by soil type, topography, landscape position, and microclimate (e.g., rainfall patterns, winter severity) for each of the seven distinct practices investigated here. For example, winter cover crops may be more difficult to establish in northern Minnesota vs. southern Indiana, and controlled drainage will be most cost effective on flatter topographies. The assumed baseline cropping system for this work was a corn/soybean rotation, reflective of the Midwestern agricultural landscape [11], and because tillage generally has a relatively small impact upon tile drainage NO_3^- export [12], it was not included as a variable here.

Controlled drainage (also known as drainage water management) is a strategy that addresses agricultural NO_3^- loading through the use of a series of structures installed in drainage pipes or drainage ditches that allow control of the water table depth [13,14]. Though this practice can be used to achieve agronomic and/or environmental objectives [14], a major limitation is that controlled drainage becomes more expensive on slopes greater than 0.5–1% [1,15]. The second practice under consideration here, denitrification or woodchip bioreactors, uses control structures to regulate drainage water flowing through an excavation (typically >30 m long, >1 m wide) filled with a carbon source allowing enhanced denitrification of the NO_3^- in the drainage water [16,17]. These systems have been tested for treating drainage from "field-sized" areas of approximately 20 ha and usually require very little to no land to be removed from production by fitting in grassed edge-of-field areas [17]. The third of the remedial strategies, constructed wetlands, is a long-term NO_3^- reduction strategy intended for watershed-scale treatment [18,19]. A key consideration for N removal in wetlands is the wetland to treatment area ratio with increased N removal possible at increased wetland: watershed area ratios [18,20–22].

Regarding in-field, preventative practices, N fertilizer management, here in terms of rate and timing, is one of the farm operator-controlled factors to reduce N losses in agricultural drainage [1,12,23,24]. Water quality benefits of reduced application rates will be a function of the original and the modified rate [25,26]. Lawlor et al. [27] proposed that a corn (*Zea mays* L.) and soybean (*Glycine max* (L.) Merr.) rotations can be described with:

$$\text{N Concentration in Drainage} = 5.72 + 1.33e^{(0.0104 \times N\ Rate)} \quad (1)$$

where N concentration is in mg N L^{-1} and rate is in kg N applied ha^{-1} [27]. Spring N application in the U.S. Midwest more closely synchronizes the application with plant uptake [28,29], an outcome that is preferable from both water quality and agronomic perspectives [24]. Nevertheless, fall N applications are a way to manage risk associated with uncertain spring weather and spring-time field activities [30].

The "preventative" strategy of winter cover crops such as rye, oat, winter wheat, brassica, or winter-hardy legumes, utilizes plant uptake as the major water quality improvement mechanism [31,32]. Benefits of cover crops (as well as several of the other practices) extend beyond drainage water NO_3^- reduction (e.g., erosion control, pest control, enhancement of soil productivity) [29,33] but were not included here as this analysis focuses solely on NO_3^- reduction; see Table 1 for abbreviated comments and Christianson et al. [34] for a broad discussion of ecosystem services associated with the use of any of these seven practices. The main limitations of winter cover crops are that they need to grow well under non-ideal conditions [1,32], some need to be killed before planting the main crop, and a corn yield reduction following certain covers is possible [31,32]. The final practice, crop rotations that include perennials, similarly provides water quality benefits via N and water uptake [1,35] and additional benefits to the soil [36]. Although the main limitations for this sort of rotation include access to markets, crop storage, and additional machinery requirements, Dinnes [29] reported diversifying cropping systems in Iowa has the most potential to reduce NO_3^- loadings compared to any other best management practice.

The objectives of this exploratory financial assessment are two-fold: (1) characterize and quantify the financial (cost) parameters of the seven NO_3^- reduction strategies; and (2) explore and compare the average cost efficiency of each strategy (dollars per kg N removed) using published measures of N reduction effectiveness. The primary motivation of this work is that while cost assessment of this type is fairly straight-forward, cost comparison analysis across various agricultural best management practices is invariably challenged [37] by (1) limited availability of published cost information, (2) variable methodology in published financial assessments, (3) limited methodological transparency in published cost assessments, (4) variable discount rates, (5) inconsistent analysis horizons due to variable life spans or management horizons, and (6) many costs are often site specific and therefore can exhibit significant ranges. This analysis is therefore an attempt to make transparent the structure and timing of cost parameters associated with using any of these NO_3^- management strategies, and to develop comparable measures of average cost effectiveness across these seven NO_3^- management strategies. Nevertheless, we recognize an inherent limitation of this work arises from the site-specific nature of the practices being compared; their application at different sites and under different conditions will necessarily confound a comparison of their effectiveness in reducing N loads and hence their calculated cost efficiencies.

Table 1
Description of the scenarios, uncertainty ranges for the Total Present Value Costs, and the additional benefits and costs that were not quantified for seven nitrogen reduction practices for agricultural drainage; see Christianson et al. [34] for more specific discussion of ecosystem services of these practices.

Practice	Practicable lifespan (yr)	Specific scenario	Uncertainty of ranges for TPVC	Unquantified costs and benefits
Controlled drainage	40	1 structure per 4 ha–8 ha	Low uncertainty	Potential yield impacts Potential increase in soil erosion, soil compaction, or surface runoff
Bioreactor	40	20.2 ha field treated with a 0.1 ha bioreactor	Low uncertainty	None
Wetland	50	405 ha treated by a 4 ha wetland plus buffer	Moderate uncertainty due to predominance of land cost and the variability of this factor	Additional ecosystem services including pollination, wood fuel, ornamental resources, natural hazard regulation, and recreation
N rate reduction	1	168 kg N ha^{-1}–140 kg N ha^{-1}	Large uncertainty due to yield impact variability	Probabilistic variability of yields
N spring application	1	Apply N in spring instead of fall	Large uncertainty due to unquantified risk and yield impact variability	Cost of infrastructure potentially required for fertilizer storage, handling, etc. Probabilistic variability of yields Potential loss of yield by a delayed planting date
Cover crop	4	Rye drilled	Large uncertainty as this practice is primarily implemented for reasons other than N reduction and due to yield impact variability	Additional ecosystem services including pollination and erosion and pest regulation; Potential future yield enhancement due to cover crop-induced soil quality and organic matter enhancement
Rotation	10	3 years alfalfa, 2 years corn	Very large uncertainty due to rotation complexity and the variability of alfalfa-induced yield increase	Additional ecosystem services including pollination and erosion and pest regulation; Potential future yield enhancement due to perennial-induced soil quality and organic matter enhancement

2. Materials and methods

There is limited availability of published cost information regarding drainage NO_3^- reduction strategies, and the variable methodology and limited transparency for the studies that have been done in this area make comparison between published analyses difficult. The timing of costs particularly complicates comparisons of water quality practices. For example, controlled drainage, bioreactors, and wetlands all have large initial capital outlays and intermittent management costs, while N management, cover crops, and crop rotations largely involve variable annual costs. Cost assessments have been carefully constructed for all seven practices with itemized cost parameters and unit cost data for each strategy collected from various secondary sources (e.g., published literature, published custom rate surveys, and when necessary personal communication with knowledgeable individuals). Total present value costs (TPVCs) were assessed with a discounted cost model that aggregates total fixed and variable costs.

$$\mathrm{TPVC}_{\mathrm{practice}} = C_{\mathrm{est,practice}} \text{ in year } 1 + C_{\mathrm{main}} \text{ occuring over n years} \tag{2}$$

where $TPVC_{Practice}$ is the total present value of the cost of a practice, $C_{est,Practice}$ is the full establishment cost, and C_{main} involves all annual and/or periodic maintenance costs of the practice applicable for and discounted over n years. The specific variations of this general model for each individual technology are presented in Supplemental material.

To develop a range of costs for each practice, minimum and maximum values for each individual cost category were summed to develop a minimum and maximum TPVC, respectively (Tables 2–7). If only a single value (i.e., mean) was available for a cost, this value was used in both the minimum and maximum TPVC calculation for that practice. As is appropriate for this type of cost comparison assessment (e.g., [38–41]), the minimum and maximum TPVCs for each practice were then used to develop a range of equal annual costs (EACs) for the strategies (Table 9). The EAC approach involves determining the equal annual payment (in present value terms) that would be made at the end of each year to fully cover costs over a planning horizon, and allows for the direct comparison of total present value costs from practices that have different practicable life spans [42]. More pragmatically, the EAC format allows farm-level decision makers to consider environmental best management practice costs essentially on a similar basis that they consider typical farm-level production costs [43].

Following Burdick et al. [44] and Tyndall and Grala [45], conversion to EACs was done using a capital recovery factor (CRF):

$$\mathrm{EAC} = \mathrm{TPVC} \times \mathrm{CRF} \tag{3}$$

where TPVC is the total present value of the cost of the practice and the CRF is calculated using:

$$\mathrm{CRF} = \frac{i(1+i)^n}{(1+i)^n - 1} \tag{4}$$

where i is the annual real discount rate and n is the number of years in the evaluation (i.e., planning horizon). The analysis was carried out using a 4% real discount rate, and the n was set to each practice's individual practicable lifespan (Table 1). A 4% discount rate represented the average real interest rate on Iowa farmland loans during 2008–2010 and was very similar to the 2011 rate for federal water projects (4.125%) [46].

Calculated EACs were combined with published measures of NO_3^- removal efficacy (% load reduction; Table 8) to develop an average efficiency parameter of dollars per kg N removed. This literature review-based approach (as opposed to a more site-specific modeling approach, which was outside the scope of this financial parameterization work) allowed capture of some inherent variability as the literature contains observations across sites and conditions. Dividing the EAC of each strategy by the amount of NO_3^--N removed is a standard way to present total costs per unit e.g., [44,47]. To do so, a Midwestern-representative load of 31.4 kg N ha^{-1} was developed from an average of Jaynes et al. [48] tile and drain N loads and Lawlor et al. [49] drainage N loads at their 168 kg N ha^{-1} application rate. Then, the minimum and maximum EAC for each practice were each applied to that practices' range for N load reduction (mean, median, 25th, and 75th percentiles from Table 8 and Fig. 1

Table 2
Itemized costs and Total Present Value Costs for controlled drainage in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 40 years.

Item	Cost timing (yr)	Minimum cost ($ ha^{-1})	Mean cost ($ ha^{-1})	Maximum cost ($ ha^{-1})	Notes and assumptions	Reference
Structure cost	1	$61.78		$247.11	New drainage system: 1 structure per 8 ha at $500–$2000 per ea.	[15]
	1	$123.55		$494.21	Existing drainage system: 1 structure per 4 ha at $500–$2000 per ea.	
Transport structures	–				Assumed included above	Assumption
Design cost	1		$80.63		For new drainage systems but also included as design cost of existing	[100]
Contractor fees	1	$4.32	$9.47	$15.44	Structure installation: Back hoeing at $35.00 h^{-1}, $76.65 h^{-1}, $125.00 h^{-1} for 8 h to treat 65 ha	[81]
Total cost of establishment		**$146.73** **$208.51**		**$343.18** **$590.29**	**New (TPVC)** **Existing (TPVC)**	
Time to raise/lower	$1-n$	$0.99		$4.94	Four hours × two to four times a year; labor at $8–$20 h^{-1}, 65 ha treatment area	[81]
Stop log/gate replacement	8, 16, 24, …	$17.67		$35.34	Summation of single sum TPV every eight years for 5 gates per structure at original cost of $14.17–$15.32 per ea. for 15 cm structures, 1 structure per 4 (Existing) or 8.1 (New) ha	[101]
Total cost of establishment, maintenance, and replacement		**$183.96**		**$723.44**	**TPVC**	

Table 3
Itemized costs and Total Present Value Costs for a denitrification bioreactor in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 40 years.

Item	Cost timing (yr)	Minimum cost ($ ha^{-1})	Mean cost ($ ha^{-1})	Maximum cost ($ ha^{-1})	Notes	Reference
Both control structures	1	$49.42		$197.68	Two control structures at $500–$2000 ea.; 20.2 ha treatment area	[101]
Structure transport	–				Assumed included above	Assumption
Woodchip cost	1		$116.14		Two semi loads at $975 chips + $200 transport ea.; 20.2 ha treatment area	[102]
Woodchip transport to farm	–				Included above	
Design cost	1	$0.00		$31.63	Assumed: $40 h^{-1} for 2 days of work or NRCS service provider; 20.2 ha treatment	Assumption
Contractor fees	1	$27.68	$60.61	$98.84	Back hoeing at $35.00 h^{-1}, $76.65 h^{-1}, $125.00 h^{-1} for 16 h to treat 20.2 ha	[81]; Assumptions
Seeding bioreactor surface	1	$0.05	$0.11	$0.15	Seeding grass, broadcast with tractor; for 20.2 ha treatment and 0.10 ha bioreactor at $9.88, $22.61, and $29.65 h^{-1}	[81]
Seed cost	1		$1.11		Seed costs from dealer: $222.27 ha^{-1} for CRP Mix (CP23) Diversified mix; bioreactor surface 0.005 of treatment area	[82]
Misc. materials	1		$8.80		6″ tile $890 per 305 m(1000 ft); Assume 61 m needed for control structure connections for 20.20 ha treatment area	[101]
Total cost of establishment		**$203.19**		**$454.35**	**TPVC**	
Time to raise/lower	$1-n$	$1.19		$2.97	Three hours per yr with farm labor wages at $8–$20 h^{-1}, 20.2 ha treatment area	[81]; Assumption
Mowing/maintenance	$1-n$	$0.12		$0.62	Spot mowing bioreactor at $24.71–$123.55 ha^{-1} for 20.2 ha treatment	[83]
Replacement year 20	20	$65.66		$98.18	Single sum TPVC at 20 years: woodchips, contractor, seeding	Assumption
Gate replacement	8, 16, 24,…		$14.14		Summation of single sum TPV every eight years for 5 gates per structure ($14.17–$15.32 per ea. for 15 cm structure) 2 structures per 20.2 ha	[101]
Total cost of establishment, maintenance, and replacement		**$308.91**		**$637.59**	**TPVC**	

Table 4
Itemized costs and Total Present Value Costs for a wetland in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 50 years.

Item	Cost timing (yr)	Minimum cost ($ ha^{-1})	Mean cost ($ ha^{-1})	Maximum cost ($ ha^{-1})	Notes	Reference
Design cost	1		$71.17		Assumed: $40 h^{-1} for 90 days of work (8 h d^{-1}) for 405 ha site	Assumption
Contractor fees	1	$28.17	$34.43	$41.51	Building ponds at 8 h d^{-1} for 15 days with Custom Rate Survey $ h^{-1} for 405 ha wetland , not including seeding time	[81]
Seeding buffer	1	$0.35	$0.79	$1.04	Tractor broadcasting at $9.88, $22.61, or $29.65 ha^{-1} for 14 ha wetland buffer for 405 ha treatment	[81]
Seed cost	1	$7.43		$95.38	Seed costs from dealer: $212.39 ha^{-1} for CRP wetland program mix to $162.09 kg^{-1} for "wetland seed mix" at needed 16.8 kg ha^{-1}	[82,84]
Weir plate	1		$14.83		$30 per sq ft. for 40 ft width × 5 ft sheet pile plate, for 405 ha site	Assumption
Control structure	1	$3.26		$7.25	One large control structure ($1320–$2935 per ea.), for 405 ha site	[101]
Land acquisition	1	$529.08		$679.31	$11,757–$15,095 ha^{-1} for 4 ha wetland plus 14 ha buffer treating 405 ha; 2010 state-wide Iowa average for high and medium grade lands	[85]
Total cost of establishment		**$654.28**		**$910.48**	**TPVC**	
Time to manage	$1-n$	$0.09		$0.43	Spot mowing 10% of buffer area at $24.71–$123.55 ha^{-1}	[83]
Control structure and weir replacement	40	$4.55		$5.75	Single sum TPVC at year 40 includes costs of a new structure and weir and 16 hrs of earth work	Assumption
Total cost of establishment, maintenance, and replacement		**$660.69**		**$925.52**	**TPVC**	

Table 5
Itemized costs and Total Present Value Costs for N management for corn in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 1 year.

Item	Cost timing (yr)	Minimum cost ($ ha^{-1})	Mean cost ($ ha^{-1})	Maximum cost ($ ha^{-1})	Notes	Reference
Fertilizer application	$1-n$	$14.83	$24.09	$42.01	Anhydrous-injecting, w/tool bar	[81]
Diesel for equipment	–				Included above	
Fertilizer cost	$1-n$		$156.40		North Central US mean 2008–2010 anhydrous ammonia price paid: $762.80 metric ton^{-1}; 168 kg N ha^{-1}; AA:82-0-0 (82%)	[56]
Total cost of establishment for baseline application		$171.23		$198.41	Using Fertilizer cost: $156.40 ha^{-1} considering application of 168 kg N ha^{-1} in Fall	[56]
Total cost of establishment at a lower rate (from 168 kg N ha^{-1} to 140 kg N ha^{-1})		**$145.16**		**$172.34**	Using Fertilizer cost: $130.33 ha^{-1} for application of 140 kg N ha^{-1} rather than $156.40 ha^{-1} for 168 kg N ha^{-1}	[56]
Total cost of establishment of Spring application		**$178.42**		**$205.60**	Spring price of $798 metric ton^{-1} at 168 kg N ha^{-1} application rate ($163.59 ha^{-1})	[56,58]
Annual baseline revenue	$1-n$		$1850.12		Iowa mean 2008–2010 yield of 10.84 metric ton ha^{-1} and 2008–2010 mean corn price received of $0.17 kg^{-1}; at 99% yield for 168 kg N ha^{-1}	[55,56]
Annual revenue from changed yields due to N management (Lower rate)	$1-n$		$1831.44		Iowa mean 2008–2010 yield of 10.84 metric ton ha^{-1} and 2008–2010 mean corn price received of $0.17 kg^{-1}; at 98% yield for 140 kg N ha^{-1}	[55,56]
Annual revenue from changed yields due to N management (Spring application)	$1-n$		$1947.30		Iowa mean 2008–2010 yield of 10.84 metric ton ha^{-1} and 2008–2010 mean corn price received of $0.17 kg^{-1}; with 4.2% yield boost for spring application	[56]
Total cost of establishment and revenue impacts for baseline application		−$1614.32		−$1588.19	TPVC (negative represents a revenue)	
Total cost of establishment and revenue impacts at a lower application rate		**−$1621.42**		**−$1595.28**	**TPVC (negative represents a revenue)**	
Total cost of establishment and revenue impacts for Spring application		**−$1700.85**		**−$1674.71**	**TPVC (negative represents a revenue)**	
N Rate Marginal Cost		**−$7.09**		**−$7.09**	**Marginal TPVC**	
Spring N Marginal Cost		**−$86.52**		**−$86.52**	**Marginal TPVC**	

Table 6
Itemized costs and Total Present Value Costs for a cover crop in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 4 years.

Item	Cost timing (yr)	Minimum cost ($ ha^{-1})	Mean cost ($ ha^{-1})	Maximum cost ($ ha^{-1})	Notes	Reference
Seed costs	$1-n$	$14.83		$29.65	Planted at 63 kg ha^{-1}; cereal rye	[32,103]
Planting Drill	$1-n$	$18.53	$32.12	$49.42	Custom cost to have small grains drilled	[81]
Diesel for equipment	–				Included above	
Spraying	$1-n$	$11.12	$15.07	$21.99	Ground, broadcast, tractor	[81]
Herbicide cost	$1-n$		$14.09		Herbicides, Glyphosate, 480 kg m^{-3}, Price paid, US Total, 2010: $6023 m^{-3} ; 0.0023 m^3 ha^{-1}	[32,56]
Total cost of establishment		**$58.56**		**$115.15**	**TPVC**	
Annual baseline revenue (no cover crop)	$1-n$		$1868.81		Iowa mean 2008–2010 yield of 10.84 metric ton ha^{-1} and 2008–2010 mean corn price received of $0.17 kg^{-1}; at 100% yield	[56]
Annual revenue from changed yields due to cover crop	$1-n$		$1752.95		Iowa mean 2008–2010 yield of 10.84 metric ton ha^{-1} and 2008–2010 mean corn price received of $0.17 kg^{-1}; at 6.2% yield reduction for corn following rye	[56]
Difference in annual revenue from baseline			$115.87		Considered a cost of cover crop with corn grown in every other year	
Total cost of establishment and revenue impacts		**$594.98**		**$800.39**	**TPVC**	

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Table 7
Itemized costs and Total Present Value Costs for a diversified crop rotation in the U.S. Midwest at real discount rate of 4 % and analysis horizon of 10 years.

Item	Cost timing (yr)	Minimum cost ($ ha^{-1})	Mean cost ($ ha^{-1})	Maximum cost ($ ha^{-1})	Notes	Reference
Seed costs	Year 3 of every 5	$101.19		$140.48	Legume, alfalfa, public and common seed or proprietary seed, price paid, National, 2010: $273–$379 cwt^{-1}; planted 16.8 kg ha^{-1}	[56]
Planting drill	Year 3 of every 5	$18.53	$32.12	$49.42	Custom cost to have small grains drilled	[81]
Diesel for equipment	–				Included above	Assumption
Soil preparation	Year 3 of every 5		$34.10		Disking, harrow: Default values from ISU Ag Decision Maker	[72] (alfalfa)
Herbicide	Year 3 of every 5		$37.81		Default values from ISU Ag Decision Maker (machinery and chemical)	[72] (alfalfa)
Labor	3–5 of every 5		$81.54		Pre-harvest labor: 7.4 h ha^{-1} at $11.00 h^{-1}	[72] (alfalfa)
Fertilizer	3–5 of every 5	$307.15		$481.36	Default values from ISU Ag Decision Maker for establishment year (min) and production year (max); machinery and chemical	[72] (alfalfa)
Harvesting – mowing	3–5 of every 5	$19.77	$30.64	$37.07	Mowing/conditioning	[81]
Harvesting – baling	3–5 of every 5	$74.13	$123.55	$172.97	Haying baling - small square: $0.30–$0.70 bale^{-1}; 12.4 ton ha^{-1} at 45.4 kg bale^{-1}	[81]; Assumption
Total cost of alfalfa establishment	Year 3 of every 5	**$674.23**		**$860.55**		
Total cost of alfalfa maintenance	Year 4 and 5	$656.81		$772.95	Labor, fertilizer and harvesting costs from above	
Corn in year 1	YEAR 1 of 5		$1183.64		Cost of corn establishment (corn following soybean to be more accurate for years 6, etc.); land rent removed, 10.84 metric ton ha^{-1} yield	[72] (corn following soybean)
Corn in year 2	Year 2 of 5		$1312.13		Cost of corn establishment (corn following corn); land rent removed, 10.84 metric ton ha^{-1}	[72] (corn following corn); [49]
Total costs for five year diversified rotation		**$4214.00**		**$4588.79**	**TPVC: Corn in years 1 and 2 with alfalfa establishment in year 3 and alfalfa maintenance in years 4–5**	
Alfalfa revenue	4–5 of every 5		$1511.46		Alfalfa average yield 12.4 ton ha^{-1} (assuming 3 cuttings); Iowa mean 2008–2010 alfalfa hay price received: $134.85 metric ton^{-1}	[56,72]
Corn revenue			$1868.81			[56]

	1–2 of every 5				Iowa mean 2008–20109 corn yield: 10.84 metric ton ha^{-1} and 2008–2010 mean corn price received of \$0.17 kg^{-1}	
Total revenue for five year diversified rotation			**\$6850.51**		**TPV: Corn revenue in year 1 plus 4.5% yield boost, corn revenue in year 2, alfalfa revenue divided by 3 (only 1 cutting) in alfalfa establishment year, and alfalfa revenue in year 4–5**	[73]
Total costs and revenue for diversified crop rotation for 10 yr horizon		**−\$10,456.91**		**−\$8970.43**	**TPVC (negative represents a revenue)**	
Cost of corn and soybean five year rotation			\$4469.53		TPVC: Five year cost of corn soybean rotation; starting with corn (ISU Decision Maker, corn following soy, yield 10.8 metric ton ha^{-1}); soybean cost: \$637.53 ha^{-1} ISU Ag Decision Maker for herbicide tolerant soybeans following corn, yield 3.33 metric ton^{-1}; land rent removed	[72]
Revenue of corn and soybean five year rotation			\$7564.77		TPV: Five year revenue of corn soybean rotation, starting with corn; corn revenue described above; soybean revenue: Iowa mean 2008–2010 yield of 3.33 metric ton ha^{-1} and mean price \$0.38 kg^{-1} yields \$1281.05 ha^{-1}	[56]
Total costs and revenue for corn and soybean rotation for 10 yr horizon			**−\$12,276.31**		**TPVC (negative represents a revenue)**	
Marginal cost		**\$1819.40**		**\$3305.87**	**Marginal TPVC**	

Table 8
Review of nitrogen load reduction effectiveness for seven drainage water quality practices in the U.S. Midwest.

Practices and references	N load reduction			Notes
	Minimum (%)	**Mean (%)**	**Maximum (%)**	
Controlled drainage				
[86]	30		40	Overview of this N management practice
[15]	15		75	Controlled drainage factsheet
[87]	48	75	100	Load reduction for mean loads from six months of free drainage vs. controlled water tables at 0.25 m and 0.5 m above the drain; Ontario, Canada
[14]		30		Overview of this N management practice
[13]	10		20	An original paper on drainage control
[88]		43		Controlled drainage/sub-irrigation system, Canada
[29]	0		50	N technology comparison
[89]	31	44	51	Simulation of Midwestern region with Root Zone Water Quality Model-Decision Support System for Agrotechnology Transfer (RZWQM –DSSAT)
[90]		26		Mean of DRAINMOD-NII simulated N losses for drain spacing 18 m–36 m for conventional vs. controlled drainage; Waseca, Minnesota
Bioreactor				
[76]	11		13	Bioreactor in Iowa
[76]	47		57	Bioreactor in Iowa
[76]	27		33	Bioreactor in Iowa
[91]	40	55	65	Denitrification trenches surrounding tile drain, Iowa
[92]	23	33	50	Bioreactor in Illinois
[93]		47		Bioreactor in Illinois, slug of NO_3^- injected
[94]	18		47	Bioreactor in Minnesota
[94]	35		36	Bioreactor in Minnesota
Wetland				
[21]	25		78	Review table
[18]	33	40	55	Annual N load reduction for three wetlands, three years of data; Champaign County, Illinois
[95]		33		Wetland in Illinois
[20]	9		15	Mean N load reduction for two years from wetland with area treatment ratio of 1046:1; Iowa
[20]	34		44	Mean N load reduction for two years from wetland with area treatment ratio of 349:1; Iowa
[20]	55		74	Mean N load reduction for two years from wetland with area treatment ratio of 116:1; Iowa
[29]	20		40	N technology comparison
[54]	40		90	Summary of CREP wetlands in Iowa
Spring N application				
[96]	−67	6.4	44	Load difference between fall and spring (corn phase)
[97]	0	27	41	Load difference between fall and spring (corn phase)
[23]	24		30	6-yr period at Waseca, Minnesota

[29]	−10		30	N technology comparison
[59]	14	35	52	Simulation with Environmental Policy Integrated Climate (EPIC) for central Illinois; Fall vs. spring application at five rates ranging from 112 kg N ha^{-1} to 224 kg N ha^{-1} for Drummer soil
[49]	−62	−23	7.4	N load difference between spring and fall applied at 168 and 252 kg N ha^{-1}; Iowa
N rate reduction				
[23]	21		28	6-yr period at Waseca, Minnesota; 134 kg ha^{-1} vs. 202 kg ha^{-1} application
[29]	20		70	N technology comparison
[98]	17		40	Central Iowa; loadings of 48 kg N ha^{-1}, 35 kg N ha^{-1}, and 29 kg N ha^{-1} for high, medium and low N application rates, respectively
Cover crops				
[66]		13		Southwestern Minnesota, three year study
[100]		40		Based on review
[71]	−13.5	−3.3	7.6	Four year loads and mean for corn treatment vs. corn with rye cover; Gilmore City, Iowa
[31]		61		Four year average; Boone County, Iowa
[29]	10		70	N technology comparison
Crop rotation				
[36]	14		77	Review
[99]	11		14	Six year average losses from corn/soybean or soybean/corn vs. rotation with three years alfalfa followed by corn, soybean, oats; Nashua, Iowa
[35]	18	48	80	Conversion from alfalfa pasture; three year study, compared with corn and soybean and continuous corn rotations; Lamberton, Minnesota
[29]	−50		95	N technology comparison

Table 9
Nitrogen load reduction effectiveness and Equal Annual Costs in terms of treatment area or nitrogen removal for seven drainage water quality practices in the U.S. Midwest (without government payments).

	EAC (area-based)		Load reduction from Fig. 2				EAC (N-based)			
	Minimum ($ ha^{-1} yr^{-1})	Maximum ($ ha^{-1} yr^{-1})	25th (%)	75th (%)	Mean (%)	Median (%)	Mean (Standard Deviation, $ kg N removed^{-1} yr^{-1})	Median ($ kg N removed^{-1} yr^{-1})	Minimum [a] ($ kg N removed^{-1} yr^{-1})	Maximum [a] ($ kg N removed^{-1} yr^{-1})
Controlled Drainage	$9.30	$37.00	26.0	50.0	40.5	40.0	$2.00 ($1.40)	$1.70	$0.60	$4.50
Bioreactors	$16.00	$32.00	27.0	47.0	37.5	36.0	$2.10 ($0.90)	$2.00	$1.10	$3.80
Wetland	$31.00	$43.00	30.9	55.0	42.8	40.0	$2.90 ($0.80)	$2.80	$1.80	$4.40
N rate reduction	−$7.40	−$7.40	—	—	14.5	—	−$1.60 ($0.00)	−$1.60	−$1.60	−$1.60
Spring N application [b]	−$90.00	−$90.00	−2.5	31.3	9.3	19.0	−$14.00 ($12.00)	−$12.00	−$31.00	−$0.07
Cover crop	$164.00	$221.00	4.9	45.3	23.1	11.5	$55.00 ($48.00)	$38.00	$12.00	$144.00
Crop rotation	$224.00	$408.00	14.0	77.0	34.1	18.0	$43.00 ($29.00)	$39.00	$9.30	$93.00

[a] Minimum and maximum calculated using the minimum EAC and the 75th percentile load reduction and the maximum EAC and the 25th percentile load reduction, respectively.
[b] Due to confounding effects of negative EAC and negative 25th percentile load reduction (indicating a contribution to the N load), the maximum value for Spring N application was calculated using the marginal increase to the baseline load based on the 25th percentile and the minimum value was calculated from the mean load reduction.

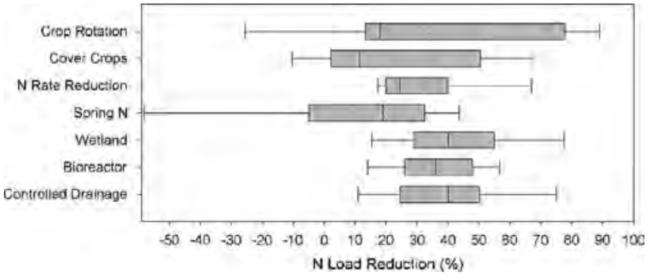

Fig. 1. Comparison of nitrogen load reductions obtained from literature for seven water quality improvement strategies in the U.S. Midwest; the box boundaries represent the 25th and 75th percentiles, the solid line represents the median, the dotted line represents the mean, and the whiskers show the 10th and 90th percentiles.

are shown in Table 9).

$$\frac{\text{EAC \$}}{\text{kg N yr}} = \frac{\text{minimum or maximum EAC \$}}{ha\ yr} \div \left(\frac{\text{31.4 kg N lost baseline}}{\text{ha}} \times \text{Load removal percentage mean, median, 25th, or 75th}\right) \quad (5)$$

In the case of modified N application rate, rather than use load reduction values from literature, a correlation from Lawlor et al. [27] was used (Eq. (1)). For this practice, literature values proved to be too variable as they were not for the specific rates used in this comparison. After drainage NO_3^-–N concentrations were developed via Eq. (1) for the two application rates, a constant drainage volume was assumed to develop a percent N load reduction. While, Eq. (1) was specifically applicable to the database and site from which it was developed (northwestern Iowa), and does not account for other factors that affect N leaching losses (e.g., soil mineralizable N, the time of N application relative to crop N uptake, soil moisture content, weather conditions), it provided a straight forward approach to estimate approximate concentrations based upon N fertilizer application rates.

Finally, because cost-share has been shown to be an important incentive for operators to make environmental mitigation decisions, the impact of existing government cost-share and incentive programming was assessed. In Iowa, USDA environmental quality incentive program (EQIP) payments were available for each of the practices evaluated here except for modification of fertilizer rate [50] (Table 10). EQIP cost rates used were standard rate, not the higher rates available for historically underserved groups. Incentives for controlled drainage, bioreactors, wetlands, and N management were treated as one time, present value payments (year 1), while the others occurred in years $1-n$ with time limits set by EQIP payment schedules. Though EQIP funding is available for wetlands, cost share payments from the Iowa Department of Agriculture and Land Stewardship's Conservation Reserve Enhancement Program (IDALS CREP) are more appropriate because the wetland in this analysis was sized based upon Iowa CREP guidelines. For a CREP 30 year easement agreement, compensation included 15 annual rental payments of 150% the soil rental rate, cost-share for 100% of the wetland installation (90% federal, 10% state), and a one-time incentive payment ($247 ha^{-1}) [19,51]. The soil rental rate was assumed to be the average cash rental rate for 2008–2010 for the state of Iowa ($447 ha^{-1}) [52].

2.1. Controlled drainage

The major cost of controlled drainage is the capital expense of the structures and their installation. Because of this expense, land slope limitations are an important factor as more structures are needed at steeper sites. Another important consideration is the cost difference between implementing controlled drainage in existing vs. newly designed drainage systems [14].

For this evaluation of controlled drainage, the costs to retrofit an existing drainage system and the cost to implement a new drainage system designed for controlled drainage are considered. To reflect

Table 10
Environmental Quality Incentives Program (EQIP) payment schedule rates for Iowa for seven nitrogen reduction practices [50] and calculated total present value ($TPVC_{Govt}$) of this government cost-share for this evaluation.

	EQIP practice name	Practice code	Payment schedule cost	Payment unit	Minimum life (yr)	Year of payment	Payment ($ ha treated $^{-1}$)	$TPVC_{Govt}$ ($ ha^{-1})
Controlled drainage [a]	Drainage water management	554	$364.08	Per number of water control zones	1	1	$44.98	$44.98
Bioreactors [b]	Denitrifying bioreactor	747	$3999.50	Per bioreactor	10	1	$197.66	$197.66
Wetland [c]	Wetland creation	658	$680.00	Per acre	15	1	$16.80	$16.80
N rate reduction [d]	—	—	—	—	—	—	$0.00	$0.00
Spring N application [d, f]	Nutrient management	590	$11.00	Per acre	1	1	$27.18	$27.18
Cover crop [e, f]	Cover crop (and green manure)	340	$53.26	Per acre	1	1–3	$131.61	$379.83
Crop rotation [f]	Conservation crop Rotation	328	$52.00	Per acre	1	1–3	$128.50	$370.85

[a] Used scenario of 65 ha, requiring eight zones.
[b] EQIP specifies treatment of drainage from 12.1 ha which was less than the treatment area assumed here of 20.2 ha; EQIP cost-share was not used in replacement years for bioreactors or controlled drainage.
[c] Based on CREP 30 yr contract incentives rather than EQIP cost share shown here (see Section 2).
[d] Based on a mid-range payment rate requiring only two additional enhancement practices.
[e] Based on “cover crop winter hardy” rate for a winter cover of rye.
[f] EQIP funding for N management, cover crop and crop rotation practices has three year payment time limits; payments for cover crop and crop rotation were assumed to happen in the first three years of the analysis period and because N management had a planning horizon of $n=1$, only 1 year of EQIP was included.

the marginal cost of water quality improvement and not just the cost of new drainage systems, contractor tiling and materials expenses for new systems are not included. Full cost components are described in Table 2. Regarding more long-term costs, the cost of maintenance for this practice includes landowner time to manipulate the control structures; this would vary based on the number of structures, distance between them, and management intensity a landowner chose. The control structure stop logs/gates need to be replaced every eight years. Because the structures themselves would need to be replaced in year 40, this determined the practicable lifespan of this practice ($n=40$).

2.2. Bioreactors

As with controlled drainage, bioreactor establishment costs include design, contractor and structure fees. However, unlike controlled drainage, bioreactor treatment area differs from the surface area of the technology. Here, the $ ha^{-1} values referred to the treatment area not the bioreactor surface footprint. On an itemized basis, a maximum value for engineering fees of $40 h^{-1} for 16 h of work is assumed, though if the bioreactor is designed by a technical service provider, these fees may not apply. Although no land is typically removed from production for bioreactors, seeding the surface is important to prevent erosion of the soil cap. Bioreactors are typically less than 0.5% of the drainage treatment area, so this area ratio is used for the seeding and mowing costs. Bioreactor full cost components are described in Table 3.

Farmer time for adjusting the control structures is minimal compared to the controlled drainage practice due to fewer structures here. In addition to annual maintenance, the bioreactor material is replaced once in year 20 (involving costs associated with new woodchips, seeding and contractor fees) before the structures' lifespan is exhausted in year 40 (bioreactor practicable lifetime, $n=40$). Similar to controlled drainage, the stop logs/gates are replaced every eight years.

2.3. Wetlands

Wetlands are unique in that their capital expense can be very high, but they are capable of treating drainage from far larger areas than the other strategies considered here. Design and construction are important components of wetland establishment but the largest single expense is the land acquisition cost. Longer-term economic considerations sometimes include the opportunity cost of lost crop income (e.g., Prato et al. [53] and Crumpton et al. [22]), as well as maintenance and mowing expense and potential income streams.

For the purposes of this comparison, a 405 ha treatment area is assumed with a wetland occupying 1% of this area (4 ha) consistent with the conservation reserve enhancement program (CREP) guidelines for Iowa which specify a wetland size of 0.5–2% of the treatment area (not including associated wetland buffer) [19,54]. Accordingly, in addition to the wetland basin, a grass buffer is required. The wetland buffer has a 3.5:1 area ratio with the wetland (i.e., 3.5% of the treatment area in buffer, 14 ha) (Iowa Department of Ag. and Land Stewardship, personal communication, 2011). Because land acquisition costs are the largest portion of CREP wetland expense, this is included here; however, land for the other practices (e.g., edge-of-field area for the bioreactor or fields for the in-field practices) is assumed to be owned. Alternatively, forgone annual land rent would be another way to account for land costs. The cost per area for this practice reflected the area treated, not the area of the wetland and associated buffer. Wetland cost components are shown in Table 4.

Structural components include a water control structure and a weir plate, which are used to control wetland flow. The annual maintenance cost involves mowing 10% of the buffer area. Replacement costs of the control structure and sheet pile weir in year 40 are included within the 50 year wetland planning horizon ($n=50$). Also, over the life of a wetland, sediment removal and earthwork maintenance would be required, though those costs are not incorporated in this analysis because their timing would be difficult to estimate and may occur at greater than the 50 year planning horizon.

2.4. N rate reduction (168–140 kg N ha^{-1})

The establishment costs for both N management practices (rate reduction and timing) are similar and include custom rates for application machinery usage and fertilizer costs as described in Table 5. Because an N management practice is an annual occurrence, there are no long-term maintenance costs but, rather, establishment cost and revenue impacts occur every year (practicable lifespan, $n=1$). For these N management strategies, a baseline scenario of fall applied 168 kg N ha^{-1} is developed for comparison. The marginal difference in TPVC between the baseline and the rate/timing alternative is used in the analysis rather than the absolute value of the rate/timing TPVC themselves. Using these marginal costs of the lower rate practice and of the spring timing practice allows evaluation of their cost solely due to water quality improvement.

Financial analysis of lowering the N application rate consists of less fertilizer expense in addition to the cost of potential yield loss depending upon the initial and final application rates [25]. This analysis is complicated by the variability of the impacts of initial and revised fertilizer rates. In practice, challenges to N fertilizer rate reduction include the fact that the optimum rate is indeterminable at application time (though soil testing can help) and is highly variable year to year. Sawyer and Randall [25] provide a detailed explanation of these variable negative and positive returns based on initial and final fertilizer rates.

In analyzing the costs of reduced fertilizer rate here, “establishment” cost consists of less fertilizer purchased (i.e., a cost savings) as well as the effect of potentially reduced yield. The Iowa State University N-Rate Calculator [55] is used to estimate the yield impact from changing the fertilizer rate. Using a three-yr average (2008–2010) anhydrous ammonia price of \$763 metric ton^{-1} [56] and a three-yr average (2008–2010) Iowa corn price of \$0.17 kg^{-1} [56], the calculated percent of maximum yield is 99% at an N application rate of 168 kg N ha^{-1} and is approximately 98% at 140 kg N ha^{-1} (corn following soybean rotation). However, it is worth noting that shifting to this lower rate permanently may not be sustainable over long periods if soil N pools become depleted [57].

2.5. Spring N application

The cost of shifting application from the fall to the spring is affected by differences in both fall/spring fertilizer price and yield. Because current fall vs. spring fertilizer prices are no longer published by USDA, the average historical difference in the fall and spring fertilizer prices, on a percentage basis, is used to calculate the average increase in expense for spring anhydrous application. Between 1960 and 1994, the average prices for September/October were \$184 metric ton^{-1} and for April/May were \$193 metric ton^{-1} [58], thus an increase of 4.6% over the average 2008–2010 anhydrous price of \$763 metric ton^{-1} is used for spring (spring: \$798 metric ton^{-1}).

Multiple authors have reported lower drainage NO_3^- loadings with corresponding higher corn yields for spring vs. fall N applications [23,59,60]. Spring N fertilizer applications may increase yield by 8–14% compared to fall applications [23,60], though this may not always be the case. For example, there was no corn yield difference between fall and spring applications at two different application rates during a study in Iowa [49]. Despite this variability, an overall 4.2% corn yield boost is included for the practice of spring application (site year average from Refs. [49,61–64]).

2.6. Cover crops (cereal rye)

For the purposes of this evaluation, cereal rye (*Secale cereale* L.) is studied as a cover crop because this crop has good potential to improve water quality in cool Midwestern climates [31] and is popular in this region [65]. First year costs of a cover crop (Table 6) (assuming a no-till system in this analysis) include planting as well as herbicide application because cereal rye overwinters [32]. Cover cropping is an annual practice, thus there are no long-term maintenance costs but rather annual establishment costs. A yield reduction for corn following rye is also an important part of the analysis. A 6.2% corn yield reduction is assumed compared to a baseline where no cover crop was used (site year average from Refs. [31,66–71]). This corn revenue reduction is assumed to occur every other year during the planning horizon (i.e., a corn/soybean rotation; cover crop practice period, $t=2$; cover crop planning horizon, $n=4$).

2.7. Crop rotation (multiple years of alfalfa)

The number of possible rotation combinations is quite large, and to simplify this work, a multi-year incorporation of alfalfa (*Medicago sativa* L.) into a corn rotation is investigated. Only one year of alfalfa in a rotation may not be as beneficial as several years considering high seed cost and potential low alfalfa yield in the establishment year [36]. Therefore, this diversified crop rotation consists of two years of corn (years 1–2) followed by three years of alfalfa (years 3–5). The major costs for such a crop rotation are the seed, planting, and harvesting. The cost components of this rotation are shown in Table 7, with the rotation practice period (t) equal to five years and the planning horizon, n, equal to 10 years.

Within the rotation, enterprise budget information published by Iowa State University is used to specifically estimate the costs of corn following soybean (for years 1, 6, etc.; most applicable for corn following alfalfa) and for corn following corn (in years 2, 7, etc.) [72]. Default Iowa State University Ag Decision Maker [72] values were used after removing land rent costs (i.e., assumed land owned) and substitution of average Iowa 2010 corn yield from USDA NASS [56].

A multiple year alfalfa rotation may provide monetary benefit via reduced fertilizer requirements, reduced tillage and other field trips, and revenue from the alfalfa harvest. Here only direct revenue streams are considered with alfalfa revenue in years 3–5 and corn revenue in years 1–2. The establishment year of alfalfa is assumed to only have one cutting rather than the three as in the maintenance years (i.e., establishment years had one third of the yield experienced in maintenance years). Corn following alfalfa may have an increased yield of 19–84% compared to corn after corn according to a review by Olmstead and Brummer [36], but Liebman et al. [73] showed more moderate corn yield increases averaging 4.5% which was used here for the first year of corn.

Additionally, the TPVCs for this crop rotation scenario are compared against TPVCs for traditional corn/soybean rotations. Similarly to the N management practices, this allowed evaluation of the cost of this water quality practice (i.e., marginal cost of the practice). The corn/soybean baseline scenario is evaluated using the same five year framework as the extended rotation with cost values taken from ISU Ag Decision maker for corn following soybeans and herbicide tolerant soybeans following corn with default values except for removal of land rent costs and use of average yields (2008–2010, USDA NASS data) (Table 7) [72].

3. Results and discussion

3.1. Equal annualized costs

The TPVCs from the seven practices ranged from a cost savings of approximately \$90 ha^{-1} for spring applied N fertilizer to a cost of \$3306 ha^{-1} for a diversified crop rotation (Tables 2–7), and the resulting EACs ranged from −\$90 ha^{-1} yr^{-1} (Spring N, representing cost savings) to \$408 ha^{-1} yr^{-1} (crop rotation) (Table 9). The highest EACs were associated with the two in-field vegetated practices, cover crops and crop rotations, and the lowest were associated with the N management strategies. However, the high EACs developed for the cover and diversified cropping practices were associated with large uncertainties (Tables 1 and 9).

With regard to spring N applications, Randall and Sawyer [24] also noted long-term economic gains of \$46–\$126 ha^{-1} yr^{-1} (seven and fifteen year averages). However, a complete shift from fall fertilization could be expensive for individual producers in terms of both additional infrastructure required for spring applications (storage, equipment, labor, handling, application, etc.) and in the potential loss of yield by a delayed planting date [74]. Additionally, when lower N rates are applied, the risk of a yield loss is increased compared to higher application rates if it is a year where corn is more responsive to N inputs (depending upon the soil mineralizable N). In these years, the probability of obtaining a certain yield percentage declines when lower rates are applied; this probabilistic variability was not reflected here. Any such potential increased risk for either of these N management practices is an important factor in terms of producer decision-making.

Along with the relatively high EACs for the rye cover crop (\$164–\$221 ha^{-1} yr^{-1}; Table 9), several comments should be noted. First, costs to kill the cover are contingent upon producer actions. For example, in a no-till system as assumed here, an early burn-down application of herbicide may be done regardless if a cover crop was present; likewise, in a tilled system, a producer may do a second tillage pass in the spring regardless of a cover crop. Second, rye cover crop implementation costs can be \$10–\$15 ha^{-1} lower if a landowner chooses not to use a custom operator [75]. Next, potential negative yield impacts will likely be reduced or minimized through several years of experience with cover crop management. This increased experience also likely means a more effective cover, though returns to farm management can improve under highly skilled managers regardless of the production practice. Finally, some of the N taken up by a cover crop will be returned to future crops. It is difficult to place an economic value on this, but it is worth noting the multiple benefits to the soil provided by cover crops [33]. Because cover crops are typically done for reasons other than drainage water quality improvement, it has been suggested that only a portion of the cost should be attributed to N. However, because this work was solely focused on N reduction cost effectiveness, see Table 1 or Christianson et al. [34] for discussion of the ecosystem services provided by these practices.

The EAC for the diversified rotation was \$224–\$408 ha^{-1} yr^{-1} (Table 9). The values developed here were contrary to values from Olmstead and Brummer [36] who showed a diversified rotation was more profitable than a conventional rotation. One major caveat worth noting is the potential for large scale market effects if this rotation were done by a large numbers of producers in a limited area; if this practice became widespread, the alfalfa price could markedly decline.

The two field-scale constructed practices, controlled drainage and bioreactors, had similar EAC ranges at \$9.30–\$37 ha^{-1} yr^{-1} (spanning both existing and new drainage systems) and \$16–\$32 ha^{-1} yr^{-1}, respectively. For reference of installation costs, bioreactor TPVC estimates (Table 3) were within the range of five bioreactor installations in Iowa (total costs of \$4400–\$11,800 to treat drainage from 12 ha to over 40 ha [76]), and overall TPVCs estimated for constructed wetlands (\$661–\$926 ha^{-1}, Table 4) compared well with cost assessments from IDALS CREP wetlands constructed in Iowa. CREP wetlands average approximately \$880 ha^{-1} including land acquisition (\$513 ha^{-1}), establishment and maintenance costs (\$297 ha^{-1}), and engineering costs (\$69 ha^{-1}). As of 2011, 72 wetlands had been installed under the CREP wetland program in Iowa with an average treatment area of 505 ha (Iowa Department of Ag. and Land Stewardship, personal communication, 2011).

3.2. Comparative average cost effectiveness of nitrogen mitigation

In addition to variation between practices in TPVCs and EACs, the practices also varied widely in terms of N removal effectiveness (Fig. 1). For example, modification of fertilizer timing had comparatively low N removal, ranging notably into the potential for negative water quality impacts, while the constructed practices tended to have relatively better water quality performance. Bioreactors had the smallest range of N load reduction between the 25th and 75th percentiles with mean and median values above 35% load reduction. The other two constructed practices, controlled drainage and wetlands, had similarly high load reduction potential (means and medians $\geq 40\%$). Note, because the 25th percentile for spring N application was a negative value (-2.5%), indicating a contribution to the N load, the resulting marginal increase to the baseline load was used to calculate the \$ kg N^{-1} yr^{-1} for this value.

When these N removal performances were combined with the cost data, spring N application timing was the most cost effective option for removing N from drainage (mean \$14 kg N^{-1} yr^{-1} cost savings or revenue) and cover crop the least (mean \$55 kg N^{-1} yr^{-1}) (Table 9, Fig. 2). Both N management practices yielded negative average cost efficiencies indicating a savings or increased profitability. However, it's important to note nutrient management practices alone may not be sufficient to meet all N water quality goals in the Midwestern Region. In addition to the highest mean values, the cover crop and the diversified rotation had the largest standard deviations (pre-government payment), which highlighted the variability of these two in-field vegetative practices both in terms of costs and N removal potential. The more constructed practices of controlled drainage, bioreactors and wetlands had fairly comparable average cost efficiencies with mean values between \$2 and \$3 kg N^{-1} yr^{-1} (Table 9, Fig. 2).

To put these average cost efficiencies in context of other reported values is difficult in light of the variable methodology and limited transparency of other assessments. Nevertheless several practices were in the range of literature, while others were distinctly different. For example, the cost efficiency of controlled drainage in this analysis was \$2.00 ± \$1.40 kg N^{-1} yr^{-1}, which was similar to reports which are often in the range of \$2–\$4 kg N^{-1} [77,78]. Moreover, the average cost efficiency of wetlands is often reported at approximately \$3–\$4 kg N^{-1} [51,54,77,79]; the value reported in our study was \$2.90 ± \$0.80 kg N^{-1} yr^{-1}. Only one report was available for bioreactors; in a multi-year cost analysis of a theoretical denitrification system, Schipper et al. [80] calculated costs of \$2.39–\$15.17 kg N^{-1}. This range was higher than what was estimated for a bioreactor in our study (\$2.10 ± \$0.90 kg N^{-1} yr^{-1}). Finally, cover crops have been reported to be less expensive per kg N removed than calculated in this analysis (mean \$55 ± \$48 kg N^{-1} yr^{-1}). Values from cover crop literature have ranged from \$1.26 kg N^{-1} to \$11.06 kg N^{-1} [32,75,77], though these previous reports may not have included corn yield impacts.

Inclusion of EQIP or CREP payments generally increased the average cost effectiveness of the practices from a farmer's perspective (Table 9 vs. Table 11) with the largest percentage change

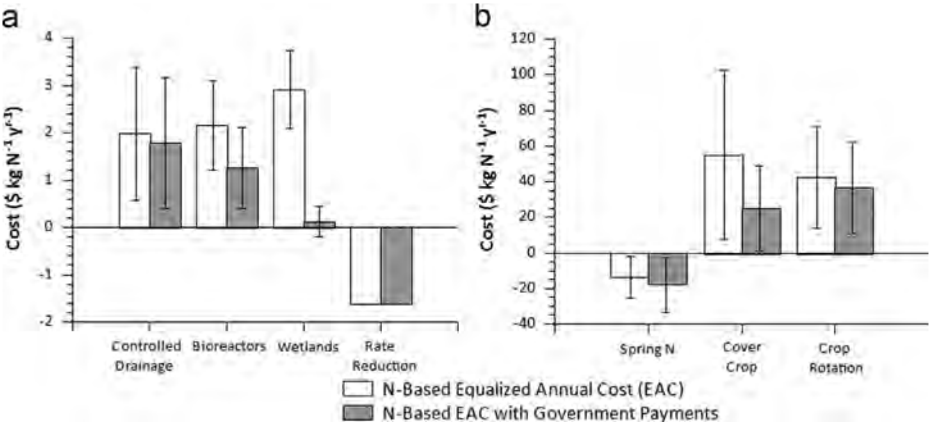

Fig. 2. Equal Annual Costs (\$ kg N^{-1} yr^{-1}) on a nitrogen removal basis for seven agricultural practices in the U.S. Midwest with and without government payments at real discount rate of 4% and analysis horizons of practicable lifespans by practice; note y-axis scales differ for figure parts (a) and (b).

Table 11
Nitrogen removal-based Equal Annual Costs for seven drainage water quality practices in the U.S. Midwest including government payments and additional revenue at real discount rate of 4 % and analysis horizons of practicable lifespan by practice.

	Equal annual costs	
	Mean (standard deviation, \$ kg N^{-1} yr^{-1})	Median (\$ kg N^{-1} yr^{-1})
Controlled drainage	\$1.80 (\$1.40)	\$1.50
Bioreactors	\$1.30 (\$0.86)	\$1.10
Wetland	\$0.12 (\$0.32)	\$0.09
N rate reduction	−\$1.60 (\$0.00)	−\$1.60
Spring N application	−\$18.00 (\$16.00)	−\$16.00
Cover crop	\$25.00 (\$24.00)	\$16.00
Crop rotation	\$36.00 (\$26.00)	\$33.00

occurring for the wetland practice. Without government payments, the practices in order of average cost effectiveness were (based on mean value): Spring N application, N application rate reduction, controlled drainage, bioreactors, wetlands, crop rotation and cover crops. When government payments were included, wetlands and bioreactors became the third and fourth most cost effective practices, respectively, and diversified crop rotations became the least cost effective (from the farmer's perspective) (Fig. 2).

4. Conclusions

Each drainage N reduction strategy provides landowners an additional distinct option for drainage water quality improvement and different strategies or combinations of such will be applicable in different locations. In this work, the N management practices were the most cost effective as both lowering the application rate (from 168 to 140 kg N ha^{-1}) and moving applications to spring resulted in negative costs. Of course, the scenarios here were limited in scope, and there is a wide range of N management and application possibilities that could yield different results. Importantly, a complete ban of fall fertilization could have large-scale economic effects, which were not investigated in this farm-level analysis. The least cost effective practices were the in-field vegetative practices of cover crop and crop rotation though these average cost efficiencies had wide standard deviations. Moreover, benefits like soil productivity, erosion protection, and management or reduction of multiple contaminants were not quantified. The three constructed practices were comparable in terms of pre-cost share \$ kg N^{-1} yr^{-1} although wetlands were very cost effective when CREP incentives were included. A final important note is that while this study focused on water quality NO_3^- mitigation, several of these practices provide significant additional ecosystem services not quantified here.

In an applied sense, these average cost efficencies need to be considered in context of the multiple agricultural and environmental objectives that will differ for each farm and for each farmer. Though the N management practices had the most attractive cost efficiencies, sole focus on N management either on farm or in policies will likely be insufficient to meet water quality goals in entirety. And while improved N management may be “low hanging fruit” for farmers aiming to improve water quality, there are important large scale impacts (e.g., infrastructure requirements for a complete fall fertilizer ban) that were not investgated in this farm level study. At the other end of the cost efficiency spectrum, the in-field vegetative practices were the least attractive in this analysis. However, with this work defined narrowly by reduction of N in drainage water, several potential additional agronomic and environmental benefits of these practices were excluded. Reduction of erosion and improved soil qualities potentially provided by these practices may be important considerations for farm decision makers. These strategies should certainly not be overlooked as Dinnes [29] reported that diversifying cropping systems in Iowa has the most potential to reduce NO_3^- loadings compared with any other best management practice.

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Appendix A. Supplementary information

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.wre.2013.09.001.

References

[1] D.L. Dinnes, D.L. Karlen, D.B. Jaynes, T.C. Kaspar, J.L. Hatfield, T.S. Colvin, C.A. Cambardella, Nitrogen management strategies to reduce nitrate leaching in tile-drained midwestern soils, Agronomy Journal 94 (2002) 153–171.
[2] United States Geological Survey, Nitrogen in the Mississippi Basin—Estimating Sources and Predicting Flux to the Gulf of Mexico. ⟨http://ks.water.usgs.gov/pubs/fact-sheets/fs.135-00.pdf⟩, 2000 (accessed 08.08.12).
[3] J.A. Delgado, R.F. Follett, Advances in nitrogen management for water quality, Journal of Soil and Water Conservation 66 (2011) 25A–26A.
[4] D.W. Lemke, D.P. McKenna, Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop, ASABE, St. Joseph, MI vii–ix.
[5] L. Christianson, J. Tyndall, Seeking a dialog: a targeted technology for sustainable agricultural systems in the US Corn Belt, Sustainability: Science Practice and Policy 7 (2011) 70–77.
[6] R.D. Battel, D.E. Krueger, Barriers to change: farmers' willingness to adopt sustainable manure management practices, Journal of Extension 43 (2005). ⟨http://www.joe.org/joe/2005august/a7.shtml⟩.
[7] J.M Gillespie, S.A. Kim, K. Paudel, Why don't producers adopt best management practices? An analysis of the beef cattle industry, Agricultural Economics 36 (2007) 89–102.
[8] R.L. McCown, New thinking about farmer decision makers, in: J.L. Hatfield (Ed.), The Farmer's Decision, Soil and Water Conservation Society, Ankeny, Iowa 2005, pp. 11–44.
[9] L.S. Prokopy, K. Floress, D. Klotthor-Weinkauf, A. Baumgart-Getz, Determinants of agricultural best management practice adoption: evidence from the literature, Journal of Soil and Water Conservation 63 (2008) 300–311.
[10] A.M. Lemke, T.T. Lindenbaum, W.L. Perry, M.E. Herbert, T.H. Tear, J.R. Herkert, Effects of outreach on the awareness and adoption of conservation practices by farmers in two agricultural watersheds of the Mackinaw River, Illinois, Journal of Soil and Water Conservation 65 (2010) 304–315.
[11] USDA ARMS, Agricultural Resource Management Survey USDA. ⟨http://www.ers.usda.gov/data-products/arms-farm-financial-and-crop-production-practices/tailored-reports.aspx#.Ua3Y9EC1GCk⟩, 2012 (accessed 05.06.13).
[12] G.W. Randall, M.J. Goss, Nitrate losses to surface water through subsurface, tile drainage, in: R.F. Follett, J.L. Hatfield (Eds.), Nitrogen in the Environment: Sources, Problems, and Management, Elsevier Science, 2001. (Chapter 5).
[13] J.W. Gilliam, R.W. Skaggs, S.B. Weed, Drainage control to diminish nitrate loss from agricultural fields, Journal of Environmental Quality 8 (1979) 137–142.
[14] R.A. Cooke, G.R. Sands, L.C. Brown, Drainage Water Management: A Practice for Reducing Nitrate Loads from Subsurface Drainage Systems. http://water.epa.gov/type/watersheds/named/msbasin/upload/2006_8_24_msbasin_symposia_ia_session2.pdf, (accessed 08.08.12).
[15] J. Frankenberger, E. Kladivko, G. Sands, D. Jaynes, N. Fausey, M. Helmers, R. Cooke, J. Strock, K. Nelson, L. Brown, Drainage Water Management for the Midwest: Questions and Answers About Drainage Water Management for the Midwest, Purdue Agriculture. ⟨http://www.extension.purdue.edu/extmedia/wq/wq-44.pdf⟩, 2006 (accessed 08.08.12).
[16] R.A. Cooke, A.M. Doheny, M.C. Hirschi, Bio-reactors for Edge of Field Treatment of Tile Outflow, Paper Number 012018, in: Proceedings of the 2001 ASAE Annual International Meeting, ASABE, St. Joseph, MI, 2001.
[17] L. Christianson, A. Bhandari, M. Helmers, Emerging technology: denitrification bioreactors for nitrate reduction in agricultural waters, Journal of Soil and Water Conservation 64 (2009) 139A–141A.
[18] D.A. Kovacic, M.B. David, L.E. Gentry, K.M. Starks, R.A. Cooke, Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage, Journal of Environmental Quality 29 (2000) 1262–1274.
[19] Iowa Department of Agriculture & Land Stewardship, ⟨http://www.agriculture.state.ia.us/waterResources/pdf/LandownerGuide.pdf⟩, (accessed 08.08.12).
[20] J.L. Baker, W.G. Crumpton, Use of Constructed/Reconstructed Wetlands to Reduce Nitrate–Nitrogen Transported with Subsurface Drainage, in: Proceedings of the 1st Agricultural Drainage Field Day, Lamberton, MN, 2002. ⟨http://swroc.cfans.umn.edu/prod/groups/cfans/@pub/@cfans/@swroc/documents/content/cfans_content_290197.pdf⟩, (accessed 08.08.12).
[21] W.G. Crumpton, G.A. Stenback, B.A. Miller, M.J. Helmers, Potential Benefits of Wetland Filters for Tile Drainage Systems: Impact on Nitrate Loads to Mississippi River Subbasins, Final Project Report to U.S. (Proj. No. IOW06682), Department of Agriculture, 2006 ⟨http://www.fsa.usda.gov/Internet/FSA_File/fsa_final_report_crumpton_rhd.pdf⟩ (accessed 08.08.12).
[22] W.G. Crumpton, D.A. Kovacic, D.L. Hey, J.A. Kostel, Potential of Restored and Constructed Wetlands to Reduce Nutrient Export from Agricultural Watersheds in the Corn Belt, in: Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop, ASABE, St. Joseph, MI29–42 (Chapter 3).
[23] G.W. Randall, D.J. Mulla, Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices, Journal of Environmental Quality 30 (2001) 337–344.
[24] G.W. Randall, J.E. Sawyer, Nitrogen Application Timing, Forms, and Additives, in: Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop, ASABE, St. Joseph, MI73–85 (Chapter 6).
[25] J.E. Sawyer, G.W. Randall, Nitrogen Rates, in: Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop, ASABE, St. Joseph, MI59–72 (Chapter 5).
[26] M.J. Helmers, J.L. Baker, Strategies for Nitrate Reduction: The Cedar River case study, in: Proceedings of 22st Annual Integrated Crop Management Conference, Iowa State University, Ames, IA, 2010.
[27] P.A. Lawlor, M.J. Helmers, J.L. Baker, S.W. Melvin, D.W. Lemke, Nitrogen application rate effect on nitrate-nitrogen concentration and loss in subsurface drainage for a corn-soybean rotation, Transactions of the ASABE 51 (2008) 83–94.
[28] K.G. Cassman, A. Dobermann, D.T. Walters, Agroecosystems, nitrogen-use efficiency, and nitrogen management, Ambio 31 (2002) 132–140.
[29] D.L. Dinnes, Assessments of Practices to Reduce Nitrogen and Phosphorus Nonpoint Source Pollution of Iowa's Surface Waters, Report for the Iowa, Department of Natural Resources in cooperation with the USDA-ARS National Soil Tilth Laboratory, 2004 ⟨http://www.iowadnr.gov/portals/idnr/uploads/water/nutrients/files/nps_assessments.pdf⟩ (accessed 08.08.12).
[30] United States Department of Agriculture Economic Research Service, Agricultural Resources and Environmental Indicators Publication, 4.5 Nutrient Management, 1997. ⟨http://www.ers.usda.gov/publications/arei/ah712/AH7124-5.PDF⟩, (accessed 08.08.12).

[31] T.C. Kaspar, D.B. Jaynes, T.B. Parkin, T.B. Moorman, Rye cover crop and gamagrass strip effects on NO_3 concentration and load in tile drainage, Journal of Environmental Quality 36 (2007) 1503–1511.
[32] T.C. Kaspar, E.J. Kladivko, J.W. Singer, S. Morse, D. Mutch, Potential and Limitations of Cover Crops, Living Mulches, and Perennials to Reduce Nutrient Losses to Water Sources from Agricultural Fields, in: Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop, ASABE, St. Joseph, MI129–147 (Chapter 10).
[33] T.C. Kaspar, J.W. Singer, The use of cover crops to manage soil, in: J.L. Hatfield, T.J. Sauer (Eds.), Soil Management: Building a Stable Base for Agriculture, American Society of Agronomy and Soil Science Society of America, Madison, WI 2011, pp. 321–337.
[34] L. Christianson, T. Knoot, D. Larsen, J. Tyndall, M. Helmers, Adoption potential of nitrate mitigation practices: an ecosystem services approach, International Journal of Agricultural Sustainability (2013)http://dx.doi.org/ 10.1080/14735903.2013.835604. in press.
[35] D.R. Huggins, G.W. Randall, M.P. Russelle, Subsurface drain losses of water and nitrate following conversion of perennials to row crops, Agronomy Journal 93 (2001) 477–486.
[36] J. Olmstead, E.C. Brummer, Benefits and barriers to perennial forage crops in Iowa corn and soybean rotations, Renewable Agriculture andFood Systems 23 (2008) 97–107.
[37] V. Afari-Sefa, E.K. Yiridoe, R. Gordon, D. Hebb, Decision considerations and cost analysis of Beneficial Management Practice implementation in Thomas Brook Watershed, Nova Scotia, Journal of International Farm Management 4 (2008) 1–32.
[38] N.S. Rao, Z.M. Easton, D.R. Lee, T.S. Steenhuis, Economic analysis of best management practices to reduce watershed phosphorus losses, Journal of Environmental Quality 41 (2012) 855–864.
[39] Y. Yuan, S.M. Dabney, R.L. Bingner, Cost effectiveness of agricultural BMPs for sediment reduction in the Mississippi Delta, Journal of Soil and Water Conservation 57 (2002) 259–267.
[40] X. Zhou, M.J. Helmers, M. Al-Kaisi, H.M. Hanna, Cost-effectiveness and cost-benefit analysis of conservation management practices for sediment reduction in an Iowa agricultural watershed, Journal of Soil and Water Conservation 64 (2009) 314–323.
[41] J.R. Williams, P.M. Clark, P.G. Balch, Streambank stabilization: An economic analysis from the landowner's perspective, Journal of Soil and Water Conservation 59 (2004) 252–259.
[42] J. Canada, W. Sullivan, D. Kulonda, J. White, Capital Investment Analysis for Engineering and Management, 3rd ed., Prentice Hall, New Jersey624.
[43] R.D. Kay, W.M. Edwards, P.A. Duffy, Investment Analysis, in: Farm Management, McGraw-Hill, New York, NY 308–329 (Chapter 17).
[44] C.R. Burdick, D.R. Refling, H.D. Stensel, Advanced biological treatment to achieve nutrient removal, Journal of the Water Pollution Control Fed 54 (1982) 1078–1086.
[45] J.C. Tyndall, R.C. Grala, Financial feasibility of using shelterbelts for swine odor mitigation, Agroforestry Systems 76 (2009) 237–250.
[46] United States Department of Agriculture Natural Resources Conservation Service, Rate for federal water projects. 〈http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/econ/references/?&cid=nrcs143_009685〉, 2012 (accessed 08.08.12).
[47] R.H. Van Note, P.V. Hebert, R.M. Patel, C. Chupek, L. Feldman, A Guide to the Selection of Cost-effective Wastewater Treatment System. 〈http://nepis.epa.gov〉, 1975 (accessed 08.08.12).
[48] D.B. Jaynes, J.L. Hatfield, D.W. Meek, Water quality in Walnut Creek Watershed: herbicides and nitrate in surface waters, Journal of Environmental Quality 28 (1999) 45–59.
[49] P.A. Lawlor, M.J. Helmers, J.L. Baker, S.W. Melvin, D.W. Lemke, Comparison of liquid swine manure and ammonia nitrogen application timing on subsurface drainage water quality in Iowa, Transactions of the ASABE 54 (2011) 973–981.
[50] United States Department of Agriculture Natural Resources Conservation Service (Iowa), Iowa Environmental Quality Incentives Program (EQIP) list of eligible practices and payment schedule FY2011. 〈http://www.ia.nrcs.usda.gov/programs/eqip/FY2011%20Iowa%20EQIP%20Practice%20Descriptions%20and%20Payment%20Rates.pdf〉, 2010 (accessed 08.08.12).
[51] S. Hyberg, Economics of CREP/CRP Treatment Wetlands for the Tile Drained Cropland in the Corn Belt. 〈http://www.fsa.usda.gov/Internet/FSA_File/hyberg_iowa_wetlands.pdf〉, 2007 (accessed 08.08.12).
[52] Iowa State University Extension, Cash rental rates for Iowa 2012 Survey File C2-10. http://www.extension.iastate.edu/agdm/wholefarm/pdf/c2-10.pdf, 2012 (accessed 08.08.12).
[53] T. Prato, Y. Wang, T. Haithcoat, C. Barnett, C. Fulcher, Converting hydric cropland to wetland in Missouri: a geoeconomic analysis, Journal of Soil and Water Conservation 50 (1995) 101–106.
[54] R. Iovanna, S. Hyberg, W. Crumpton, Treatment wetlands: cost-effective practice for intercepting nitrate before it reaches and adversely impacts surface waters, Journal of Soil and Water Conservation 63 (2008) 14A–15A.
[55] J. Sawyer, E. Nafziger, G. Randall, L. Bundy, G. Rehm, B. Joern, 2006. Concepts and Rationale for Regional Nitrogen Rate Guidelines for Corn (PM 2015), Iowa State University Extension.
[56] United States Department of Agriculture National Agricultural Statistics Service, Quick Stats 2.0. 〈http://quickstats.nass.usda.gov/〉, 2011 (accessed 08.08.12).
[57] D.B. Jaynes, D. Karlen, Sustaining Soil Resources While Managing Nutrients, in: Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop, ASABE, St. Joseph, MI149–158 (Chapter 11).
[58] United States Department of Agriculture Economic Research Service, Data Sets: Fertilizer use and Price. 〈http://www.ers.usda.gov/Data/FertilizerUse/〉, 2012 (accessed 08.08.12).
[59] R.M. Rejesus, R.H. Hornbaker, Economic and environmental evaluation of alternative pollution-reducing nitrogen management practices in central Illinois, Agriculture, Ecosystems and Environment 75 (1999) 41–53.
[60] G. Randall, 2008. Managing Nitrogen for Optimum Profit and Minimum Environmental Loss, in: Proceedings of Annual Integrated Crop Management Conference, Iowa State University, Ames, IA.
[61] G.W. Randall, J.A. Vetsch, J.R. Huffman, Corn production on a subsurface-drained mollisol as affected by time of nitrogen application and Nitrapyrin, Agronomy Journal 95 (2003) 1213–1219.
[62] J.A. Vetsch, G.W. Randall, Corn production as affected by nitrogen application timing and tillage, Agronomy Journal 96 (2004) 502–509.

[63] M.W. Clover, Impact of nitrogen management on corn grain yield and nitrogen loss on a tile drained field (M.S. thesis), University of Illinois at Urbana-Champaign, Urbana-Champaign, IL, 2005.
[64] G.W. Randall, J.A. Vetsch, Corn production on a subsurface-drained mollisol as affected by fall versus spring application of nitrogen and Nitrapyrin, Agronomy Journal 97 (2005) 472–478.
[65] J.W. Singer, Corn Belt assessment of cover crop management and preferences, Agronomy Journal 100 (2008) 1670–1672.
[66] J.S. Strock, P.M. Porter, M.P. Russelle, Cover cropping to reduce nitrate loss through subsurface drainage in the Northern U. S. Corn Belt, Journal of Environmental Quality 33 (2004) 1010–1016.
[67] J. Sawyer, J. Pantoja, D. Barker, Nitrogen fertilization of corn grown with a cover crop, Iowa State University Research Farm Report, , 2009 ⟨http://www.agronext.iastate.edu/soilfertility/info/2009CoverCrop-NFertilization.pdf⟩ (accessed 08.08.12).
[68] C. Pederson, R. Kanwar, M. Helmers, A. Mallarino, Impact of liquid swine manure application and cover crops on ground water quality, Iowa State University Northeast Research and Demonstration Farm Annual Report (RFR-A10111), 2010 ⟨http://www.ag.iastate.edu/farms/10reports/Northeast/ImpactLiquidSwine.pdf⟩ (accessed 08.08.12).
[69] J. Sawyer, J. Pantoja, D. Barker, Nitrogen fertilization of corn grown with a cover crop, Iowa State University Research Farm Report (RFR-A1064), 2010 ⟨http://www.ag.iastate.edu/farms/10reports/Northeast/ImpactLiquidSwine.pdf⟩ (accessed 08.08.12).
[70] Practical Farmers of Iowa, Cover Crop Effect on Cash Crop Yield: Year 2. ⟨http://www.practicalfarmers.org/assets/files/field_crops/cropping-systems/Cover_Crop_Effect_on_Yield.pdf⟩, 2011 (accessed 08.08.12).
[71] Z. Qi, M.J. Helmers, R.D. Christianson, C.H. Pederson, Nitrate-Nitrogen losses through subsurface drainage under various agricultural land covers, Journal of Environmental Quality 40 (2011) 1578–1585.
[72] Iowa State University Extension, Estimated costs of crop production in Iowa—2011 (including corn following corn, corn following soybeans, herbicide tolerant soybeans following corn, and alfalfa or alfalfa-grass hay). http://www.extension.iastate.edu/agdm/crops/html/a1-20.html, 2011 (accessed October 2011).
[73] M. Liebman, L.R. Gibson, D.N. Sundberg, A.H. Heggenstaller, P.R. Westerman, C.A. Chase, R.G. Hartzler, F.D. Menalled, A. S. Davis, P.M. Dixon, Agronomic and economic performance characteristics of conventional and low-external-input cropping systems in the central Corn Belt, Agronomy Journal 100 (2008) 600–610.
[74] D. Otto, Economic impacts of fall commercial nutrient regulation, Iowa State University Department of Economics. ⟨https://www.econ.iastate.edu/sites/default/files/publications/papers/p11215-2008-03-01.pdf⟩, 2008 (accessed 08.08.12).
[75] A. Saleh, E. Osei, D.B. Jaynes, B. Du, J.G. Arnold, Economic and environmental impacts of LSNT and cover crops for nitrate-nitrogen reduction in Walnut Creek watershed, Iowa, using FEM and enhanced SWAT models, Transactions of the ASABE 50 (2007) 1251–1259.
[76] L. Christianson, Design and performance of denitrification bioreactors for agricultural drainage (Ph.D. dissertation), Iowa State University, Ames, IA, 2011.
[77] J. Baker, 2009. The UMRSHNC Workshop: the Basis for the Cedar River Watershed Case Study, in: Proceedings of the Science to Solutions: Reducing Nutrient Export to the Gulf of Mexico, a Workshop for Managers, Policy Makers, and Scientists, Soil and Water Conservation Society, Ankeny, IA.
[78] D.B. Jaynes, K.R. Thorp, D.E. James, 2010. Potential Water Quality Impact of Drainage Water Management in the Midwest USA, Paper Number IDS-CSBE100084, in: Proceedings of the 9th International Drainage Symposium of the ASABE, ASABE, St. Joseph, MI.
[79] M.O. Ribaudo, R. Heimlich, R. Claassen, M. Peters, Least-cost management of nonpoint source pollution: source reduction versus interception strategies for controlling nitrogen loss in the Mississippi Basin, Ecological Economics 37 (2001) 183–197.
[80] L.A. Schipper, W.D. Robertson, A.J. Gold, D.B. Jaynes, S.C. Cameron, Denitrifying bioreactors—an approach for reducing nitrate loads to receiving waters, Ecological Engineering 36 (2010) 1532–1543.
[81] Iowa State University Extension, 2010 Iowa Farm Custom Rate Survey Ag Decision Maker. ⟨http://www.extension.iastate.edu/publications/fm1698.pdf⟩, 2010 (accessed October 2011).
[82] Prairie Land Management, CRP Mix—Seed Mix Pricing. http://www.habitatnow.com/store/shop/shop.php?pn_selected_category=37, 2005 (accessed 08.08.12).
[83] Iowa State University Extension, Natural Resources Custom Rate Survey Ag Decision Maker. http://www.extension.iastate.edu/agdm/crops/pdf/a3-11.pdf, 2009 (accessed 08.08.12).
[84] Ernst Conservation Seed, Seed mixes. ⟨http://www.ernstseed.com/seed-mixes/⟩, 2011 (accessed October 2011).
[85] Iowa State University Extension, 2010 Farmland value survey. ⟨http://www.extension.iastate.edu/agdm/wholefarm/html/c2-70.html⟩, 2011 (accessed October 2011).
[86] R.A. Cooke, G.R. Sands, L.C. Brown, Drainage water management: A practice for reducing nitrate loads from subsurface drainage systems, in: Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop, ASABE, St. Joseph, MI 19–28 (Chapter 2).
[87] V. Lalonde, C.A. Madramootoo, L. Trenholm, R.S. Broughton, Effects of controlled drainage on nitrate concentrations in subsurface drain discharge, Ag, Water Management 29 (1996) 187–199.
[88] C.F. Drury, C.S. Tan, J.D. Gaynor, T.O. Oloya, T.W. Welacky, Influence of controlled drainage-subirrigation on surface and tile drainage nitrate loss, Journal of Environmental Quality 25 (1996) 317–324.
[89] K.R. Thorp, D.B. Jaynes, R.W. Malone, Simulating the long-term performance of drainage water management across the Midwestern United States, Transactions of the ASABE 51 (2008) 961–976.
[90] W. Luo, G.R. Sands, M. Youssef, J.S. Strock, I. Song, D. Canelon, Modeling the impact of alternative drainage practices in the northern Corn-belt with DRAINMOD-NII, Ag, Water Management 97 (2010) 389–398.
[91] D.B. Jaynes, T.C. Kaspar, T.B. Moorman, T.B. Parkin, In situ bioreactors and deep drain-pipe installation to reduce nitrate losses in artificially drained fields, Journal of Environmental Quality 37 (2008) 429–436.
[92] K.P. Woli, M.B. David, R.A. Cooke, G.F. McIsaac, C.A. Mitchell, Nitrogen balance in and export from agricultural fields associated with controlled drainage systems and denitrifying bioreactors, Ecological Engineering 36 (2010) 1558–1566.
[93] J.A. Chun, R.A. Cooke, J.W. Eheart, J. Cho, Estimation of flow and transport parameters for woodchip-based bioreactors: II. field-scale bioreactor, Biosystems Engineering 105 (2010) 95–102.

[94] A. Ranaivoson, J. Moncrief, R. Venterea, M. Dittrich, Y. Chander, P. Rice, Bioreactor Performance In Minnesota. ⟨http://www.extension.umn.edu/AgDrainage/components/Ranaivoson.pdf⟩, 2010 (accessed 08.08.12).
[95] P.S. Miller, J.K. Mitchell, R.A. Cooke, B.A. Engel, A wetland to improve agricultural subsurface drainage water quality, Transactions of the ASAE 45 (2002) 1305–1317.
[96] G.W. Randall, J.A. Vetsch, J.R. Huffman, Nitrate losses in subsurface drainage from a corn–soybean rotation as affected by time of nitrogen application and use of Nitrapyrin, Journal of Environmental Quality 32 (2003) 1764–1772.
[97] G.W. Randall, J.A. Vetsch, Nitrate losses in subsurface drainage from a corn–soybean rotation as affected by fall and spring application of nitrogen and Nitrapyrin,, Journal of Environmental Quality 34 (2005) 590–597.
[98] D.B. Jaynes, T.S. Colvin, D.L. Karlen, C.A. Cambardella, D.W. Meek, Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate, Journal of Environmental Quality 30 (2001) 1305–1314.
[99] R.S. Kanwar, R.M. Cruse, M. Ghaffarzadeh, A. Bakhsh, D.L. Karlen, T.B. Bailey, Corn-soybean and alternative cropping systems effects on NO_3–N leaching losses in subsurface drainage water, Applied Engineering in Agriculture 21 (2005) 181–188.
[100] D.B. Jaynes, Personal communication, USDA ARS National Laboratory for Agriculture and the Environment, Ames, IA, USA (2011).
[101] Agri Drain Corp., Personal communication, Adair, IA, USA (2011).
[102] Iowa Soybean Association, Personal communication, Ankeny, IA, USA (2011).
[103] T. Kaspar, Personal communication, USDA ARS National Laboratory for Agriculture and the Environment, Ames, IA, USA (2011).

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Title: Subsurface drain losses of water and nitrate following conversion of alfalfa and conservation reserve land to row crops

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Interpretive Summary: The conversion of annual row crops to alfalfa (ALF) and perennial grasses achieved with Conservation Reserve Program (CRP) plantings has reduced losses of nitrate nitrogen through subsurface tile drains in the Upper Midwest. Conversion of alfalfa or CRP back to row crops could have rapid, adverse affects on water quality of tile drainage. Our objectives were to evaluate how prior perennial crops affect water and N use efficiency of annual row crops, and losses of water and nitrate to subsurface tile drains. Tile flow volumes increased to levels similar to row-crops during the first season following conversion of ALF and CRP to corn. Residual soil nitrate (RSN) in the root zone increased by 125% in first year corn following CRP and was 32% greater than continuous corn (CC) after 3 years. High N uptake efficiencies of corn following ALF helped to slow buildup of RSN, but levels were equal to row crop systems after two years. Nitrate losses and concentrations in tile drainage remained low during the initial year of conversion, but were similar to row crop systems during the subsequent two years. Thus, low tile flows and nitrate losses will likely require a rotation of perennial and annual crops in the Upper Midwest.

Technical Abstract: Nitrate losses through subsurface tile drains pose a serious threat to surface water quality. Large reductions in drainage losses of nitrate can be achieved with alfalfa (Medicago sativa L.)or perennial grasses often used in Conservation Reserve Program (CRP). Conversion of alfalfa or CRP back to row crops could have rapid, adverse affects on water quality. Our objectives were to evaluate how prior perennial crops affect water and N use efficiency of annual row crops, and losses of water and nitrate to subsurface tile drains. Four cropping systems [continuous corn (Zea mays L.), corn-soybean [Glycine max (L.) Merr.], alfalfa (ALF), and CRP] were established in 1988. ALF and CRP were converted to a corn-corn-soybean sequence from 1994 through 1996 while continuous corn (CC) and corn-soybean (CS) rotations were maintained. Beneficial rotation effects occurred following CRP including a 14% increase in corn yield and a 20% increase in water use efficiency (WUE) as compared to CC. Yield was 19% and WUE 21% greater for soybean following corn in CRP and ALF as compared to CS. Tile flow volumes were correlated to water supplies (Ws) and drainage differences were small following conversion of CRP and ALF to row crops. Residual soil nitrate(RSN) in the top 1.5 m increased by 125% in first year corn following CRP and was 32% greater than CC by 1996. High N uptake efficiencies of corn following alfalfa helped to slow buildup of RSN, but levels were equal to row crop systems after two years. Nitrate losses and

concentrations in tile drainage remained low during the initial year of conversion, but were similar to row crop systems during the subsequent two years. Thus, low tile flows and nitrate losses will likely require a rotation of perennial and annual crops in the Upper Midwest.