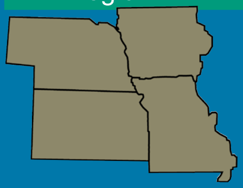
RP189
(Revised May 2013)

Agricultural Nitrogen Management for Water Quality Protection in the Midwest

HEARTLAND
Regional Water Coordination Initiative

EPA Region VII



University of Nebraska–Lincoln
Iowa State University
Kansas State University
University of Missouri-Columbia
USDA-ARS

CSREES · Iowa State University · K-State Kansas State University · University of Missouri Columbia · University of Nebraska Lincoln · EPA Region 7 · National Water Program

Corresponding author
Charles Wortmann, Nutrient Management Specialist
University of Nebraska–Lincoln

Co-authors
Iowa State University
Mahdi Al-Kaisi, Soil and Water Management
Matt Helmers, Agricultural and Water Resources Engineer
John Sawyer, Extension Soil Fertility Specialist

Kansas State University
Daniel Devlin, Water Quality Specialist
Charles Barden, Forestry Specialist

University of Missouri-Columbia
Peter Scharf, Soil Fertility Specialist

University of Nebraska–Lincoln
Richard Ferguson, Soil Fertility Specialist
William Kranz, Irrigation Engineering Specialist
Charles Shapiro, Soil Scientist – Crop Nutrition
Roy Spalding, Hydrochemist
David Tarkalson, Soil Fertility Specialist
John Holz, Water Quality Specialist

USDA-ARS
Dennis Francis, Soil Scientist
Jim Schepers, Soil Scientist

The Heartland Regional Water Coordination Initiative is a partnership of Iowa State University, Kansas State University, the University of Missouri, the University of Nebraska–Lincoln, and the USDA Cooperative State Research, Education and Extension Service. The Heartland Initiative creates and strengthens multi-state, multi-institutional partnerships and collaboration to make research, education, and extension resources of the land grant universities more accessible to federal, state, and local efforts on regional priority water issues.

This material is based upon work supported by the Cooperative State Research, Education and Extension Services, U.S. Department of Agriculture, under Agreement No. 2004-51130-02249.

Introduction

Nitrogen is an essential nutrient for growth of crops and aquatic vegetation and often needs to be applied for optimal crop production. Land application of nitrogen in animal manure, biosolids (sewage sludge), and mineral fertilizer can increase the risk of nitrogen entering ground and surface waters.

This publication provides an overview of factors influencing nitrogen loss to ground and surface waters in the four-state Heartland region of Iowa, Kansas, Missouri, and Nebraska. After a discussion of nitrogen in the environment, the implications of agricultural nitrogen management practices for nitrogen loss to ground and surface water are discussed. More detail on supporting research is available in several review papers. For specific, detailed management options and recommendations, consult Extension resources for your area (see page 5).

Nitrogen exists in many forms in soil and water systems, and these are quite reactive. The various nitrogen forms have unique and important chemical, biological, and environmental properties and occupy specific and important roles in the nitrogen cycle (*Figure 1*). The nitrogen cycle has several major components:

1) Nitrogen is added to soil and water from various natural and industrial processes of conversion of elemental nitrogen gas (N_2) to ammonium, organic sources such as manure and plant residue, and atmospheric deposition.

2) Ammonium is released from organic matter through microbial decomposition.

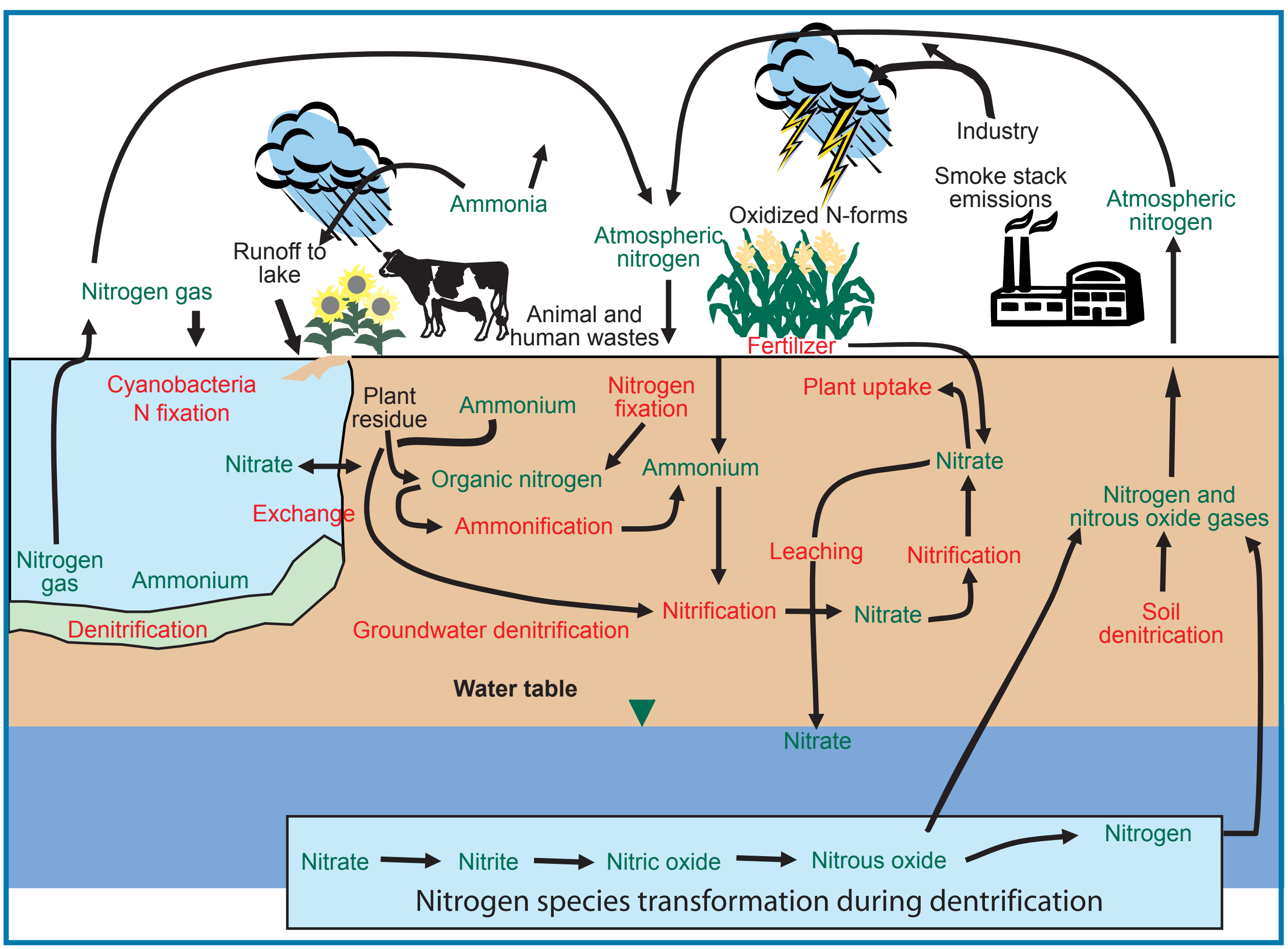
3) Some ammonium is taken up by plants or soil microbes and some is lost via ammonia volatilization, but most is converted to nitrate by soil microbes through the process of nitrification. Nitrification occurs when the soil is sufficiently warm for ammonium to be oxidized by Nitrosomonas bacteria to nitrite and by Nitrobacter bacteria to nitrate.

4) Ammonium and nitrate are taken up by plants and soil microbes and converted to organic nitrogen forms.

5) Nitrogen is lost to surface waters and groundwaters through overland flow and leaching and belowground movement of nitrate.

6) The cycle is completed when nitrate is converted to various nitrogen gases through denitrification.

Figure 1. Processes and nitrogen species involved in the nitrogen cycle. The cycle begins with nitrogen in its simplest stable form, dinitrogen (N_2), and continues through the processes of man-made synthesis and natural fixation to nitrification, leaching, plant assimilation, ammonia volatilization, denitrification, mineralization, and mobilization.



Denitrification is an important process and reduces nitrate concentration in soil solution, groundwater, and surface water, releasing nitrogen to the atmosphere as N_2. Some is also lost as N_2O, a greenhouse gas. Denitrifying bacteria are common in most soils, fresh water bodies, and aquifers. In anaerobic conditions with a carbonaceous energy source, denitrifying bacteria sequentially reduce nitrate to nitrite, nitrite to nitric oxide, nitric oxide to nitrous oxide, and nitrous oxide to nitrogen gas.

$$\underset{\textbf{(Nitrate)}}{\mathbf{NO_3^-}} \longrightarrow \underset{\textbf{(Nitrite)}}{\mathbf{NO_2^-}} \longrightarrow \underset{\textbf{(Nitric oxide gas)}}{\mathbf{NO}} \longrightarrow \underset{\textbf{(Nitrous oxide gas)}}{\mathbf{N_2O}} \longrightarrow \underset{\textbf{(N gas)}}{\mathbf{N_2}}$$

Most natural ecosystems evolved under conditions of scarce bio-available nitrogen. Human activity has increased nitrogen availability in most aquatic ecosystems. Human-driven fixation of nitrogen (e.g., for nitrogen fertilizer production, burning of fossil fuels, and burning of biomass) probably exceeds natural fixation (biological nitrogen fixation and fixation during electrical storms) of atmospheric nitrogen.

Nitrate-N consumption has been linked to human health problems, including methemoglobinemia ("blue baby syndrome"). Because of this toxicity to infants, nitrate-N has an allowable maximum contaminant level (MCL) of 10 ppm (mg/L) in drinking water. Nitrite-N has an MCL of 1 ppm, but is chemically unstable in most aquatic environments and seldom exceeds its MCL.

To Dig Deeper

For more information on agricultural nitrogen management for water quality protection in the Midwest, check these resources:

Nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils, 2002. Agronomy Journal, 94:153-171.

Water quality effects of drainage in humid regions, 1999. Agronomy Number 38. Agronomy Society of Agronomy. Madison, WI.

Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: a review of current science and regulations.

Managing Farming Systems for Nitrate Control: A Research Review from Management Systems Evaluation Areas. 2001. Journal of Environmental Quality, 30:1866-1880.

Nitrate losses to surface water through sub-surface, tile drainage. G.W. Randall and M.J. Goss. In Nitrogen in the Environment: Sources, Problems and Management. Elsevier Science.

Nitrogen application timing, forms, and additives.

Land application of manure for beneficial reuse, 2001. National Center for Manure and Animal Waste Management. 46 pages.

Gulf Hypoxia and Local Water Quality Concerns Workshop, draft proceedings, 2005. Nitrogen rates.

Determining crop available nutrients from manure, University of Nebraska–Lincoln NebGuide G1335.

Concepts and rationale for regional nitrogen rate guidelines for corn. 2006. PM 2015. Iowa State University Extension, Ames, Iowa (*http://www.extension.iastate.edu/Publications/PM2015.pdf*).

Sources of Water Pollutants

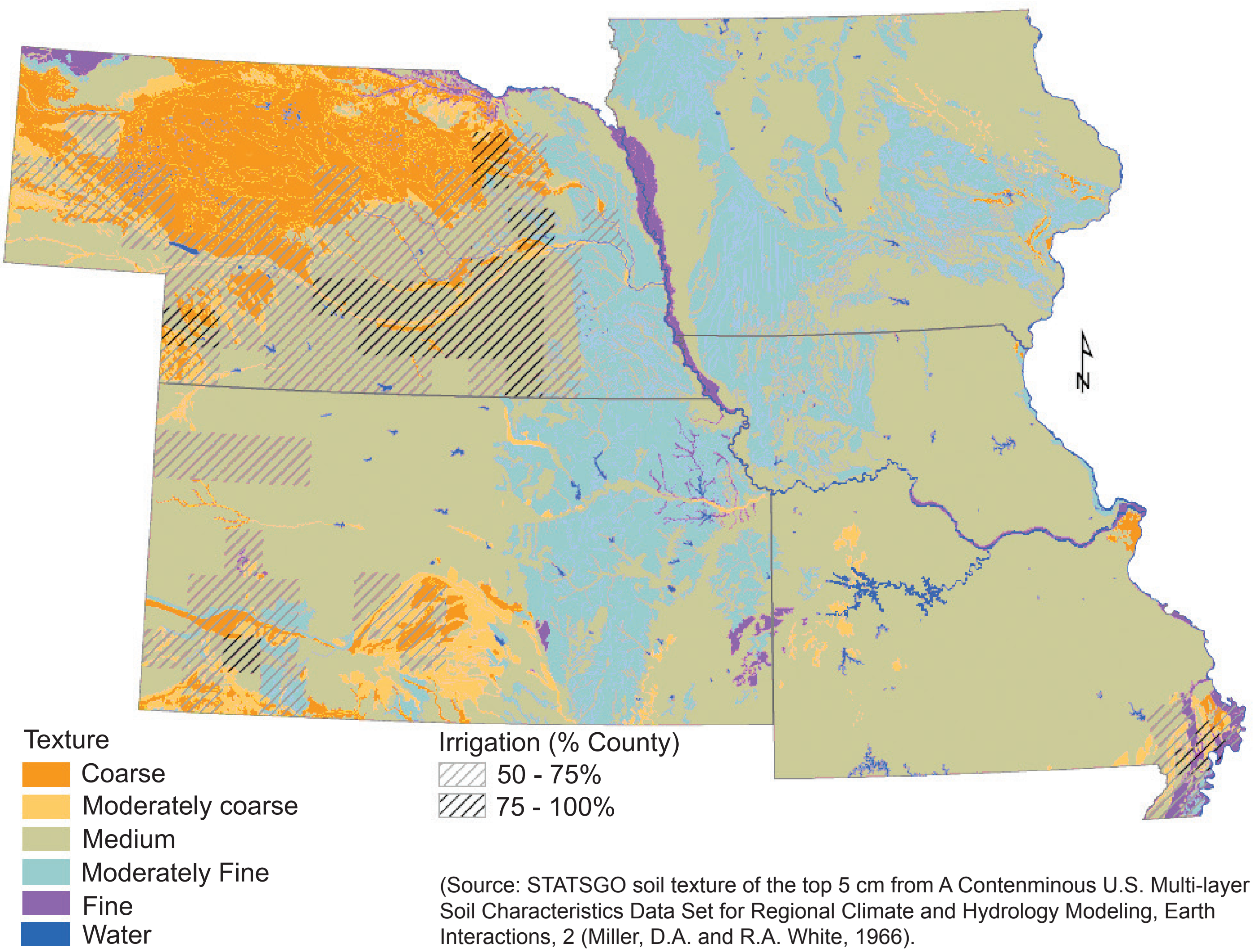
Point source pollution (PSP): Contamination plumes are usually small in areal expanse and may be traced back to their point of origin.

Nonpoint source pollution (NPSP): Groundwater NPSP normally impacts large areas. Chemicals in commercial fertilizers and precipitation are examples of potential nonpoint water pollutants from diffuse sources.

Nitrogen in Groundwater

Protecting groundwater from nitrate leachates is important as groundwater supplies 39 percent of the public drinking water for cities and towns and 96 percent of the water for domestic self-supplied systems (Nolan et al., 1998). Point sources of nitrate contamination of groundwater may be related to the disposal of human and animal sewage, fertilizer manufacturing and distribution, food processing, munitions and some poly-resin manufacturing, and leaks from fertilizer storage tanks.

The primary source of nitrate in groundwater in this region is nonpoint and is especially evident beneath irrigated cropland in areas with shallow depths to groundwater (Spalding and Exner, 1993). Most nitrate that is leached below the root zone to a coarse-textured vadose zone (the aerated zone above the permanent water table) eventually reaches groundwater. Not all leached nitrate reaches groundwater. Some may be denitrified in wet, fine-textured vadose zones. In other cases, leached nitrate may perch on a more or less impervious zone, move laterally by subsurface drainage, and emerge at the surface as seepage.



(Source: STATSGO soil texture of the top 5 cm from A Contenminous U.S. Multi-layer Soil Characteristics Data Set for Regional Climate and Hydrology Modeling, Earth Interactions, 2 (Miller, D.A. and R.A. White, 1966).

Figure 2. Irrigation and soil texture can be especially important to the potential for leaching and denitrification losses of nitrogen.

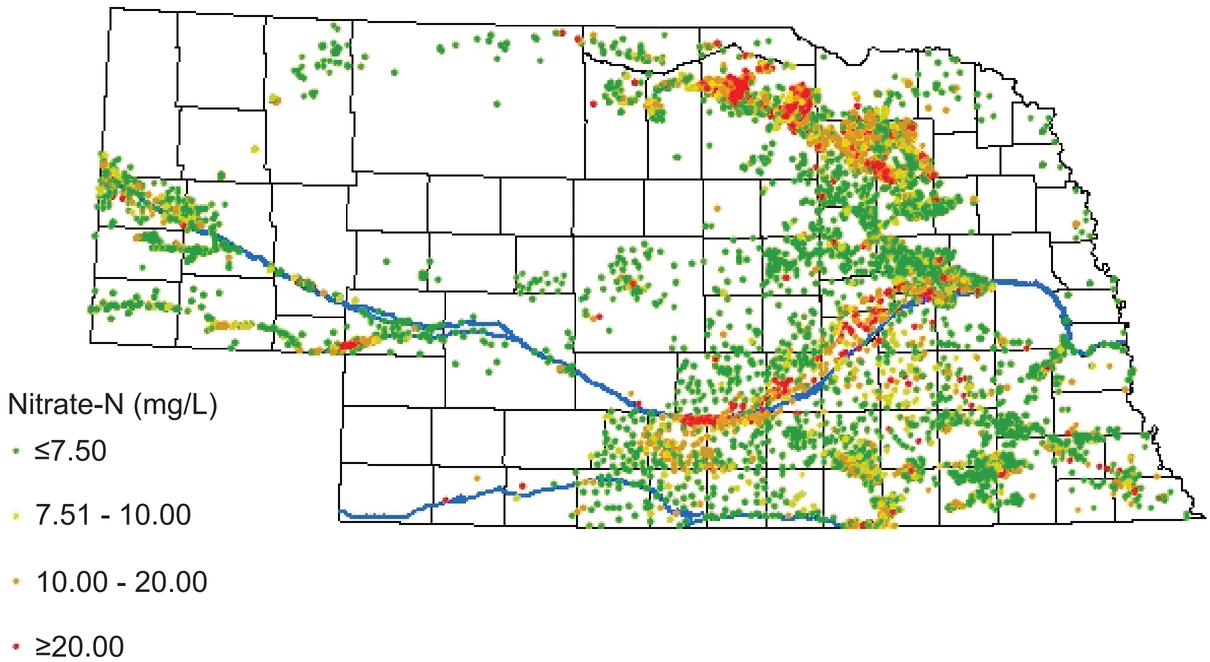


Figure 3. Nitrate-N concentrations in wells sampled in Nebraska in 2007-2011.

Areas most adversely affected by nitrate contamination generally occur in irrigated areas with coarse textured soils such as in fluvial bottomlands and terraces with less than 50 feet to the water table (*Figures 2* and *3*). The more than 500,000 contiguous acres of the Central Platte region is underlain by the largest expanse of nitrate-impacted groundwater in Nebraska.

The rate and quantity of nitrate leaching is related to water infiltration and flow rates through the soil and vadose zone. In the more vulnerable areas, infiltrating rainwater and irrigation water transporting nitrate may enter the shallow groundwater in less than a year. With deep, fine-textured vadose zones, nitrate may move downward 1 to 2 feet per year. Excessive early spring rainfall or irrigation increases nitrate loading to groundwater, resulting in both an economic loss of nitrogen and reduced groundwater quality. Careful management of both applied nitrogen and water are key to decreasing nitrate leaching.

Nitrogen in Surface Waters

Growth of aquatic vegetation in fresh surface waters commonly reduces concentrations of nitrogen and phosphorus because increasing aqueous concentrations of nitrogen and phosphorus result in increased vegetative growth. When this biomass dies and decomposes, dissolved oxygen is depleted, possibly leading to changes in aquatic flora and fauna such as fish kills and increased growth of blue green algae. The result is reduced quality and recreation potential of the water body. Cropland is the source of most of the nitrogen load that enters streams via surface runoff, subsurface drainage (e.g., tile drains), and groundwater flow.

Ammonium loading of fresh waters is a concern. Sources of ammonium in water include soil, fertilizer, manure, urban wastes, atmospheric deposition, fish and animal excretion, and bacterial decomposition of organic material. The conversion of ammonium to ammonia, which is relatively more toxic to freshwater organisms, is increased when dissolved oxygen concentration, pH, and/or temperature are increased. Fish, followed by invertebrates, are most sensitive to ammonia while vegetation is less sensitive.

Ammonium is also converted to nitrate, which is the major inorganic nitrogen form in most well-aerated water bodies. Much nitrate is taken up by algae and eventually released as ammonium when the biomass decomposes. Much nitrate-N is lost from water bodies to denitrification when oxygen is depleted, such as in wetlands, farm ponds, larger reservoirs, and slow-moving streams.

Transport of Nitrogen to Surface Waters

Erosion, runoff, subsurface drainage, groundwater flow, and atmospheric deposition are the major sources of nitrogen in surface waters.

Most of the nitrogen transported in runoff is organic nitrogen; the concentration would be expected to increase as surface soil organic matter concentration and erosion increase. Much of the organic nitrogen entering water bodies is not immediately available to aquatic vegetation, but a large proportion can become available over time, especially if water cycling and resuspension of sediments occurs. Several processes are involved in erosion, including raindrop splash effect, sheet erosion, rill erosion, and gully erosion. Conservation practices that reduce soil removal and increase sediment trapping reduce the amount of organic nitrogen lost from the field.

Runoff carries some inorganic nitrogen, primarily as nitrate and ammonium, at concentrations that are commonly 3 ppm or less. Nitrate-N is generally leached into the soil and ammonium nitrogen becomes attached to soil particles with precipitation that occurs before runoff begins. However, if a sudden runoff event occurs shortly after surface application of nitrogen, concentration of inorganic nitrogen in runoff may be abnormally high.

Subsurface drainage, including tile and natural drainage systems, is a major mechanism of nitrate transport to surface waters (*Figure 4*). The amount of nitrogen delivered depends on the volume of drainage water and nitrate concentration in the soil solution.

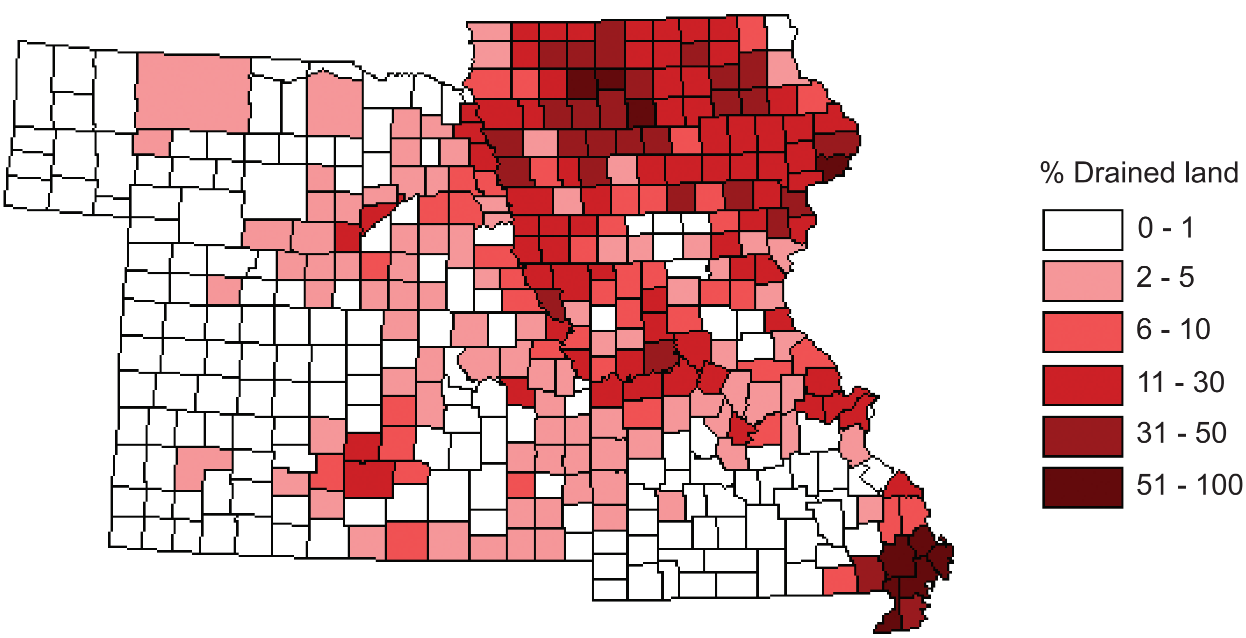


Figure 4. Occurrence of drained land in the four-state region. (Source: National Soil Tilth Lab, USDA-ARS, based on Farm Drainage in the United States, History, Status and Prospects. 1987. Misc. Pub No. 1455, Washington D.C.)

Ammonia Emission and Nitrogen Deposition

Deposition of ammonia and ammonium in nitrogen-scarce aquatic and land-based ecosystems may initiate changes in the competitive relations among plant species with fast growing, nitrogen-responsive species replacing slow growing species.

Ammonium from numerous sources is emitted into the atmosphere as ammonia. Annual global emission of ammonia is estimated at 59 million tons of nitrogen, with about 80 percent of this coming from human-related activities. Agriculture is estimated to contribute 50 to 95 percent of the human-related emissions, including 64 to 86 percent from concentrated feeding operations. Much of the emission in the Midwest is from animal feeding operations, land-applied manure, ammonium-based fertilizer, and crop canopies. Most of the nitrogen emitted as ammonia is deposited on nearby land and surface waters, but some may be removed from the atmosphere by precipitation and carried long distances before deposition.

Total annual ammonia emissions from manure are dependent on livestock density, protein intake, animal species, and manure management practices (*Tables 1 and 2*). A greater percentage of consumed nitrogen is excreted and lost to volatilization as nitrogen content of beef rations increases. A greater percentage of excreted nitrogen is lost to volatilization in the summer compared with winter. Most nitrogen in manure held in uncovered holding lagoons is also lost as ammonia via volatilization to the atmosphere. Much ammonium-N can be lost if land-applied manure is not injected or incorporated.

Table 1. Estimated ammonia emissions from various U.S. livestock operations (source USEPA, 2005).

Animal group	Estimated ammonia emissions (1000 tons/year)	Percent of loss associated with land application
Dairy cows and heifers	558	13
Beef, feeding and breeding	657	3
Chickens and turkeys	664	15
Swine breeding and market	429	10

Table 2. Estimated loss of excreted nitrogen to ammonia emission from beef cattle waste management systems (source: Midwest Plan Service, 1993).

Waste management system	N loss, percent
Open lot - unpaved mounds	40 to 60
Open lot - paved (scraped regularly)	10
Open lot (daily haul)	15 to 35
Stacks, bunkers, bedding packs	20 to 40
Earthen pit	20 to 40
Aboveground storage	10 to 30
Anaerobic lagoon	70 to 80

Ammonia volatilization from anhydrous ammonia and from unincorporated, surface-applied urea containing fertilizer ranges from near zero to as high as 50 percent of the applied nitrogen. Soil properties, especially pH and water content, as well as relative humidity and temperature, affect the rate of ammonia volatilization. Management factors affecting ammonia loss from fertilizer include time, rate, and method of application; fertilizer product; time to incorporation; depth of application; irrigation; and amount of surface residue.

Ammonia emissions from crop canopies have been estimated to contribute up to 15 percent of global ammonia emissions. Net loss of ammonia from leaves occurs when internal ammonia concentrations are higher than that in the ambient atmosphere. This is often the case when available nitrogen substantially exceeds crop needs. Up to 5 percent of aboveground plant N may be emitted from the crop canopy. Biomass burning accounts for 10 to 14 percent of total ammonia emissions globally. The burning of agricultural biomass produces fewer ammonia emissions than fires in natural ecosystems.

Agricultural Nitrogen Management

Nitrogen application is essential to sustainable, highly productive cropping systems comprised largely of non-legume crops. Nitrogen fertilizer use in the United States increased steadily after 1960, reaching a plateau in the 1980s (*Figure 5*). The main use of nitrogen fertilizer in the four-state Heartland

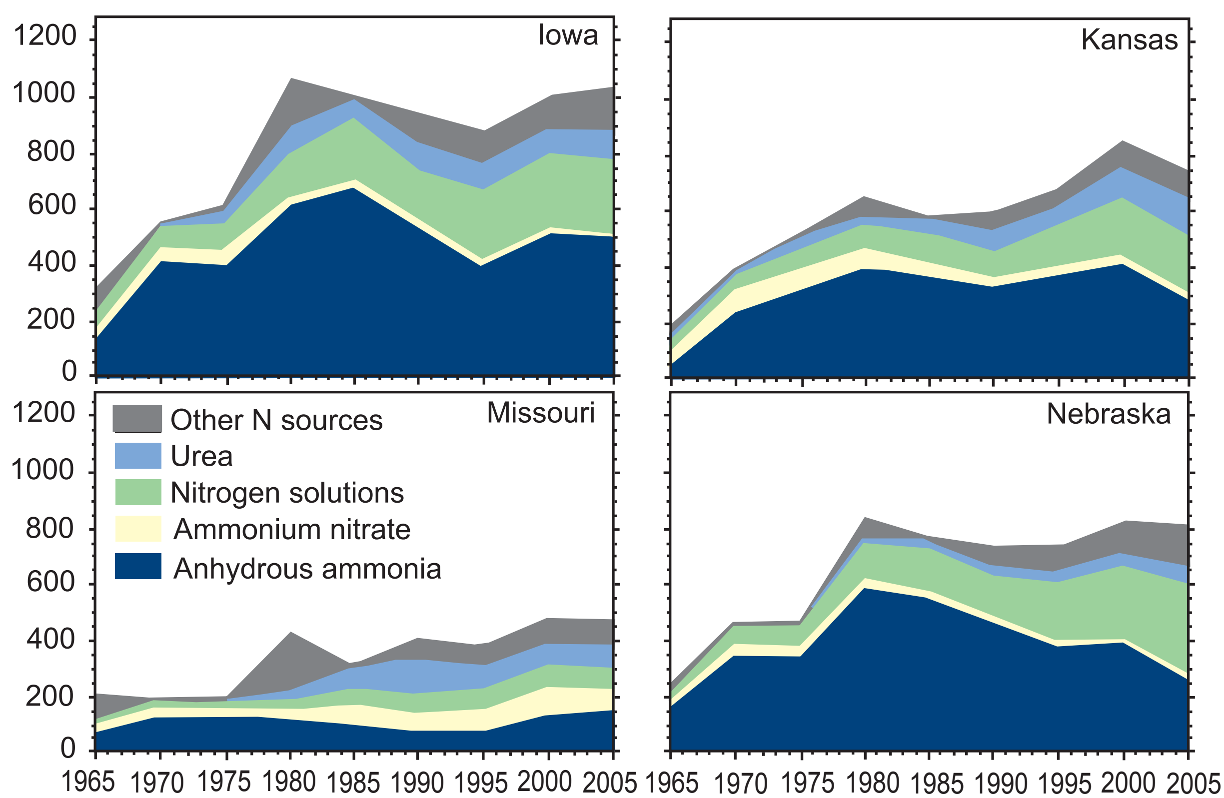


Figure 5. Tons of nitrogen used for agricultural production in each of the four Heartland region states from 1965 to 2005 from selected nitrogen sources. Width of each fertilizer source area represents the fertilizer amount. The width of all fertilizer sources combined represents the total nitrogen use (USDA-ERS, 2006).

region is for corn production (*Table 3; Figure 6*). The benefit of industrially fixed nitrogen to people is widely acknowledged, but there is ample evidence that in many places, excessive use of fixed nitrogen is diminishing the net benefit. Agriculture is the major source of nitrogen to ground and surface water in the United States. A major goal of agricultural nitrogen management is to supply adequate nitrogen for near optimal crop performance while minimizing the nitrogen loss from the soil/crop system.

A major goal of agricultural nitrogen management is to supply adequate nitrogen for near optimal crop performance while minimizing the nitrogen loss from the soil/crop system.

The crop nitrogen requirement is met from several sources, including mineralization of nitrogen from soil organic matter and crop residues and biological fixation of atmospheric nitrogen. It is also met from external sources such as irrigation water, precipitation, animal manure, and commercial fertilizer. Common nitrogen fertilizers used in the United States include anhydrous ammonia, urea-ammonium nitrate solution (UAN), and urea (*Figure 5*).

Table 3. Nitrogen applied to selected crops in the Heartland region states.

State	Corn grain[a]	Winter wheat[a]	Sorghum[a]
	N used (1000 tons)		
Iowa	826	—[b]	—
Kansas	241	394	131
Missouri	245	63	12
Nebraska	581	38	28

Data from USDA National Agriculture Statistics Service, *http://usda.mannlib.cornell.edu/reports/nassr/other/pcu-bb/*

[a]The data for corn, wheat, and sorghum are for 2005, 2004, and 2003, respectfully. Much nitrogen is applied to grassland as well, but the data were not available.

[b]No data due to low acreage.

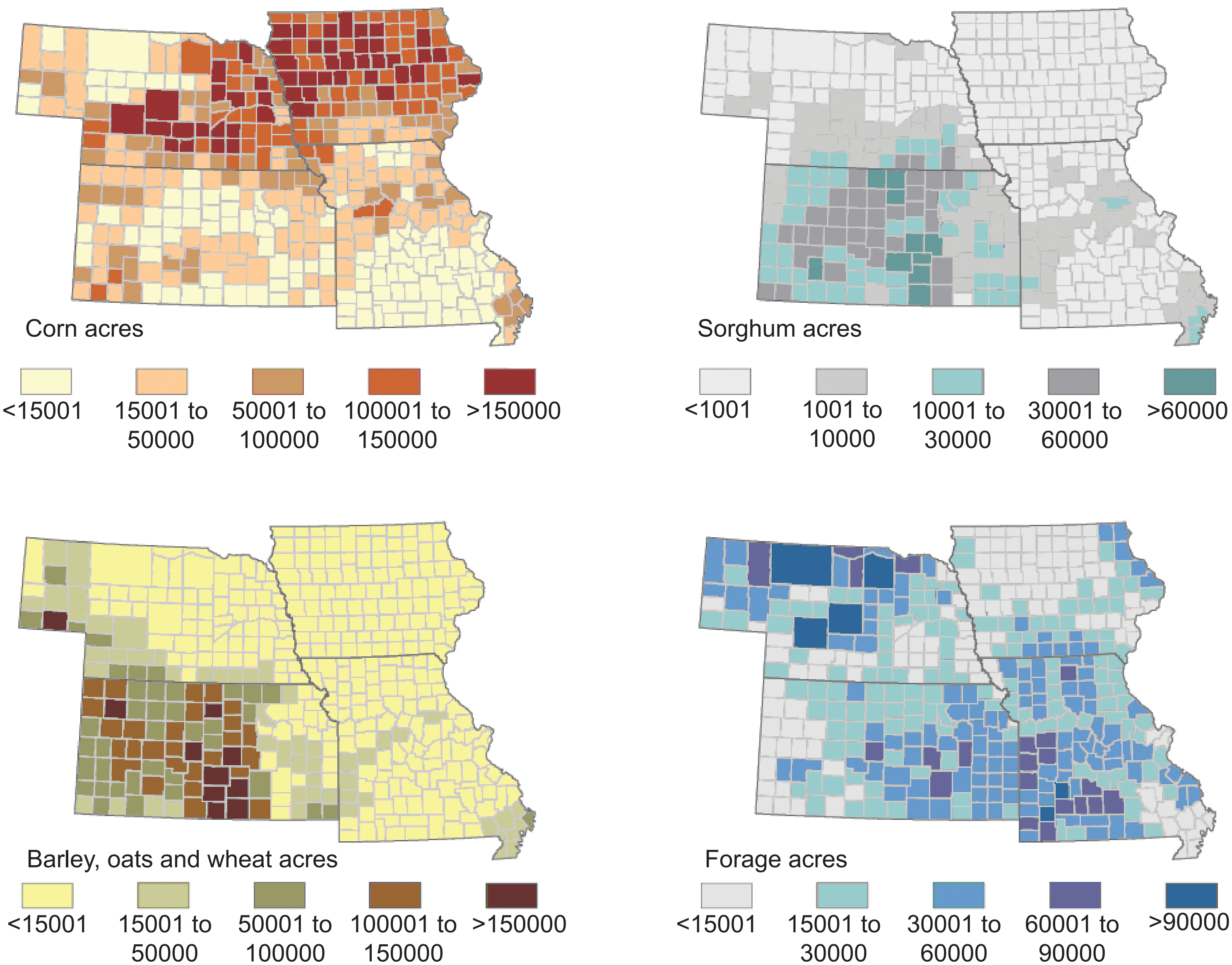


Figure 6. Major crops receiving nitrogen fertilization for the four-state Heartland region, expressed as acres per county. (Data is from the County Summary Highlights table of the 2002 USDA Agricultural Census.)

Nitrogen Application

Account for all nitrogen sources in determining the rate for nitrogen fertilizer application.

The rate, time, and method of nitrogen application can affect the risk of nitrogen loss to surface water and groundwater. Leaching of nitrate to groundwater and nitrate in subsurface drainage is typically more concentrated with higher nitrogen rates (*Figure 7*), but the effect of nitrogen rate varies across locations. Nitrogen rate determination needs to not only consider effect on crop yield, but also profitability and nitrate in subsurface drainage and leachate. Achieving a balance of productivity, profit, and water quality protection is the goal for nitrogen rate guidelines.

Nitrate concentration in soil solution and tile-flow increases continuously with increasing nitrogen application rates and may increase more rapidly at rates above the economic optimum nitrogen rate. When no nitrogen is applied, there is a baseline nitrate concentration in subsurface drainage from cropland. This baseline concentration or load varies with climate, crops, soil properties, and tile system characteristics, but is often 3 to 10 ppm or 8 to 20 lb/acre/year nitrate-N. The concentration of nitrate-N in subsurface drainage increases above this baseline with increasing nitrogen rates.

Nitrogen rate guidelines are designed to determine economically optimum rates of nitrogen while considering nitrogen available from other sources. Therefore, it is important to account for all fertilizer nitrogen sources and subtract these amounts before making the primary nitrogen fertilizer application. Examples include nitrogen in ammoniated phosphate fertilizer (10-34-0, 11-52-0), starter fertilizer, "weed and feed" herbicide – urea-ammonium nitrate (UAN) solution mixes, and early split-N. For some nitrogen sources, especially manure, appropriate "crop-available" accounting should reflect specific soil and climatic conditions.

The nitrogen application rate can be modified to reduce overapplication or guide additional application when unexpected losses

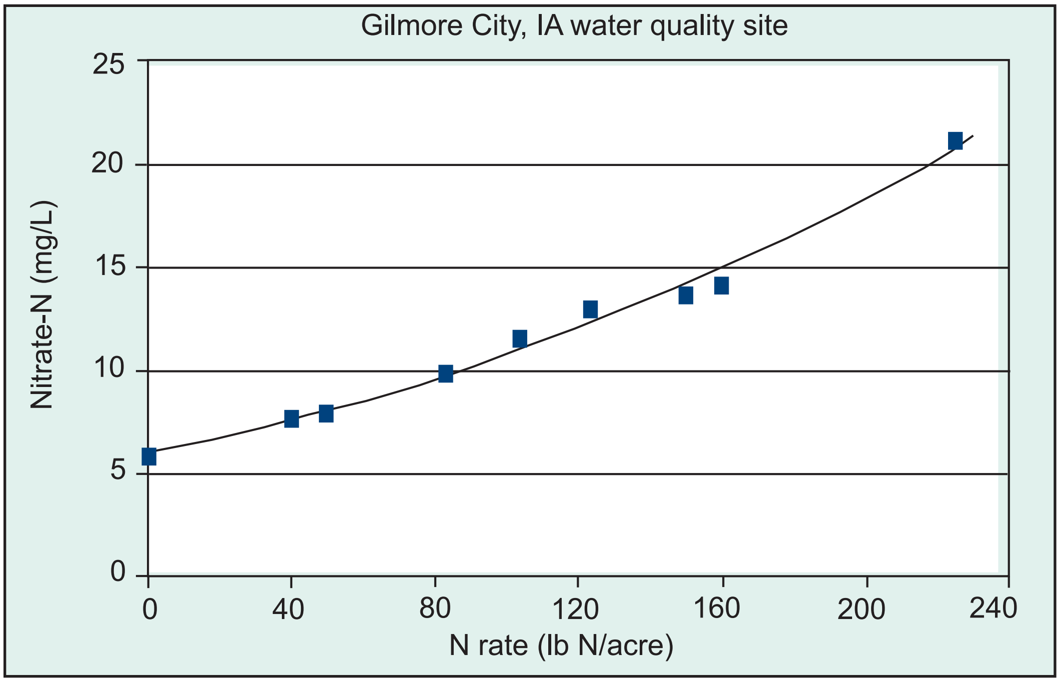


Figure 7. Tile-flow nitrate-N average annual concentrations in a soybean-corn rotation, with nitrogen rates applied in various years from 1990 to 2004 at the Gilmore City, IA, water quality site (adapted from Lawlor et al., 2005).

Resources for Nitrogen Rate Guidelines for Iowa, Kansas, and Nebraska

Iowa

- Concepts and Rationale Nitrogen Rate Guidelines for Corn, PM 2015. *http://www.extension.iastate.edu/Publications/PM2015.pdf*
- Nitrogen Fertilizer Recommendations for Corn in Iowa, Pm-1714. *http://www.extension.iastate.edu/Publications/PM1714.pdf*
- Cornstalk Testing to Evaluate Nitrogen Management, PM 1584, *http://www.extension.iastate.edu/Publications/PM1584.pdf*
- Fertilizing Pasture, Pm-869. *http://www.extension.iastate.edu/Publications/PM869.pdf*
- Sensing Nitrogen Stress in Corn, PM 2026. *http://www.extension.iastate.edu/Publications/PM2026.pdf*
- Using Manure Nutrients for Crop Production, PMR-1003. *http://www.extension.iastate.edu/Publications/PMR1003.pdf*

Kansas

- Soil Test Interpretation and Recommendations. Kansas State University Pub. MF-2586, *www.agronomy.ksu.edu/soiltesting/doc1813.ashx*

Nebraska

- Nutrient Management for Agronomic Crops in Nebraska. University of Nebraska–Lincoln Extension EC155, *www.ianrpubs.unl.edu/sendIt/ec155.pdf*
- UNL NebGuides on fertilizer use, *extension.unl.edu/publications*

occur through use of soil and plant diagnostic testing for crop-available nitrogen. Such tests include:

1) soil nitrate in the fall or preplant in the spring to a depth of 24 or more inches;
2) pre-sidedress soil nitrate in the surface 12 inches when corn is 6-12 inches tall;
3) plant nitrogen stress determination using a hand-held chlorophyll meter, aerial color, and near-infrared images or reflected light sensors;
4) end-of-season stalk nitrate; and
5) post-harvest soil profile nitrate.

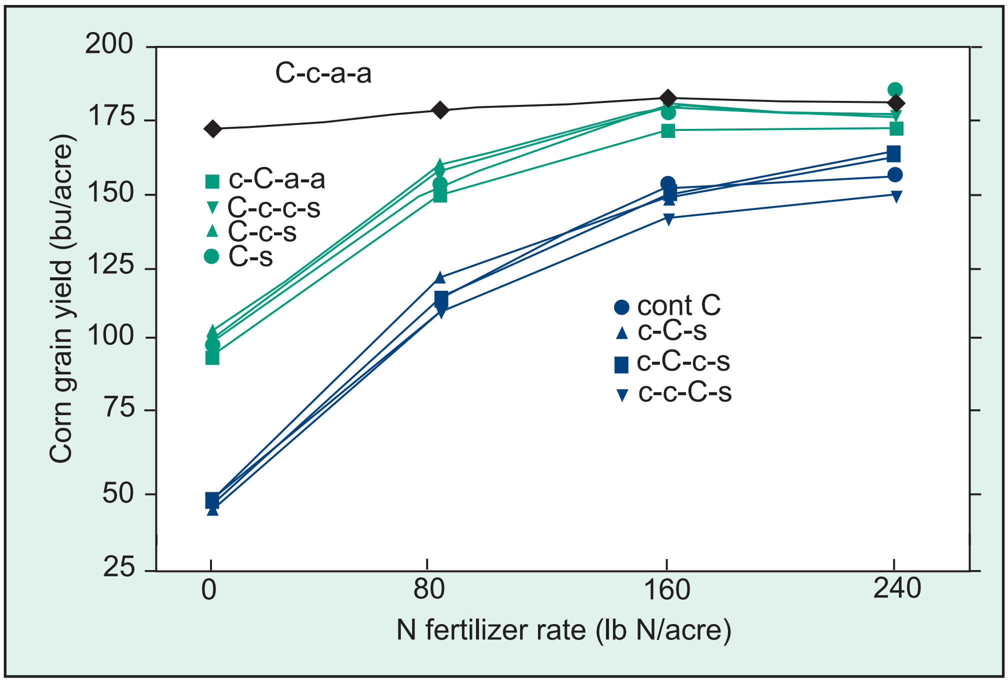


Figure 8. Average rotation and nitrogen fertilizer effects (2001-2004) on corn yield after 23 years at Nashua, IA (adapted from Mallarino et al., 2005). The letters a, c, and s refer to alfalfa, corn, and soybean. The upper case "C" indicates the position in the crop rotation for the corn crop to which the response curve refers.

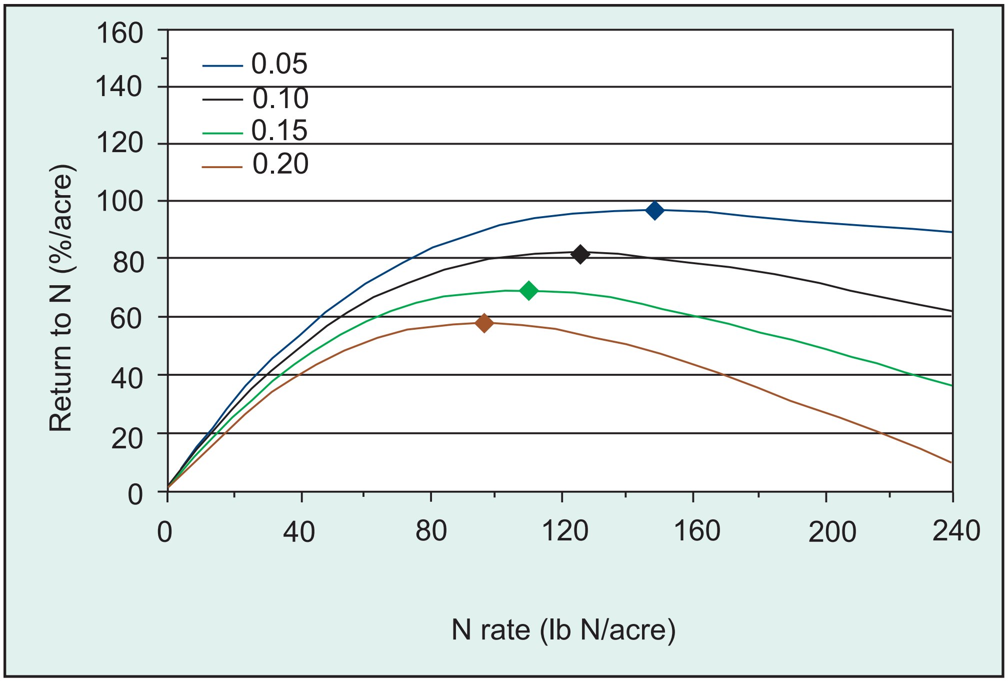
When selecting the right test for your situation, consider whether the test has been validated and calibrated for specific soil and climatic conditions; its cost effectiveness; and its fit with the specific production system.

Criteria considered in nitrogen rate determination vary across the Midwest and are based on research to determine economic return from applied nitrogen for varying locations and cropping systems. A classic example of previous crop and rotation effects on corn response to nitrogen rate is given for a long-term site in northeast Iowa (*Figure 8*). Corn following corn required more applied nitrogen than corn following soybean, and corn following established alfalfa required little to no nitrogen fertilization.

As the price of fertilizer nitrogen increases, the importance of accurate estimation of the most economical application rate increases.

Profitability is the major concern to crop producers. The nitrogen rate that provides maximum return to nitrogen (MRTN) can be calculated for any geographic area, soil, or cropping system if adequate data from nitrogen response trials are available (Sawyer and Nafziger, 2005; Nafziger et al., 2004). Also, the risk of nitrogen inadequacy for crop production at MRTN can be determined. If nitrogen and grain prices remain fairly stable, suggested rates change little (*Figure 9*); however, if nitrogen costs change substantially relative to grain prices (that is, there is a large change in the ratio of nitrogen price to grain value), suggested nitrogen rates can change significantly. As nitrogen prices increase relative to grain value, the loss in profit from applying more or less nitrogen than the economically optimal nitrogen rate increases, requiring more precise nitrogen management.

The environmental cost for nitrate loss to water systems could be added to the economic optimum nitrogen rate analysis; however, society has not required that this cost be partitioned off and identified, and, thus, this information is not available. Inclusion of an environmental cost is likely to reduce suggested nitrogen rates.



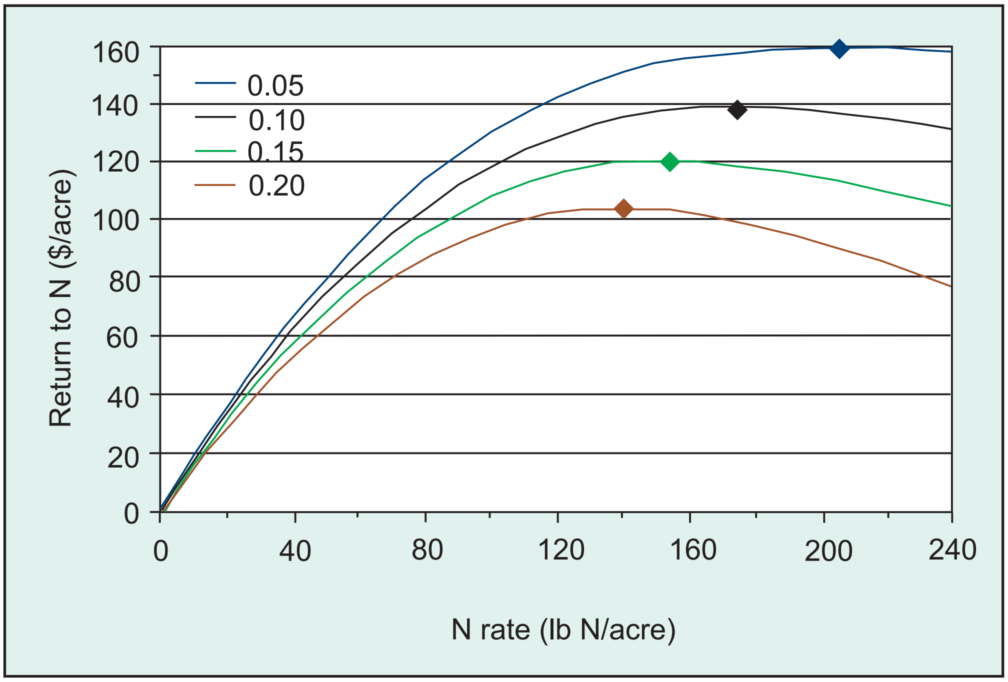


Figure 9. Net economic return to nitrogen and rate at the maximum return to nitrogen (MRTN) for corn following soybean (121 sites) and continuous corn (56 sites) in Iowa. The MRTN is indicated by the solid symbol for each price ratio. Corn grain price held constant at $2.20/bu and nitrogen price varied from $0.11, $0.22, $0.33 to $0.44/lb nitrogen, giving corn:N price ratios of 0.05, 0.10, 0.15, and 0.20, respectfully. Adapted from Sawyer and Nafziger (2005).

Manure Nitrogen

Manure is a valuable source of nitrogen for crop production. Manure-handling technology is improving to lessen management difficulties and uncertainties about nitrogen availability that have caused some producer reluctance to credit manure as a major source of nitrogen. Poorly estimated manure-N application can result from inadequate calibration of the application equipment, uneven rates of manure application, nonuniformity of manure nitrogen content, and the potential for error in estimating ammonium-N loss to volatilization and organic nitrogen mineralization. Uncertainty about using manure as a major nitrogen source has been greatly reduced due to improved application equipment, better management and calibration of the equipment, better manure sampling and interpretation of results, and better understanding of applied nitrogen losses and crop availability. Ammonium can be lost to the atmosphere as ammonia if the manure is applied to the soil surface and not incorporated immediately. The rate of loss to volatilization is

More accurate tests and improved application equipment and management have made it easier to use manure as a major nitrogen source.

affected by temperature, humidity, wind speed, soil pH, manure water content, manure particle size, and other factors. Ammonium-N is quickly converted to nitrate-N when soil temperatures are warm.

Much of the nitrogen in animal feces is in organic forms from undigested feed such as amino acids and proteins. To be available to plants, organic nitrogen must be mineralized to ammonium through microbial digestion. Mineralization of organic nitrogen begins during storage and continues after land application. Mineralization occurs for several years with some manure sources. The rate of organic nitrogen mineralization depends on environmental conditions, livestock species, manure storage methods, and the carbon to nitrogen (C/N) ratio of manure. Generally, the longer manure is stored after excretion, the slower the mineralization of organic nitrogen after land application as nitrogen from the more easily decomposed material mineralizes during storage. In Nebraska, for example, the amounts of organic nitrogen mineralized and available to the first crop are estimated to be 25, 15, and 50 percent for beef cattle feedlot manure, composted feedlot manure and fresh swine manure, respectively.

Organic nitrogen mineralized from manure can be substantial during the years following application, especially with solid manures, and should be accounted for in determining nitrogen fertilization rates. Predictions of organic nitrogen mineralization during subsequent years become less reliable as time since application increases.

Organic nitrogen mineralized from manure can be substantial in the years following application and should be accounted for in determining nitrogen rates.

Most states have standard values to predict the loss of ammonium-N to volatilization and the amount of organic nitrogen mineralized to supply the first crop after manure application. Sources of such estimates include Midwest Plan Service publications, the NRCS Agricultural Waste Management Field Handbook, and state Extension publications. Estimates of ammonia volatilization and organic nitrogen mineralization tend to underestimate manure nitrogen availability, which reduces the risk of inadequate nitrogen supply to the crop. Risk of inadequate nitrogen supply is further reduced by applying most of the needed nitrogen as manure and then applying additional fertilizer nitrogen, especially if the manure, such as feedlot manure, contains mostly organic nitrogen and little ammonium-N. Another approach is to check soil nitrogen availability using the pre-sidedress soil nitrate test and to sidedress fertilizer nitrogen if needed, although this does not easily address problems of nonuniform rates of nitrogen application. In-season monitoring with a chlorophyll meter, aerial imagery, or crop canopy reflectance sensors can verify adequate nitrogen supply throughout the field.

Time of manure application may have implications for nitrogen loss to surface water or groundwater.

Time of manure application may have implications for nitrogen loss to surface water or groundwater. Manure applied in the summer following small grain harvest may present little risk to nitrogen loss in runoff and erosion; however, the ammonium will be converted to nitrate long before the next crop needs the nitrogen and the nitrate will be subject to leaching to subsurface drainage or groundwater. A similar problem exists with early fall-applied manure when soil temperatures are above 50°F. Risk of nitrogen loss to runoff and erosion to surface waters is increased with manure application on snow or frozen ground, especially where late winter or early spring melt events result in runoff. Nitrate leaching loss with manure application in the spring should be less than with summer or fall application, but total nitrogen loss in runoff and erosion following surface application may be relatively greater since most runoff occurs in the spring.

Method of manure application may affect nitrogen losses to surface water and groundwater. Injection or incorporation of manure reduces volatilization nitrogen loss, as well as odor, fly problems, and manure runoff. If injection or incorporation greatly reduces ground cover or otherwise increases the potential for erosion, it may result in increased nitrogen loss. The implications of tillage in increasing erosion risk may be minor if the land is typically tilled, but may be significant for no-till where risk of soil nitrogen loss with erosion may be less without incorporation. The risk of nitrogen loss in runoff following surface application of manure diminishes rapidly with time as inorganic nitrogen is carried into the soil with rainfall events and the organic material reacts with the soil, improving soil aggregation and reducing susceptibility to runoff.

Time of Nitrogen Application

Some producers apply nitrogen fertilizer for corn as anhydrous ammonia in the fall when there is more time for application, the nitrogen price is often lower, and the soil is more likely to be dry. These good conditions allow for field operations with little compaction. The disadvantage of fall application is increased risk of loss before the main crop nitrogen uptake period in June and July. Nitrification of ammonium nitrogen will be very slow while the soil temperature remains low after application (e.g., below 50°F). Fall-applied nitrogen may be nitrified before the crop is planted due to fall application when soil temperatures are relatively high, unexpected warming of the soil after application, periodic warming during the winter, and early warming of the soil in spring. This nitrate will be subject to leaching and denitrification with spring rains and waterlogged soils that occur before and after the crop is established. Anhydrous ammonia is slower to convert to nitrate than ammonium from other fertilizers and is the only fertilizer nitrogen source that should be considered for fall application.

Nitrogen use efficiency with fall application can be reduced by 10-20 percent in sandy soils and in warm wet areas.

Nitrogen losses with fall application are expected to be greatest in sandy soils and in the wetter and warmer parts of the region. Nitrogen use efficiency with fall application may typically be reduced by 10-20 percent under such conditions. Several studies suggest an average 8 bu/ac corn yield penalty for fall application of nitrogen relative to spring application. This loss may cancel the benefit of lower fertilizer prices in the fall. This yield loss is associated with nitrogen loss that in most cases will end up in groundwater or surface waters, or be lost to denitrification. Nitrate leaching also occurs with spring nitrogen applications when soil water exceeds the water-holding capacity of the soil and there is little or no nitrogen uptake by the crop.

Nitrogen use efficiency may be increased and nitrate leaching reduced by applying a major part of the nitrogen in-season, at or near the time when crop nitrogen demand is high. Determination of surface soil nitrate-N levels at about six weeks after planting with the pre-sidedress nitrate test (PSNT) assesses nitrogen availability after accounting for the cumulative effects of residual nitrogen, applied nitrogen, nitrogen mineralized from soil organic matter, and nitrogen losses. The PSNT results are then used to determine how much nitrogen should be applied during the season. More accurate estimation of crop nitrogen need on fine-texture soils is likely to be more important to improved nitrogen recovery than the timing of application.

Many producers are reluctant to apply nitrogen in-season as they may be busy with other operations, concerned about yield loss due to early nitrogen stress, or concerned that wet weather will prevent application. Delayed in-season nitrogen application may reduce yield, but this loss can be avoided or minimized by applying a portion of the needed nitrogen at planting.

Applying nitrogen through irrigation systems (fertigation) is an important form of in-season nitrogen application in irrigated areas. Fertigation can be a very efficient method of nitrogen application, but must be practiced with appropriate safeguards against backflow contamination of groundwater if irrigating from wells. Fertigation has the risk of needing to apply nitrogen to already moist soil if adequate rainfall coincides with the time for nitrogen application. Nitrogen application through irrigation systems should not be delayed beyond the period of maximum nitrogen demand by the crop and generally should be complete by the silking stage of corn.

Variable Rate Nitrogen Application

Soil nitrogen supply, crop nitrogen demand, and potential for nitrogen loss to surface water and groundwater vary within fields and landscapes. Identification and interpretation of this spatial variability provides a basis for variable rate application of nitrogen as a means to reduce nitrogen loss to groundwater and surface water, although the basis for interpretation of this spatial information in terms of optimal nitrogen application rates needs to be improved.

Yield maps or aerial images of the crop are valuable and increasingly available spatial information. Consistent poor crop performance in one part of the field may indicate greater potential for nitrogen loss if nitrogen is applied uniformly across the field. Variation in soil organic matter and soil texture can be important to efficient nitrogen management. Soil maps, bare soil images, grid soil sampling, and/or mapping of electrical conductivity may indicate this variation, although the variation often is not enough to justify variation of nitrogen rates.

In-season application of nitrogen according to crop growth and canopy reflectance is a developing technology. Crop canopy color is well-correlated with leaf nitrogen concentration. Crop nitrogen requirement can vary over short distances. Technology exists for automated, spatially intensive adjustment of application rates in response to variation in canopy color using applicator-mounted sensors or aerial images; however, interpretation of the sensed data in terms of optimal nitrogen application rates is still a developing science.

Soil nitrogen levels and crop needs often are not defined by field borders. Variable rate nitrogen application can place fertilizer where it's needed at the rate it's needed, reducing loss of nitrogen to water resources.

Different canopy color sensors are best suited for different situations. Hand-held chlorophyll meters are useful for trouble-shooting, spot checking, and determining if a whole field or management zone needs nitrogen (*Figure 10*).

Applicator-mounted sensors offer promise for on-the-go adjustments of in-season nitrogen rates. The sensors may be passive or active. Passive sensors use natural light and determine the difference between in-coming and reflected radiation, thereby adjusting for variations in the intensity of in-coming radiation due to cloud movement, sun angle, and time of day. Active sensors provide a controlled light and can be used in low light situations. A drawback of the applicator-mounted sensor option is that the leaf area needs to be sufficiently developed to reflect enough light to reliably indicate nitrogen need. This increases risk as wet weather may delay or prevent sidedress application. The technology for in-season variable rate nitrogen application through fertigation is improving.

Aerial imagery is useful once the crop canopy is sufficiently developed and soil reflectance no longer dominates the image. Aerial photos are particularly suited for surveying large areas, such as when wet weather creates potential for nitrogen loss. Aerial photos potentially can be calibrated to predict likely yield gain from applying additional nitrogen.

Figure 10. Hand-held chlorophyll meters are clamped over a leaf to measure light transmittance through a leaf.

Inhibitors and Controlled Release Nitrogen Fertilizers

Inhibitors and controlled release nitrogen fertilizers should be considered a form of insurance against nitrogen loss, rather than a guarantee of increased yield. If conditions are not conducive to nitrogen loss (volatilization, denitrification, or leaching), there will be no benefit to using these products.

Nitrification inhibitors suppress Nitrosomonas bacteria and slow the conversion of ammonium to nitrate (*Figure 1*). This suppression results in reduced potential for denitrification and nitrate leaching and increased potential for nitrogen use efficiency. Nitrification inhibitors are useful only with fertilizers containing predominately ammonium-N, such as anhydrous ammonia. Nitrification inhibitors are most useful with preplant nitrogen application on sandy (excessively well-drained) soils prone to leaching or with fall nitrogen application on poorly drained soils subject to denitrification.

Urease inhibitors temporarily block the function of the enzyme urease in soil, which facilitates the conversion of urea to ammonium. By blocking urease activity, the potential for ammonia volatilization of surface-applied urea to the atmosphere is reduced. Urease inhibitors are most useful where urea or fertilizers containing urea are used without incorporation, such as with broadcast application for no-till systems or surface application to high residue or high pH soils, and when a substantial rainfall or irrigation event is unlikely for several days after application. Potential for loss increases with temperature.

Both nitrification and urease inhibitors are effective for a limited time, as they gradually decompose in soil and lose their efficacy. Under normal conditions inhibitors are effective for approximately one to three weeks, but much longer with cool, dry conditions.

Nitrogen losses to water through leaching may be reduced with controlled-release fertilizers. Controlled-release nitrogen reduces how much nitrate is in the soil solution at any one time, and less nitrate is lost when leaching occurs. Controlled-release nitrogen fertilizers generally fall in two categories: coated products and those which are slow-release due to their chemical structure. Coated products include sulfur-coated urea and polymer-coated urea. Fertilizers composed of methylene-urea chains are chemically slow release. With increased nitrogen prices and advances in manufacturing techniques, these fertilizers may be alternatives to inhibitors or sidedress application for increasing nitrogen use efficiency.

Cultural Practices and Nitrogen Delivery to Water Bodies

Cultural practices affect crop performance, residue cover, soil aggregation and structure, soil organic matter, and other factors that affect soil and nutrient losses to water resources. Conservation and no tillage systems often result in reduced surface runoff and increased preferential soil water flow compared with other tillage systems that can lead to deeper nitrate movement into the soil profile.

More than 30 percent of the cropland in the Midwest is tile drained (*Figure 4*), with a significant impact on soil hydrology and nitrate movement through the soil profile. In one three-year study in Iowa, nitrate removal through tile drainage ranged from 21 to 56 lb/ac, depending on cultural practices and annual rainfall (Kanwar and Baker, 1993). Nitrate-N losses were much lower with perennial, compared to annual, cropping systems in a study conducted in Minnesota. In comparing continuous corn and corn-soybean rotation, nitrate leaching was reduced and yield was increased with rotation (Randall et al., 1993a; Kanwar et al., 1997).

Conservation Practices to Reduce Nitrogen Loss to Ground and Surface Waters

Crop rotations often result in lower nitrogen application rate, better use of applied nitrogen, less nitrate leaching and less nitrogen loss in erosion.

Residue management often results in better ground cover and less nitrogen loss in erosion.

Tillage practices affect ground cover and water infiltration thereby affecting erosional loss of organic nitrogen and nitrate leaching.

Grassed waterways reduce gully erosion and trap sediment to reduce nitrogen loss in erosion.

Terraces reduce erosion and trap sediments, resulting in reduced nitrogen loss in erosion.

Conservation buffers trap sediment and increase infiltration for reduced nitrogen loss in erosion.

Contour farming reduces erosion loss of organic nitrogen.

Tillage Practices

Tillage affects the rate of mineralization of soil organic matter, crop residue, and manure nitrogen, with higher rates of mineralization resulting from increased mixing of the organic material with soil. Increased tillage increases decomposition of crop residues as decomposing microbes are more protected from extreme temperature and moisture fluctuations than on the soil surface, and nitrogen is more available for decomposition of materials with high C/N ratios. Organic nitrogen becomes available for use by plants and soil microbes as the organic matter decomposes, but excess nitrate may be leached to groundwater or to subsurface drainage.

Tillage affects soil physical properties and nitrate movement through the soil profile.

Tillage systems affect soil physical properties and nitrate movement through the soil profile. The rate of water infiltration is often improved with continuous no-till due to increased soil organic matter in the top few inches of surface soil. This results in an increase in soil pore size and preferential flow, which may increase leaching of nitrate. Greater channel development in no-till soil may result from root growth and increased activity of soil organisms such as earthworms, but also because these channels are not regularly destroyed during tillage.

In other cases, nitrate has been observed to move through soil at a slower rate with no-till and ridge-till than with more intensive tillage systems. The increased risk of nitrate leaching due to greater preferential flow with no-till can be countered by more nitrogen mineralization and greater water infiltration of spring precipitation with tillage to the extent that there is more leaching of nitrate with tillage than with no-till. On clay pan soils, nitrate leaching occurs primarily with heavy rains following a dry period that leaves the clay pan dry and cracked. In these situations, leaching may be greater with tillage than with no-till due to more soil cracking and preferential flow; however, much leached nitrate is likely to be denitrified from such clay soils before the nitrogen reaches surface or ground waters. In short, total nitrate leached may be more, less, or the same when comparing tillage systems.

Tillage influences nitrogen movement to surface water in runoff. Increased ground cover and water infiltration with reduced or no tillage serve to reduce organic nitrogen movement, which represents the major part of runoff nitrogen loss. Generally, nitrate and ammonium loss in surface runoff is minor compared to organic nitrogen loss for all tillage systems.

Crop Residue Cover

Living and dead ground cover absorbs the energy and splash effect of falling raindrops and thereby reduces the potential for soil erosion. Crop residues on the soil surface greatly reduce sediment transport by reducing the velocity of runoff and sediment transport potential (*Table 4*) and by reducing wind erosion. Leaving crop residue on the soil surface may improve nutrient cycling and soil productivity. When harvesting corn residues for animal feeding or lignocellulosic biofuel, the producer needs to consider impacts of the above benefits of crop residue cover of the soil.

When averaged over the length of the corn-soybean rotation, the mean annual rate of nitrogen application is typically less than half as much for the rotation as compared to continuous corn.

Cropping Systems

Cropping systems that synchronize nitrogen availability with crop uptake should have less nitrogen loss. Multiple and diverse cropping systems typically are more favorable to water, soil, and air quality than an annual or even biannual row crop system. Less nitrate leaching, higher yields, and better fertilizer nitrogen recovery often occur, for example, with a corn-soybean rotation rather than with continuous corn. When averaged over the length of the rotation, the mean annual rate of nitrogen application is typically less than half as much for the rotation compared to continuous corn.

Table 4. Crop residue cover after planting and relative soil loss for various tillage and manure management systems.[a]

Tillage and/or manure management system	Residue cover, %	Relative soil loss, %
Tillage and manure application effects on residue cover		
Fall shovel injected, fall chisel plow, spring field cultivation, planting	23	100
Fall shovel injected, spring field cultivation, planting	29	88
Fall disc-covered, fall chisel plow, spring field cultivation, planting	28	90
Fall disc-covered, spring field cultivation, planting	36	78
Spring slot injected, planting		
Tillage and manure application effects on residue cover		
Moldboard plow, spring field cultivation, planting	1	100
Chisel plow, spring field cultivation, planting	13	44
Fall strip-tillage, spring field cultivation, planting	24	31
Spring field cultivation, planting	34	24
Planting	43	19

[a]Calculations are for a 5 percent slope in central Iowa and are based on an Iowa State University Extension publication, PM 1901a, and a Midwest Plan Service publication, MWPS-45.

Cropping systems that extend the growing season and period of nitrogen uptake are likely to have less leached nitrate. Deep-rooted legumes, such as alfalfa, effectively scavenge leached nitrate-N but also deplete soil water and reduce deep soil water percolation. Despite the potential for positive impact on groundwater quality, inclusion of such legumes in the rotation may reduce profitability where there is little demand for such products.

Cover Crops

Cover crops may reduce nitrogen loss in runoff and leaching loss of nitrate-nitrogen.

Well-established cover crops reduce nitrate concentration in the soil solution and the potential for nitrate leaching. Cover crops are usually grown to protect soil from wind or water erosion during the off-season, or to protect another crop (e.g., alfalfa) during establishment. The most obvious benefit of cover crops in terms of protecting water quality is to reduce sediment losses in surface runoff through ground cover and soil stabilization. Indirectly, the biomass produced by the cover crop has positive implications on nutrient cycling because the vegetation either functions as a green manure if it is incorporated into the soil or as residue for ground cover if chemically destroyed and left on the surface.

Integration of cover crops into cropping systems is challenging. The cover crops use soil water that might otherwise be stored for the next crop although this is often not an issue in wetter parts of the region, especially if the cover crops are terminated during the winter or in early spring. Stand establishment is often difficult due to sowing with less than optimal times and conditions. Seed is often sown into heavy crop residue cover. The time between establishment and harvest of the main crop and the onset of cold weather is often short. Legume cover crop establishment is generally slower than for Brassica species or winter cereals such as wheat, rye, or triticale. Fall-sown oats can be an effective cover crop as it establishes easily and quickly to take up nitrate in the fall; oats winter-kills so it does not use soil water in the spring and does not require a herbicide application to kill it.

Riparian Buffers

Riparian buffers are areas of trees or other vegetation located adjacent to a water body and managed to reduce the negative impact of nearby land use (Mayer et al., 2005). Riparian buffers can have several roles including: 1) separating the crop field from the stream; 2) filtering runoff to remove sediment, nutrients, pesticides, and microorganisms; 3) increasing water infiltration; 4) taking up nitrate from shallow groundwater; and 5) stabilizing streambanks.

Buffers reduce nitrogen loading to the stream by: 1) filtering and sedimentation of organic and other particulate-bound nitrogen; increasing infiltration; 2) increasing nitrogen uptake, especially if there is subsurface flow through the root zone; and 3) denitrification. Denitrification may be relatively high with a mature riparian forest, intermediate with a grass buffer, and least with cropland. Well-designed and maintained buffers can trap about 50 percent of incoming sediment, but are less effective in reducing sediment-bound nutrients, and much less effective for reducing surface runoff of dissolved nutrients. Nitrate concentration in shallow groundwater may be reduced by more than 50 percent due to nitrogen uptake by vegetation in the buffer, although uptake is much less if the amount of groundwater interacting with the root zone is small. Mayer et al. (2005) concluded that subsurface removal of nitrogen in riparian buffers is often high, especially where denitrification is induced, for a wide range of vegetation types, while surface removal of nitrogen by buffers is relatively inefficient.

Well-designed and maintained buffers can trap about 50 percent of incoming sediment, but are less effective in reducing sediment-bound nutrients and surface runoff of dissolved nutrients.

Buffers need to be managed to ensure that water flows through as a slow sheet flow and there is vigorous growth of the buffer vegetation. Buffer effectiveness is reduced when uneven field topography results in concentrated flows leaving the field. In such cases, terraces, in-field filter strips, or wetlands may be better options for reducing nitrogen loading.

Subsurface Tile Drainage Management

Installation of artificial subsurface drainage systems at the end of the 19th and first half of the 20th century enabled conversion of poorly or somewhat poorly drained lands in humid areas to highly productive agricultural land. Excess precipitation is removed via subsurface drainage systems that intercept soil water and divert it to surface waters. Drainage allows timely seedbed preparation, planting, and harvesting, and protects crops from extended periods of flooding. More than 30 percent of Midwest cropland is tile drained (*Figure 6*).

Subsurface drainage results in less nitrogen loss in surface runoff. Surface runoff normally contains much higher concentrations of sediment, organic nitrogen, phosphorus, and pesticides than subsurface drainage. Improved subsurface drainage systems also reduce leaching of nitrate to groundwater.

Subsurface drainage, however, discharges nitrate to surface waters. The quantity of nitrate discharge varies with agronomic practices, the site, cropping system, soil, and climatic factors. The discharge of nitrate in subsurface drainage to surface waters in Iowa and other areas in the Midwest often exceeds 25 lb N/acre/year.

Water Table Management

Renovation or reconstruction of drainage systems and construction of new systems provide opportunities to incorporate water quality benefits. Subsurface drainage management, including shallower drain tube installation and controlled drainage for water table management, has potential to reduce the export of nitrate to downstream water bodies. Shallow drainage consists of placing conventional tile drains at shallower depths (e.g., at 2-3 feet rather than 4 feet). Controlled drainage raises the outlet of the drainage system at certain times to raise the water table (*Figure 11*). These modifications can reduce subsurface drainage volume, thereby decreasing the export of nutrients and other pollutants from agricultural landscapes. In addition, by managing the outflow of subsurface drainage there is the potential to store additional soil water for use by the crop.

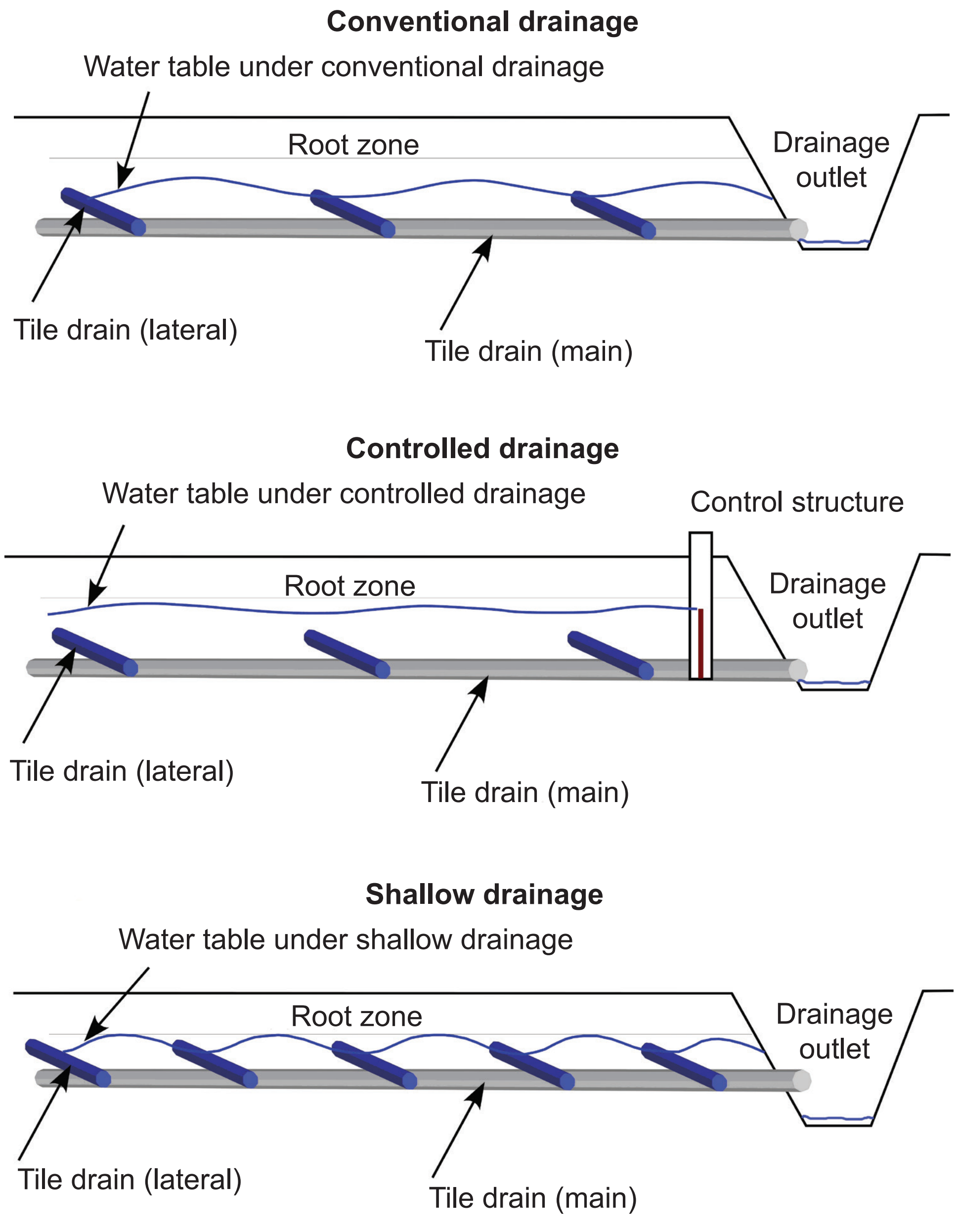


Figure 11. A representation of water table depth with conventional and controlled drainage.

Subsurface drainage volume has been reduced by about 20-40 percent in Ohio, Illinois, and Minnesota research through the use of shallower drainage or controlled drainage, but the technology needs to be validated or adapted for other climatic and soil conditions in the upper Midwest. Shallow drainage requires that drainage lines be closer together in the field, which increases system costs, and controlled drainage requires low slope conditions (0-1 percent).

Controlled drainage requires increased management since slide panels are raised and lowered to manage the water table to drain only as much water as needed for healthy crop growth. The water table is allowed to rise after harvest, and again for a time after planting and early crop germination. During planting and harvesting the water table is lowered to facilitate field operations.

Treatment of Drainage Water

Constructed wetlands. Since the 1980s, wetland restoration in the Midwest has been supported by state and federal programs, primarily for improved waterfowl habitat. However, there often has been inadequate attention to the siting and design for maximum interception of subsurface drainage water for removal of nitrate by denitrification (Crumpton, 2001). Decomposition of vegetative biomass, such as cattails, produced in wetlands consumes dissolved oxygen to create anaerobic conditions necessary for denitrification.

Nitrate discharge to surface waters is further reduced as the "pass-through" drainage water is reduced in the wetland by seepage to groundwater and evapotranspiration. The efficiency of a wetland to remove nitrate by denitrification is significantly affected by temperature (the rate of denitrification is less at lower temperatures), the C/N ratio, the amount of nitrate entering the wetland, and the residence time of the water in the wetland. These last two factors are affected by site selection and sizing of the wetlands in the landscape or watershed. The wetlands must be situated so that drainage water from cropland containing significant amounts of nitrate passes through the wetland. The wetland must be properly designed and large enough that the drainage water has adequate residence time in the wetland for much of the nitrate to be lost to denitrification. The C/N ratio of the water in the wetland is important to denitrification efficiency, with efficiency increasing as the C/N ratio increases.

Effective use of wetlands for nitrate reduction requires interception of a significant nitrate load. The importance of wetland location was demonstrated in a modeling study for central Iowa where the annual nitrate export from the watershed was reduced by less than 4 percent when wetlands intercepted only 4 percent of the total tile drainage. When the wetlands were located to intercept 50 percent of the total drainage before discharging it to the stream, the annual export of nitrate was reduced by about 35 percent. The wetland areas within the watershed were of the same acreage, but the more effective wetlands were situated to intercept a greater volume of drainage water.

The sizing of a nitrate removal wetland relative to the watershed will vary due to climatic factors, but in general a wetland area sized at 1 to 5 percent of the watershed area would be expected to remove significant amounts of nitrate. Expenses involved in use of wetlands for reducing nitrate in drainage water include wetland installation and maintenance, and land lost to crop production.

Biofilters. Denitrification biofilters are constructed around tile drainage lines to reduce nitrate in the drainage water. Woodchips, cornstalks, cardboard fiber, a sand-sawdust mixture, or other cellulose-based materials are placed around the underground tile or in-line with the drainage system prior to discharging the drainage water. The objective of routing drainage water through these biofilters is to create anaerobic conditions and to provide carbon for denitrification to remove nitrate before the water enters the tile or before the water exits to surface water. In Iowa (Kaspar et al., 2002), wood chips surrounding a newly installed tile line reduced nitrate concentrations in subsurface drainage by 65 percent compared to a tile line with no wood chips.

Denitrification in streams and reservoirs. Some denitrification occurs in flowing streams but relatively little compared to a wetland system. Denitrification in reservoirs may be significant with adequate residence time.

Nitrate leaching is increased with application of water in excess of crop needs, either on the whole field or part of the field. An inch of deep percolation can move between 5 and 25 pounds of nitrogen per acre out of the root zone toward groundwater. Nitrate leaching can be reduced with good irrigation management, including appropriate equipment selection, long-term maintenance practices, and irrigation scheduling. To effectively manage water and nitrogen, irrigation systems and management strategies must be matched to field soil and slope conditions. Irrigation systems include surface or furrow, sprinkler, and drip or trickle irrigation.

Gravity or furrow irrigation is a relatively low-cost irrigation option with much potential for nitrate leaching due to uneven and excessive water application. Average application depths can reach 9 inches if the field layout and system management are not matched properly. Application depths vary along the furrow, typically with longer infiltration time at the upper, rather than the lower, end of the field. Uniformity of application can be improved by adjusting the set time and the furrow stream sized (*Figure 12*) to push water to the end of the furrow in half to three-quarters of the irrigation time. Use of surge irrigation to reduce the advance time and compacting furrows may increase uniformity of water along the furrow. Furrow irrigation of alternate rows with nitrogen applied in the nonirrigated row may improve water and nitrogen use efficiency for some rainfall and soil conditions. Efficient furrow irrigation is more difficult with reduced tillage due to crop residue cover, increased infiltration rates, and unequal infiltration rates for wheel track and non-wheel track furrows. A lower percentage of the rows are impacted by compaction if dual wheels are not used or the planter equipment has more than 12 rows.

Water application is better controlled with center pivot than with furrow irrigation. In an effort to reduce pumping costs, many center pivots are equipped to operate at reduced pressures and with drop tubes to reduce the sprinkler height above the soil surface. Positioning the sprinkler near or in the corn crop canopy will reduce the impact of wind drift and canopy evaporation, but may cause poor water distribution due to plant interception of the water pattern, resulting in zones of excessive and inadequate application

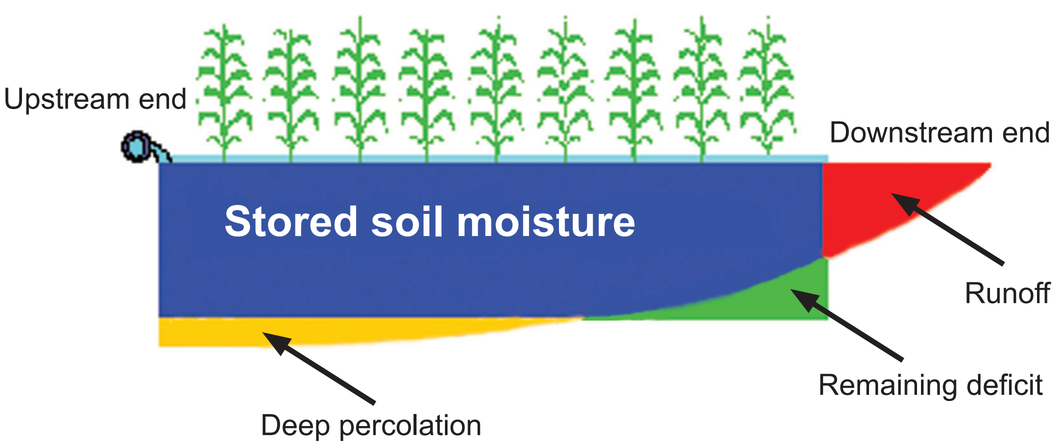


Figure 12. Conceptual drawing of water distribution down an irrigated furrow when the flow rate and irrigation time are selected to minimize deep percolation losses.

(*Figure 13*). If this sprinkler positioning is used for fertigation, nonuniform fertilizer and water application may result in increased potential for nitrate leaching. Uniform in-canopy application requires selection of a sprinkler package appropriate for the field and a maximum sprinkler spacing of 7.5 feet.

Properly managed sprinkler irrigation systems provide an opportunity to apply much of the needed nitrogen by fertigation at times when the potential for rainfall is low and crop uptake of nitrogen is high. Fertigation applications generally range from 20 to 50 lb nitrogen per acre with 0.5 to 1.25 inches of water.

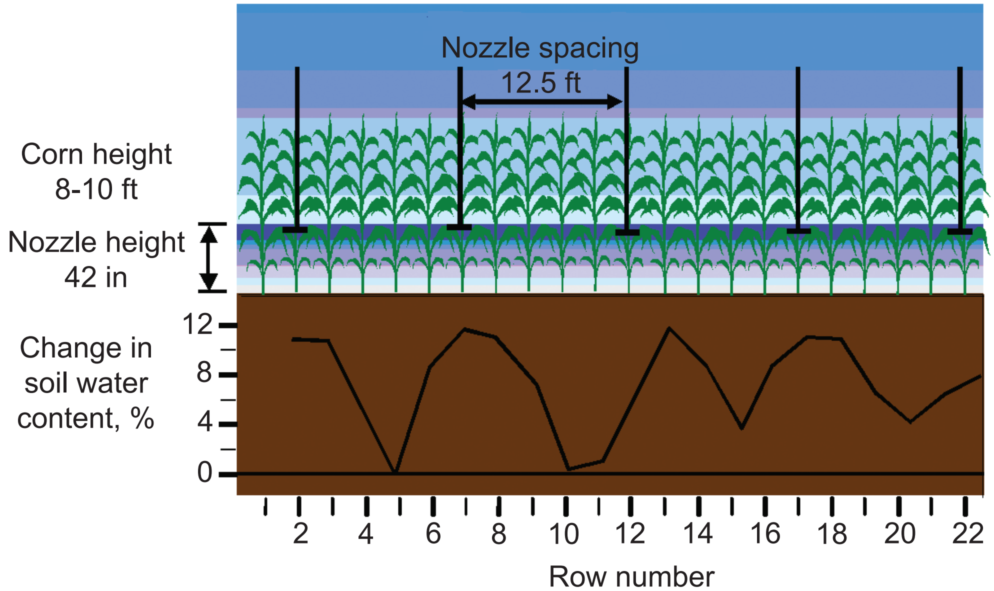


Figure 13. Change in soil water content resulting from a single sprinkler application event using low pressure spray nozzles positioned 3.5 feet above the soil surface and a nozzle spacing of 12.5 feet between nozzles.

Recent developments in drip irrigation have resulted in installation of subsurface drip irrigation systems in fields that were previously furrow irrigated. Drip lines are typically placed 12-16 inches below the soil surface at a 60-inch spacing between drip lines and emitter spacings of 18 inches. Subsurface drip irrigation systems provide the opportunity to place water and nitrogen near roots on a frequency that mirrors plant uptake. Consequently, with proper management, less soil nitrogen and water are available for leaching.

Irrigation management practices for minimizing nitrogen leaching include:

1) knowing the water-holding capacity of all soils in the field;
2) monitoring soil water content to determine how much water has been removed and to evaluate the effectiveness of current irrigation management practices;
3) recording how much water is being delivered to the field;
4) recording precipitation and estimating how much enters the crop root zone;
5) estimating crop water use rates for each crop; and
6) calculating a soil water balance based on stored soil water, crop water use, and water applied via precipitation or irrigation.

Nitrate-N in irrigation water is used as efficiently as fertilizer nitrogen if applied with appropriate rates and during the period of active nitrogen uptake by the crop (*Table 5*). Since the need for irrigation is uncertain when nitrogen application rates are determined, nitrogen management plans should account for the nitrogen in at least 75 percent of the normal irrigation depth applied.

Table 5. Available nitrogen applied in irrigation.

Water nitrate-N (ppm)	Irrigation amount (inches)			
	5	10	15	20
	N applied (lb/ac)			
10	11	23	34	46
20	23	46	68	91
30	34	68	102	137
40	46	91	137	182
50	57	114	171	228

For example, if the normal application depth is about 12 inches of water with a nitrate-N concentration of 20 ppm, the nitrogen application plan should include the nitrogen contained in 9 inches of irrigation water (0.75 x 12 inches = 15 inches) or 41 lb nitrogen per acre. For application depths not included in *Table 5*, managers can determine the nitrogen application per inch of water applied by multiplying the irrigation water nitrate-N concentration by 0.228 lb nitrogen per ppm (9 inches/acre x 20 ppm x 0.228 lb N/ppm = 41 lb N/acre).

Cost-effectiveness of Nitrogen Management Practices

The effectiveness of management practices in reducing nitrogen runoff and leaching loss to water systems has been estimated, considering farm data, expert opinion, and the results of numerous field, laboratory, and computer modeling studies (*Table 6*). The estimates are considered median values for cost and effectiveness, realizing that the actual values will be very different for some situations. The effect of a practice on nitrogen loss is for adoption of a single practice. The benefit of adopting two or more practices will not be fully additive and is more likely to be multiplicative.

The estimated cost of a practice is the expected loss in producer profitability associated with adoption. Alternatively, it can be treated as the payment to producer required to fully compensate for the costs. Actual costs and effectiveness vary with situations. This information (*Table 6*) is intended to complement local expertise in the selection of practices for a given field.

Table 6. The estimated typical cost and effectiveness of practices for reducing nitrogen runoff and leaching losses (adapted from Kansas State University publication MF-2572).

Management practice	Cost/ acre, $/acre	Nitrate leached		Total N in runoff
		Medium to fine soil texture	Coarse soil texture[a]	
Preplant incorporate into the soil prior to the first runoff	7.15	-	-	L[b]
Sub-surface apply N fertilizer	3.50	-	-	L
Eliminate fall application of N fertilizer	0.00	L	M	-
Split apply N fertilizer	6.00	L	M	L
Apply N fertilizer according to in-season N test	6.00	L	M	L
Use a nitrification inhibitor	8.00	L	M	-
Rotate crops	0.00	L	M	M
Maintain >30% residue cover following planting	0.00	-	-	M
Practice no-till farming	0.00	-	-	H
Improve irrigation management	2.00	L	L-H	L
Farm on the contour (without terraces)	6.80	-	-	L
Use terraces	–[c]	-	-	H
Establish buffer strips	–[d]	-	-	M
Sample and test soil	1.00	L	L-M	L
Use sound fertilizer rates	0.00	L-M	M-H	L
Test manure for nutrient value	1.00	L-M	M-H	L-M

[a] The impact of these practices to reduce nitrate leaching is generally greater for sandy than for fine-texture soils.

[b] L, M, H = low, medium, and high effectiveness corresponding approximately to N loss reductions of <20, 20 to 40, and >40%.

[c] One-time installation cost of $40 per acre for tile-inlet terraces and $30 per acre for grass waterway terraces, plus an annual cost of $13.60 per acre.

[d] Establishment cost of $100 per acre of buffer area plus an annual cost equal to the average per acre land rental rate for the acreage within the vegetative buffer strip.

This information is based on the estimates of a team from Kansas State University. Actual costs and benefits will vary with situations and the information needs to be complemented by local expertise in the selection of practices for a given field.

References

Crumpton, W.G. 2001. Using wetlands for water quality improvement in agricultural watersheds; importance of a watershed scale approach. Water Science and Technology 44:559-564.

EPA's 2004 National Emission Inventory – Ammonia Emissions from Animal Husbandry Operations. Draft Report. Jan. 30, 2004.

Goolsby, D.A. and W.A. Battaglin. 1997. Sources and transport of nitrogen in the Mississippi River basin. United States Geological Survey. Available online: *http://co.water.usgs.gov/midconherb/html/st.louis.hypoxia.html.*

Kanwar, R.S., and J.L. Baker. 1993. Tillage and chemical management effects on groundwater quality. p. 455-459. Proc. Agric. Res. to Protect Water Quality, Minneapolis, MN. 21-24 Feb. 1993. Soil and Water Conserv. Soc., Ankeny, IA.

Kanwar, R.S., T.S. Colvin, and D.L. Karlen. 1997. Ridge, moldboard, chisel, and no-till effects on tile water quality beneath two cropping systems. J. Prod. Agric. 10:227-234.

Kasper, T.C., D.B. Jaynes, T.B. Mooreman, and T.B. Parkin. 2002. Reducing nitrate levels in subsurface drain water with organic matter incorporation. Final Report to the American Farm Bureau Foundation for Agriculture, Park Ridge, IL, 23p.

Lawlor, P.A., M.J. Helmers, J.L. Baker, S.W. Melvin, and D.W. Lemke. 2005. Nitrogen application rate effects on corn yield and nitrate-nitrogen concentration and loss in sub-surface drainage. ASAE Paper no. 052025. St. Joseph, MI, ASAE.

Lory, J.A., R.E. Massey, S.M. Zulovich, J.A. Hoehne, A.M. Schmidt, M.S. Shannon, and C.D. Fulhage. 2004. Feasibility and costs of phosphorus application limits on 39 US swine operations. J. Environmental Quality. 33:1114-1123.

Mallarino, A.P., E. Ortiz-Torres, and K. Pecinovsky. 2005. Effects of crop rotation and nitrogen fertilization on crop production. In annual progress reports-2005. Northeast research and demonstration farm. ISRF04-13. Iowa State Univ., Ames, IA. *http://www.ag.iastate.edu/farms/reports.html.*

Midwest Plan Service. Livestock Waste Facilities Handbook. 3rd ed. 1993. MWPS-18. Ames, IA

Nafziger, E.D., J.E. Sawyer, and R.G. Hoeft. 2004. Formulating N recommendations for corn in the Corn Belt using recent data. p. 5-11. In Proceedings of the North Central Extension-Industry Soil Fertility Conference, Des Moines, IA. p. 17-18. Nov., 2004. Vol. 20. Potash and Phosphate Inst., Brookings, SD.

Nolan, B.T., B.C. Ruddy, K.J. Hitt, and D.R. Helsel. 1998. A national look at nitrate contamination of ground water. United States Geological Survey. *http://water.usgs.gov/nawqa/nutrients/pubs/wcp_v39_no12/*.

Randall, G.W., J.L. Anderson, G.L. Malzer, and B.W. Anderson. 1993a. Impact of nitrogen and tillage management practices on corn production and potential nitrate contamination of groundwater in southwest Minnesota. p. 172-175. In: Proc. Agric. Res. to Protect Water Quality, Minneapolis, MN. 21-24 Feb. 1993. Vol. I. Soil and Water Conserv. Soc., Ankeny, IA.

Sawyer, J.E., and E.D. Nafziger. 2005. Regional approach to making nitrogen rate decisions for corn. p. 16-24. In Proc. Thirty-Fifth North Central Extension-Industry Soil Fertility Conf., Des Moines, IA. 16-17 Nov. 2005. Vol. 21. Potash & Phosphate Inst., Brookings, SD.

Schrader, L.E. 1984. Functions and transformations of nitrogen in higher plants. In: Nitrogen in Crop Production. R.D. Hauck (ed.). American Society of Agronomy/Crop Science Society of America/Soil Science Society of America. Madison, WI.

Spalding R. F. and M.E. Exner. 1993. Occurrence of nitrate in groundwater - A review. J. Environmental Quality 22(3): 392-402.

Tisdale, S.L., W.L. Nelson, J.D. Beaton, and J.L. Havlin. 1993. Soil fertility and fertilizers. Macmillan Publishing. New York, New York.

USDA-ERS. 2006. U.S. fertilizer use and price. *http://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx.*

USGS. 2005. Moving from monitoring to prediction: National assessment of nitrate in ground water. Fact Sheet prepared by B.T. Nolan, Washington, D.C. p. 3.

Applying knowledge to improve water quality

Heartland

Regional Water Coordination Initiative

A Partnership of USDA CSREES
& Land Grant Colleges and Universities