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Foreword

The Upper Mississippi River Sub-basin Hypoxia Nutrient Committee (UMRSHNC) is very pleased to have been able to sponsor the Gulf Hypoxia and Local Water Quality Concerns Workshop. We thank the workshop steering committee and each of the speakers and panelists who through their efforts and expertise contributed so much to the success of the workshop. The committee appreciates the financial contributions of our co-sponsors: the College of Agriculture and Life Sciences, Iowa State University; the US Environmental Protection Agency; and the USDA Agricultural Research Service. We especially thank Dr. James Baker, Professor Emeritus of the Department of Agricultural and Biosystems Engineering at Iowa State University, for bringing together such a distinguished group of researchers to aid in identifying the most effective ways to reduce nutrient losses from agricultural land in the Corn Belt.

The Upper Mississippi River Sub-basin Hypoxia Nutrient Committee includes the Illinois Department of Agriculture, the Iowa Department of Agriculture and Land Stewardship, the Minnesota Pollution Control Agency, the Missouri Department of Natural Resources, and the Wisconsin Department of Natural Resources. Each of these agencies is represented on the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force and the Coordinating Committee of the Task Force.

The workshop is a key component of a reassessment of the science underlying the Action Plan for Reducing, Mitigating, and Controlling Hypoxia in the Northern Gulf of Mexico that was adopted by the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force in October, 2000. The activities of the Upper Mississippi River Sub-basin Hypoxia Nutrient Committee are intended to achieve a near-term goal of a technically sound and economically viable nutrient reduction strategy for the Upper Mississippi River Sub-basin and a long-term goal of reducing nutrient loadings to streams and lakes within the five states and to the Northern Gulf of Mexico which will, in turn, address the coastal, within Basin, and quality of life goals of the Action Plan.

We hope that the results of this workshop have made a significant contribution in describing the physical and cost effectiveness of the various management practices currently available to agriculture to reduce nutrient losses. However, even with the best set of tools, we face an extremely challenging task in getting the right practices on the ground in the right places.

Nutrient impairment of surface and ground waters in the Corn Belt is largely due to a complex set of factors involving landscape and land use changes (which affect ground cover, need for additional nutrient inputs, and hydrology). The current Corn Belt landscape, now dominated by annual row crops and local concentrations of intensive livestock production systems, will require improved management of fertilizer inputs and manure utilization practices to minimize nutrient losses from those systems. Changes in off-site practices, and possibly in some cropping systems, will likely also be needed to reach water quality goals. The potential and limitations of improving both in-field and off-site management practices and systems need to be assessed in order to efficiently plan for future actions. Improvements in current management systems do need to be made, and new, innovative technologies designed and tested. Because of the economic and social consequences of returning lands to their prior condition, society will need to decide how far to go in promoting land use changes (e.g., growing less row crops and/or having longer rotations including sod-based crops) and landscape modifications (e.g., creating more wetlands and buffer strips, and possibly redesigning drainage systems).

There are about 100 million acres of cultivated cropland in the Corn Belt states and with limited state and federal resources for technical assistance and cost sharing, accurate targeting will be critical to achieving water quality improvements. Because phosphorus is typically the limiting nutrient in freshwater systems and nitrogen is the primary limit on algal growth in the Gulf, state and local agencies face a difficult choice in designing programs to meet multiple, if not conflicting, goals. Accurate targeting to achieve reductions in agricultural nonpoint sources is further complicated because potential pollutants from agriculture have different chemistries and, consequently, different pathways to water bodies. For example, nitrate is a soluble, non-reactive chemical and is readily leached through soils, while phosphorus is slightly soluble and reactive in soils and the highest concentrations are in the upper soil layers.

In most of the Corn Belt, nitrate concentrations in streams and reservoirs are much higher in those areas underlain by flat, black, tile-drained soils and sandy soils. Phosphorus loads attributable to agricultural nonpoint sources are highest in areas with high surface runoff or erosion rates. In addition, different management practices are often necessary to reduce nitrate and phosphorus movement to surface water. In some instances, practices to reduce nitrate leaching and movement to surface waters may increase losses of phosphorus.

The costs, whether in incentive payments for changes to management practices or for constructed management practices, are relatively constant for an acre of land treated with a particular practice. However, loadings of sediment and nutrients vary greatly across the Corn Belt and within individual states, within counties or small watersheds, and even from differing areas of fields. The most cost-effective strategies to achieve pollutant reduction will require targeting of the delivery and implementation of improved management practices.

Targeting must include the right practice in the right area. For example, educational and incentive programs to encourage changes in nitrogen management practices will be most fruitful if they are targeted to tile-drained areas and erosion control practices are likely to be most efficient if they are targeted to fields contributing high sediment loads. Government programs based primarily, or sometimes solely, on a first-come-first-served approach or on a dominant goal of spending the allocated funds are relatively easy to implement, but will not get the job done.

Variable payment rates in financial incentive programs may also play a part in an effective strategy for pollutant reduction. A higher cost-share rate for installation of erosion control practices on a sloping field immediately adjacent to a stream, for example, may be the most cost-effective way to reduce losses of sediment and particulate phosphorus.

Choices will need to be made among the competing demands for reductions, changes, and improvements, and we must design programs to most cost-effectively address the agreed-upon goals. While many of the management practices discussed in this workshop have secondary benefits in reducing sediment, sequestering carbon, and providing wildlife habitat, not all of these environmental benefits can be primary goals along with nutrient reduction to water resources.

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Acknowledgements

Funding for this water quality workshop was provided by College of Agriculture and Life Sciences, Iowa State University; the US Environmental Protection Agency; and the USDA Agricultural Research Service.

The success of the workshop was dependent on the planning of the steering committee, the dedication to writing of the authors and co-authors, and the participation of the panelists and audience during the meeting. In all, approximately 200 persons attended the three-day meeting. Thanks are also due the anonymous reviewers who provided written feedback to the authors for improving the manuscripts that appear in this volume.

Special thanks are due Brent Pringnitz, who was the on-site conference coordinator. His close involvement from the start and attention to detail contributed significantly to a well-run meeting. His experience and efficiency resulted in a quality draft proceedings being available to participants at the meeting, even though he received electronic copies of some papers as late as two working days before the meeting start.

Thanks are also due Glenn Laing for serving as the ASABE editor and facilitating all the communications with reviewers and authors. The results of his diligence are evident in this volume.

Preface: The Process Behind This Book

The credibility and value of this work stems from the process used to produce and review the papers found in this volume. First, a steering committee was formed (its members are listed below) and fourteen topic areas were chosen for a workshop to be held on the Iowa State University campus in Ames, Iowa, on September 26-28, 2005. Next, potential primary authors were identified (some of these were on the steering committee) to participate in the workshop and draft papers to be presented and critiqued at the workshop. The papers were to offer consensus and expert opinion on the state of the science of the topics, not extensive literature reviews. Authors were given the option of selecting additional scientists as co-authors. The steering committee suggested some possible co-authors and approved all co-authors finally selected.

Then, the steering committee selected additional scientists to serve on panels to help with the oral discussion after each of the fourteen paper presentations at the workshop. For each topic there were at least five people in the author/co-author/panelist roles. Most of the panelists were able to review the paper for their panel in advance of the workshop.

At the workshop, the presenters had no more than 25 minutes of a 60-minute session to make their presentations. In the remaining time, there was first a discussion with the panelists, and then written questions were taken from the audience. These were addressed by the assembled authors and panelists, with some selection by the moderators as to which questions to address if time was limited. All the written questions were later given to the authors. At the end of the workshop, panelists were given the option of becoming co-authors if agreeable to all.

After the meeting, the authors were given time to revise their papers based on what they had heard at the meeting, further input from co-authors and panelists, and the written questions. The revised papers were then sent to an ASABE editor, Glenn Laing, who sent them on for external anonymous review. There were at least three reviewers per paper; at least two of which were not actively involved in the workshop. Reviews were returned to the authors, and with the help of the ASABE editor and to his satisfaction, each paper was revised to correct any errors, fix any omissions, and generally improve the written paper.

James L. Baker
Chair, Workshop Steering Committee

Steering Committee

James Baker, Iowa State University (retired)
Larry Bundy, University of Wisconsin
George Czapar, University of Illinois
Mark David, University of Illinois
Tom Hunt, University of Wisconsin
Dan Jaynes, National Soil Tilth Lab, ARS-USDA
Newell Kitchen, ARS USDA, University of Missouri
Eileen Kladivko, Purdue University
Dean Lemke, Iowa Department of Agriculture and Land Stewardship
Dennis McKenna, Illinois Department of Agriculture
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Pp. 1-17 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop*. St. Joseph, Michigan: ASABE.

Understanding Nutrient Fate and Transport

1

Including the Importance of Hydrology in Determining Field Losses, and Potential Implications for Management Systems to Reduce Those Losses

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Losses of the major nutrients nitrogen (N) and phosphorus (P) from agricultural fields to water resources cause water quality concerns relative to the health of both humans and aquatic systems, and impair water resource uses. In addition to concerns for hypoxia in the Gulf of Mexico, work is currently underway by the states, with guidance from the U.S. EPA, to develop nutrient water quality criteria to be protective of local flowing (streams and rivers) and standing (lakes and reservoirs) waters.

At this time, it is not clear if or how production economics and carbon sequestration/soil quality-sustainability will be taken into account in the development of the criteria. Current water quality concerns and use impairments give rise to the expectation that the criteria, when developed and implemented, will lead to additional water bodies being listed as impaired. Thus, there is an immediate and continuing need to assess and improve tools to reduce nutrient losses from agricultural lands in the Corn Belt.

Understanding nutrient fate and transport is critical in designing and implementing the correct practices/systems to effectively reduce nutrient losses. However, it is probably equally as important, if not more so, to not promote the wrong practices (practices that may have no effect or even increase losses, or may increase loss of another nutrient). Knowledge of the potential, limitations, and factors that affect the efficiency of individual practices is the first necessary piece of design. Being able to combine that knowledge, including that of any interactions between practices, with site-specific conditions is the second necessary piece to develop the overall system of in-field and off-site practices. Part of the discussion in choosing and implementing improved practices/systems is predicting and measuring the water quality changes needed to meet the outcomes desired (assuming we know what we want and how much nutrient reduction is needed to get there, which is the possible topic of a future workshop). The evaluation or assessment of practices/systems can range from being "directionally correct" to a strictly quantified reduction that is needed in a "performance-based" approach.

Foremost in a "performance-based" approach is the question of whether there are any practices/systems capable of reaching the stated criteria (for drainage from a landscape driven by nature's highly variable weather, it is not a given that there always will be). Given that there are options, this provides producers the flexibility to choose practices/systems that fit their needs and that, hopefully, will be efficient in reducing water pollution. However, although the performance-based approach worked well for point-source pollution, and is appealing because performance (i.e., meeting water quality criteria) is what is sought, there are some issues/concerns that need to be overcome. The main four are: (1) the number of choices of practices/systems available to producers can be limited by economic constraints; (2) being able to accurately predict the nutrient reductions expected for practices/systems under a constant or standard set of conditions is difficult; (3) the highly variable nature of weather (in time and space), and the highly variable spatial nature of soils and their properties that affect nutrient fate and transport, makes predictions for realistic field/watershed conditions even more difficult; and (4) the high cost and effort needed to accurately monitor what the outcomes are, especially for large numbers of fields or watersheds, is prohibitive. To overcome the last three issues, nutrient criteria that allow some exceedence of a standard would be needed (based on frequency and duration of exceedence, as recommended by the National Research Council), as well as an acceptable mathematical modeling approach to quantify outcomes on a temporal basis. Some monitoring would still be needed to confirm water quality improvements.

In the following sections, transport mechanisms, hydrology, as well as nutrient availability and concentrations will be discussed relative to potential nutrient losses. Detailed discussions of infiltration and the mixing of rainfall with surface soil that, along with soil adsorption, determine nutrient concentrations in surface runoff water are given to provide understanding and illustrate how complicated the processes are. Two general landscapes common to the Corn Belt (nearly flat, tile-drained areas, and rolling hills, with well-developed surface drainage) will then be briefly discussed relative to the resultant impacts on the need for and choice of management practices/systems to reduce losses.

The purpose of this chapter is therefore to "set the stage" for the 13 chapters that follow on the potential and limitations of specific management practices or "tools" to reduce agricultural nutrient losses to water resources in the Corn Belt states. As such, the intent of this chapter is to present background information in order to promote understanding of how and why some practices might be effective in reducing field losses for some nutrient forms under certain conditions, but possibly not for other nutrient forms and/or under other conditions. This understanding is also important when combining management practices into a management system for reducing field nutrient losses. Thus, this introductory chapter does not broach the subject of any specific management practices.

Transport Mechanisms

The various nutrient forms can be lost from fields dissolved in water, attached to eroded soil/sediment in surface runoff, and when dissolved in leaching water. These three transport mechanisms, or nutrient carriers, are illustrated in figure 1-1. Which

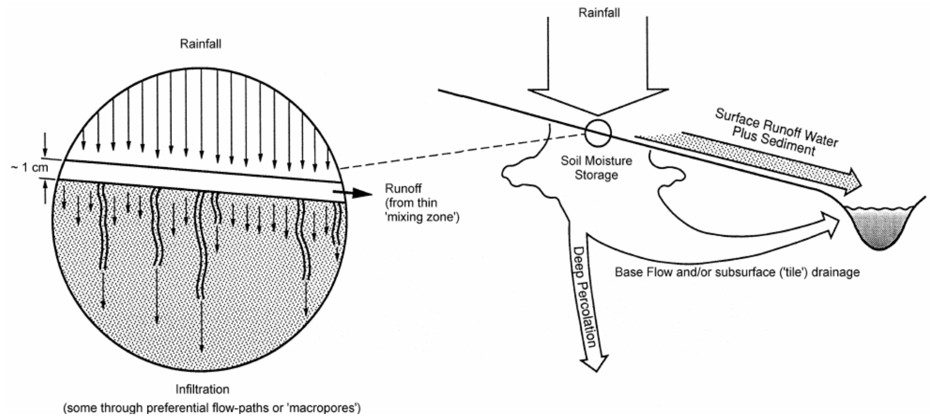

Figure 1-1. Schematic of transport processes and the "thin mixing zone."

carrier is dominant for an individual inorganic nutrient in ionic form is largely determined by the soil adsorption properties of that form: weakly to non-adsorbed forms are lost mainly with leaching water, moderately adsorbed forms are lost mainly with surface runoff water, and strongly adsorbed forms are lost mainly with sediment. Concern is focused on the inorganic ions ammonium-nitrogen (NH_4-N, moderately adsorbed), nitrate-nitrogen (NO_3-N, weakly to non-adsorbed) and phosphate-phosphorus (reactive P, strongly adsorbed). For organic forms of nutrients, the combination of generally low solubility and strong soil adsorption results in sediment being the dominant carrier. Total N and total P, including organic and inorganic forms both in solution and associated with sediment (with P, there may need to be an additional delineation, i.e., that of "bioavailable P"), are also a focus of concern.

National guidelines proposed for flowing waters (streams and rivers) for the western Corn Belt ecoregion are 2.18 mg total N L^{-1} and 0.076 mg total P L^{-1}. National guidelines proposed for standing waters (lakes and reservoirs) are 0.78 mg total N L^{-1} and 0.038 mg total P L^{-1} (www.epa.gov/waterscience/criteria/nutrient/ecoregions/index.html).

Nutrient loss from cropland (either in total or for individual forms) is equal to the summation of the products of the masses of the carriers times the nutrient concentrations in the respective carriers, i.e.:

$$\text{Total loss} = \sum_{n=1}^{3} \text{mass}_n \times \text{concentrations}_n$$

where n = 1 through 3, representing surface runoff water, leaching water (including artificial subsurface drainage, commonly called "tile drainage"), and sediment. Thus, a management practice/system that reduces a carrier mass or concentration in that carrier, without increasing the other factor, reduces loss with that carrier. However, the overall impact will be determined by the effects on and summation for all three carriers. For certain water quality concerns, consideration of individual nutrient forms is necessary (e.g., nitrate-nitrogen for drinking water). When a single management practice is not sufficient to provide the desired level of control, a system of practices will be needed. To devise a single practice or a system of management practices that is efficient in reducing nutrient transport to water resources, knowledge of the major mechanism of transport is needed. This requires information on the nutrient properties, the source(s)/availability of the nutrient, and the soil and climatic conditions that exist. A system may include a combination of in-field and off-site practices.

Hydrology

The "driver" for hydrology is the timing, intensity, and amount of precipitation. These parameters in conjunction with evapotranspiration determine the amount of "excess" water that drains from a field at any given location and time. While in general the amount of precipitation and resultant excess water increases from the western to the eastern Corn Belt, the actual values are highly variable in space and time. In addition to variation on a spatially large scale, there is considerable small-scale variation, especially when short time intervals are considered that include droughts and extreme events. Therefore, the total volume of drainage from any particular field in any particular year is highly variable, which makes controlling field nutrient losses to a certain predetermined level very difficult.

On the field scale, probably the most important hydrologic factor affecting nutrient losses from agricultural lands is whether tile drainage has been installed, which significantly affects the relative volumes of surface runoff and subsurface drainage. Tile drainage has both positive and negative effects on water quality, as alluded to below and discussed later (see reviews by Gilliam et al., 1999; Baker et al., 2004; and Baker et al., 2006).

On a smaller scale of a few square feet, probably the most important hydrologic factor is the soil water infiltration rate, which is highly variable, both temporally and spatially (Baker, 1997). Infiltration refers to the entry of water into the soil profile from the surface. Two forces drive water to infiltrate: one is gravity, and the other is the "suction" of water by dry soil. The degree of "suction" depends on the moisture gradient between wet soil at the surface and drier soil deeper in the profile. Thus, antecedent moisture conditions are important. During the early stages of infiltration (at the beginning of a rainfall event), the suction forces add to (and predominate over) the force of gravity, and the infiltration rate is at its highest. As water infiltration wets the soil, the wetting front advances down into the soil profile and reduces the moisture gradient and suction forces, and the infiltration rate decreases with time. When the rainfall rate is less than the initial infiltration rate but greater than the final gravity-dominated rate, a point will eventually be reached where the water cannot be taken up by the soil profile as fast as it is being added. At this time, the surface soil becomes saturated, and ponding (and runoff from sloping soils) begins.

It is the infiltration rate in conjunction with the rainfall rate that determines the volume and timing of surface runoff. Both rates can change by the minute, making measurement and prediction so difficult. Subtraction of runoff volume from precipitation amount gives the volume of water that enters and is temporally stored in the soil root zone. Eventually, this water is removed from the root zone through either evapotranspiration or by movement via percolation to groundwater or back to surface water resources through natural or artificial subsurface drainage. In general, the higher the infiltration rate and the greater the infiltration, the lower the field losses will be for all nutrient forms, with the exception of NO_3-N because of its leaching potential.

Infiltration rate can also play a role in determining the concentrations as well as the masses of carriers. As shown in figure 1-1, there is a thin "mixing zone" at the soil surface that interacts with and releases sediment and nutrients to rain and runoff water. The volume of rainfall that infiltrates before runoff begins, as well as the soil adsorption properties of the nutrient form of interest, affect the amount of a particular nutrient form remaining in the "mixing zone" (illustrated in fig. 1-1 as having a thickness of about 1 cm or 0.5 in.) potentially available to be lost. During a rainfall event, the amount of nutrient remaining in this mixing zone decreases with movement of water over and/or down through this zone. Obviously, the higher the rate of infiltration, the longer it is before runoff begins, and the lower the nutrient concentrations in runoff water. This is also true for nutrient concentrations in sediment derived from soil in the mixing zone when considering soluble nutrient forms that have some affinity for soil.

A second important small-scale hydrologic factor that affects nutrient leaching loss is the route of infiltration. Infiltrating water can move through the whole soil matrix, or some of it can find "macropores" or preferential flow paths through which to move quickly deeper, thereby "bypassing" much of the soil. This is also illustrated in figure 1-1. If the nutrient of concern is within soil aggregates, water flowing through macropores can bypass the nutrient, and leaching will be reduced. However, if the nutrient is on the soil surface and dissolves in infiltrating water that is moving though macropores, then leaching will be greater, quicker, and deeper than otherwise expected.

Outside of intrinsic soil factors such as texture and slope, antecedent soil moisture content is one of the important (noted earlier) and variable factors affecting infiltration. Other important factors are soil compaction, soil structure, and surface residue cover. Besides affecting infiltration rate, surface residue cover (and soil "roughness") creates ponding conditions which extend the opportunity time for infiltration (and therefore the volume of infiltration). Soil structure as affected by soil compaction will determine the porosity and number, stability, and continuity of preferential flow paths that can increase infiltration rates. All of these non-intrinsic factors can be affected to some degree by management practices such as artificial drainage, cropping, implement traffic/compaction, residue management, and tillage.

Of these, the first two, artificial drainage and cropping, go hand-in-hand and have the greatest effect on hydrology, where installation of artificial subsurface drainage has in turn allowed intensive annual row-cropping. Over the last 120 years in Illinois and Iowa, wetlands have been drained and the prairie-wetland landscape, where it existed, has been transformed from perennial vegetation to primarily annual, shallow-rooted, corn and soybean row-crops. Figure 1-2 shows the reduction in wetland area in Iowa over roughly the last 140 years, and figure 1-3 shows the trend in drain tile production in Illinois during the period of intense drainage activity. Other states in the Corn Belt had similar periods of intense drainage activity either before (more eastern states) or after (states west of Illinois) Illinois. Improvements in drainage continue every year, although there are no data available to document this activity. Cropping systems have

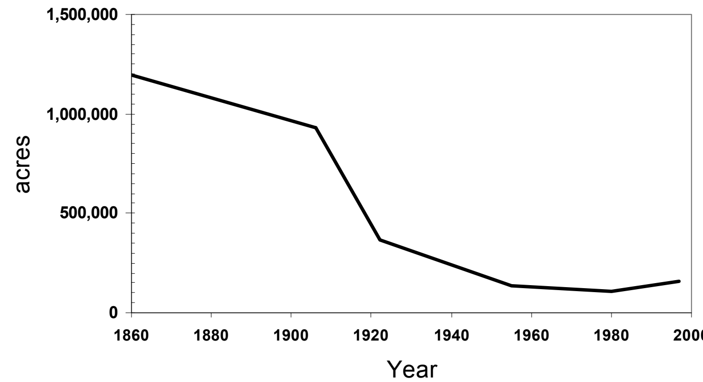

Figure 1-2. Trends in total wetland area in Iowa with time.

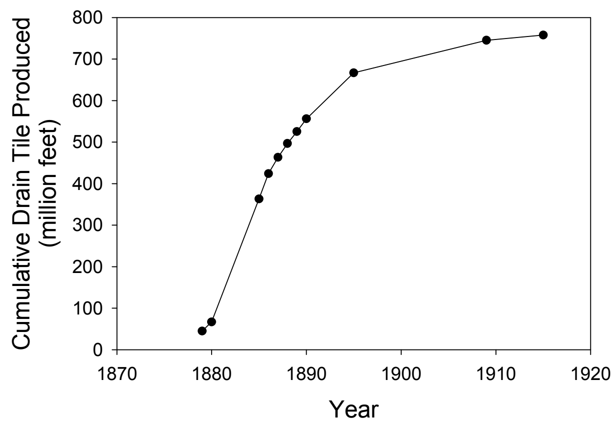

Figure 1-3. Cumulative drain tile production for the state of Illinois. Most of Illinois was tiled during the 1880s and 1890s, with record keeping stopped in 1915 due to lack of production.

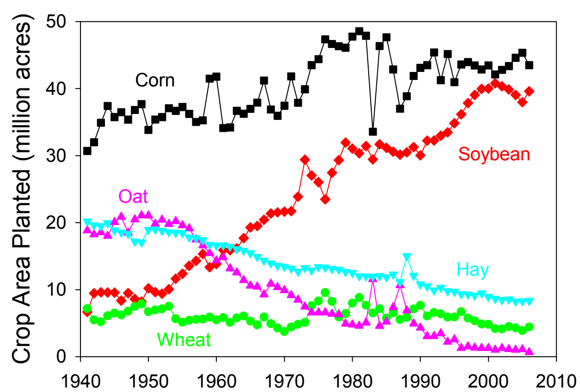

Figure 1-4. Annual area planted in major crops for the Corn Belt through 2006. Sum of Minnesota, Iowa, Wisconsin, Illinois, Indiana, and Ohio.

also changed greatly across the Corn Belt during the last 70 years (figure 1-4). Rotations have been simplified to corn and soybean in rotation, with few oats now planted and declining hayfields. It is important to realize from these trends the magnitude of the scale of change in this landscape that now produces very high productivity for the dominant two crops, making a return to predevelopment conditions for water quality purposes infeasible.

Analysis of stream flow data over the second half of the 20th century (Schilling, 2005) indicated that baseflow and baseflow percentage have generally increased in that time frame and are "significantly related to increasing row crop intensity." The subsurface drainage that has been installed reduces the moisture contents of the surface soils, increasing infiltration rates, and in turn, reducing surface runoff volumes but increasing subsurface flows. In addition, this finding is directly in line with the fact that there is less evapotranspiration and more subsurface drainage with row-crops than there would be with grasses. In a six-year study in Minnesota (Randall et al., 1997), it was shown for wet years that drainage from row-crops exceeded that from perennial crops by 1.1 to 5.3 times. This is especially evident and important in the April-May-June period when rainfall amounts usually far exceed the water needs of shallow-rooted corn and soybean crops just getting established. This is also a time before the major uptake of nutrients, N and P, by the row-crops. Data show that a major portion of annual subsurface drainage takes place in that April-May-June period. For example, in a 15-year study in north-central Iowa (Helmers et al., 2005), over 70% of the tile flow occurred in those three months. In another 15-year study in southern Minnesota, Randall (2004) found that 68 to 71% of the flow and 71 to 73% of the NO_3-N leaching losses occurred in those three months. Kladivko et al. (2004) in a 15-year study in Indiana showed that most of the flow and NO_3-N leaching losses occurred during the fallow season. Jin and Sands (2003), in a hydrologic analysis of subsurface drainage for south-central Minnesota, showed on average for an 85-year climatic period that 74% of infiltration in the March to June period was removed by subsurface drainage. Royer et al. (2006) showed for Illinois watersheds that most export from agricultural watersheds was during high discharge events that occurred from mid-January through June.

Farming the tile-drained landscape presents an environmental challenge with respect to NO_3-N leaching. However, that challenge would seem to be less than the multiple challenges for landscapes not needing artificial drainage with a more sloping/rolling topography, which results in better surface drainage and thus more surface runoff. Use of subsurface drainage under most designs generally reduces losses of nutrients and sediment in surface runoff because of not only reduced surface runoff volumes but also reduced concentrations in the surface runoff water (Baker et al., 2006). To meet the tile drainage, NO_3-N challenge and preserve the use of drained and highly productive lands, special attention will need to be paid to in-field soil, cropping, and nutrient (N) management to minimize NO_3-N leaching, as well as possible use of improved water management practices and wetlands in the overall system design.

Nutrient Forms and Availability

Three chemical properties largely determine the fate and possible off-site transport of individual forms of nutrients with water: resistance to transformation, solubility, and soil adsorption. Solubility and adsorption are usually related, with adsorption generally increasing with decreasing solubility.

Table 1-1 provides a set of numbers for the important nutrient forms for N and P relating their concentrations in the soil and water of a field (at or near equilibrium) to expected concentrations in the three carriers (surface runoff water, sediment, and subsurface drainage). Although these numbers in reality are highly variable, both temporally and spatially, for simplicity of comparison, a single set of numbers is given to represent the annual averages for the row-crop planted (corn rotated with soybeans) in much of the Corn Belt. The numbers were generated from experience with many field and watershed monitoring studies (e.g., Johnson and Baker, 1982, 1984; Baker, 1980, 1987; Baker et al., 2003; Baker et al., 2004). As shown in table 1-1, the concentration and therefore the amount of NO_3-N in soil water generally predominates over that of ammonium-nitrogen (NH_4-N). With respect to field nutrient losses and water quality, the question is how the concentrations in surface runoff and subsurface drainage from a rainfall event relate to those in soil water. Dilution and incomplete mixing of surface soil water with rain causes the lower NH_4-N and NO_3-N concentrations in surface runoff compared to soil water. Soil adsorption/desorption also has an effect. The stronger soil adsorption of NH_4-N compared to NO_3-N reduces the downward movement of NH_4-N from the soil surface mixing zone during the addition of water, maintaining a higher relative concentration in surface runoff. Therefore, while the absolute NH_4-N concentration in surface runoff is lower than that for NO_3-N, the reduction relative to that in soil water is less (50% in the table 1-1 example, as opposed to 92% for NO_3-N).

Table 1-1. Example concentrations of the nutrient forms in soil or soil water, and in surface runoff, subsurface drainage, and sediment from a corn-soybean rotation in the Corn Belt.

Nitrogen (N)			
Soluble	Soil[a] Water (mg L^{-1})	Surface Runoff (mg L^{-1})	Subsurface Drainage (mg L^{-1})
NH_4-N	1.0	0.5	0.1
NO_3-N	50.0	4.0	15.0
Solid/adsorbed	Soil[a] (ppm)	Sediment (ppm)	
NH_4-N	15	20	
NO_3-N	0	0	
Organic N	1500	2000	
Phosphorus (P)			
Soluble	Soil[a] Water (mg L^{-1})	Surface Runoff (mg L^{-1})	Subsurface Drainage (mg L^{-1})
Reactive P	0.6	0.2	0.050
Total P	0.9	0.3	0.075
Solid/adsorbed	Soil[a] (ppm)	Sediment (ppm)	
Available P	30	40	
Total P	600	800	

[a] Top 12 inches of soil, 3% organic matter.

Dilution and incomplete mixing of rain water with soil water (as well as N crop uptake) also reduce NH_4-N and NO_3-N concentrations in water that is in excess of what the soil can retain and is eventually released from the soil profile as subsurface drainage. But in the case of subsurface drainage, NH_4-N adsorption to subsoils low in NH_4-N causes the concentrations to be much lower in subsurface drainage with respect to both NH_4-N concentrations in surface soil water and NO_3-N in subsurface drainage.

As also shown in table 1-1, the concentration of organic N dominates over that of NH_4-N in the solid soil itself, and with no adsorption/affinity for the soil, NO_3-N concentration is shown as 0. The ratios of NH_4-N and organic N concentrations for sediment compared to their respective values for in-place soil are greater than unity, which is due to the selective erosion process where more chemically active, smaller, and less dense (with greater organic matter content) soil particles are preferentially transported. Some organic N can dissolve in surface runoff water, but concentrations are usually less than 2 mg L^{-1}.

As shown in table 1-1 for P in soil water, surface runoff, and subsurface drainage, reactive P (dissolved inorganic or molybdenum-reactive P, sometimes termed PO_4-P or ortho-P) generally makes up more than 60% of the total soluble P. The difference between total soluble P and reactive P is soluble organic P. In the soil, total (organic plus inorganic) P predominates over what is classified as "plant-available" P, as determined by one of several soil P tests (in this case a Bray-1 or Mehlich-3 extractant). As with NH_4-N, reactive P is held somewhat by the surface soil, so runoff concentrations may only be reduced three-fold over that in soil water, but concentrations are much lower in subsurface drainage because of adsorption/precipitation of reactive P in generally P-deficient subsoils. As with N, P concentrations in sediment are greater than in the in-place soil because of the selective erosion process. Given the very high adsorption and low solubility for total P, and realizing that the ratio of the mass of surface runoff water to sediment can be as small as 100 to 1 for rainfall-runoff events, P loss for row-cropped fields is often dominated by that lost with sediment.

Nutrient Concentration-Time Relationships and Watershed Losses

Monitoring activities for 1999, 2000, and 2001 were performed on the Upper Maquoketa River and three intrabasin sites in northeast Iowa, as shown in figure 1-5 (Baker et al., 2003). At the four sites, measurements of flow, N, P, chemical oxygen demand (COD), and suspended sediment concentrations were performed. The site for flow measurement/sampling for the whole basin is just above Backbone State Park, with a drainage area of 39,260 acres (in 1998, 40% corn, 27% soybeans, 11% oats-hay-alfalfa, 10% pasture, and 9% forest). The three intrabasin sites range from 570 acres (designated site 2: 82% corn, 12% soybeans, and 5% pasture) to 8030 acres (designated site 3: 57% corn, 26% soybeans, 13% oats-hay-alfalfa, and 2% pasture) to 4280 acres (designated site 1: 44% corn, 40% soybeans, 10% oats-hay-alfalfa, and 4% pasture). Although exact areas are not known, significant portions of the monitored watersheds have subsurface drainage.

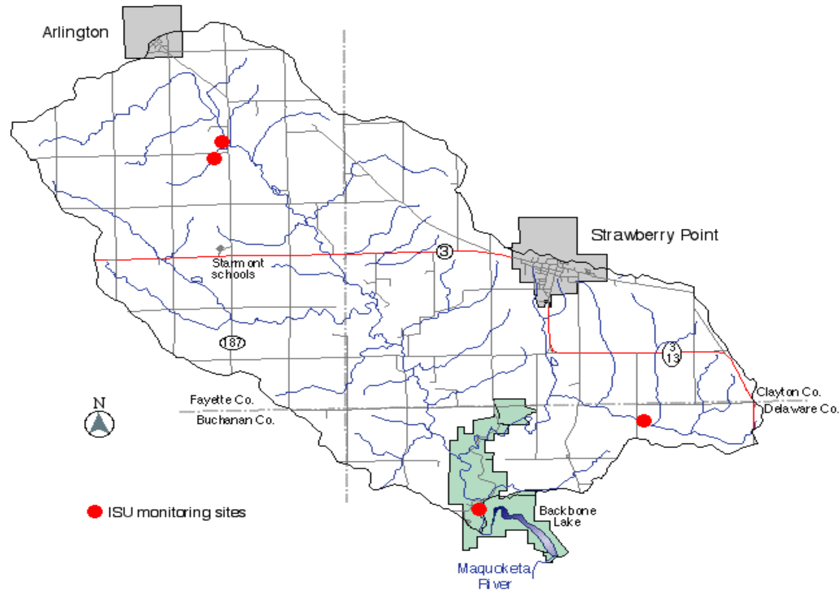

Figure 1-5. Maquoketa watershed monitoring locations.

In general, the data from this watershed study showed that the soluble, non-adsorbed nature of NO_3-N results in N losses being dominated by this form in watersheds where soils and hydrologic conditions result in a significant proportion of streamflow being subsurface drainage. Inorganic P in the form of PO_4-P, because of its tendency to be adsorbed or precipitated from solution, usually has low concentrations in subsurface drainage and higher concentrations in sediment relative to concentrations in surface water. Therefore, these watersheds with lower relief and more subsurface drainage had lower P losses than watersheds with steeper more erosive soils and more surface runoff water.

To illustrate the temporal variability in concentrations in relation to stream flow and hydrology, figure 1-6, as an example, shows flow and suspended sediment concentration data for three rainfall-surface runoff events in a two-week period in May 2001 for the whole-basin monitoring site (site 4). These three events, preceded and separated by flow periods of only subsurface drainage, were caused by rainfall amounts of 1.2 to 1.6 inches and are representative of growing-season events. Figures 1-7, 1-8, and 1-9 show NO_3-N, NH_4-N, and total N (including N associated with sediment) concentrations versus time for the same period. Figures 1-10 and 1-11 show reactive P and total P (including P associated with sediment) concentrations versus time; and figure 1-12 shows COD concentrations versus time, again for the same period. In agreement with the previous discussion and table 1-1, which shows lower NO_3-N and higher NH_4-N concentrations in surface runoff than in subsurface drainage, NO_3-N concentrations decrease during a surface runoff event, while all other concentrations, such as NH_4-N, increase.

Tables 1-2, 1-3, and 1-4 show total annual precipitation, flow, and losses of sediment and nutrients for each of the four sites for 1999, 2000, and 2001, respectively. Nutrient

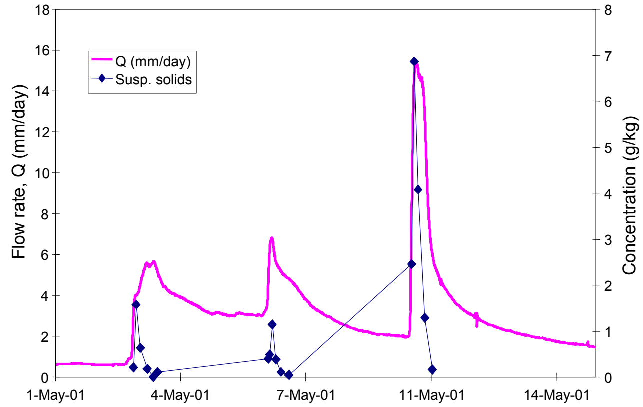

Figure 1-6. Suspended solids at site 4.

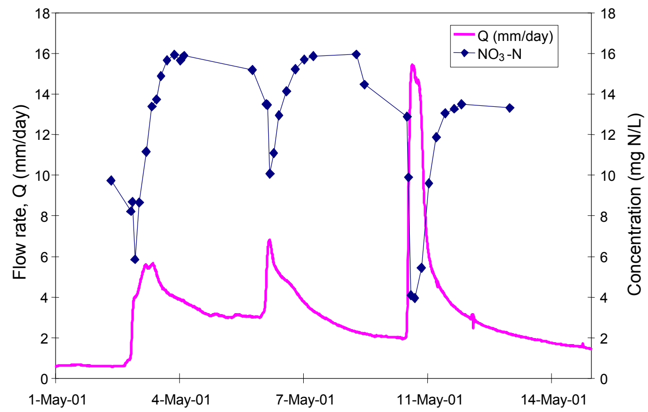

Figure 1-7. Nitrate-nitrogen at site 4.

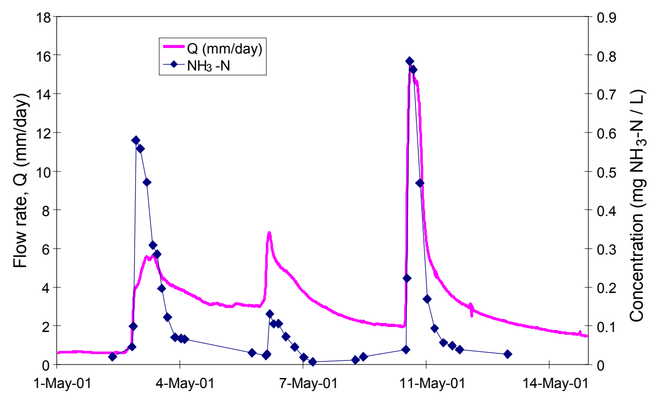

Figure 1-8. Ammonia-nitrogen at site 4.

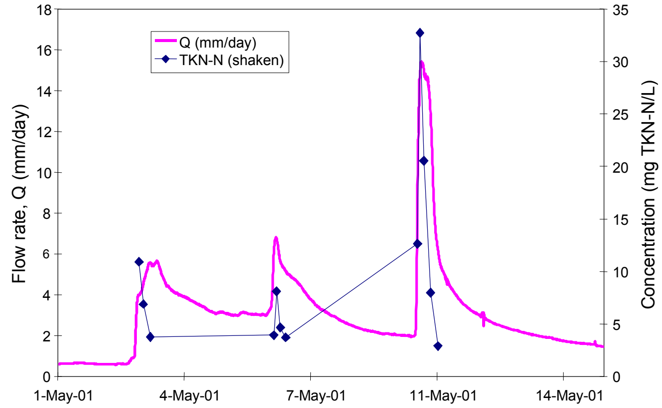

Figure 1-9. Organic nitrogen (shaken) at site 4.

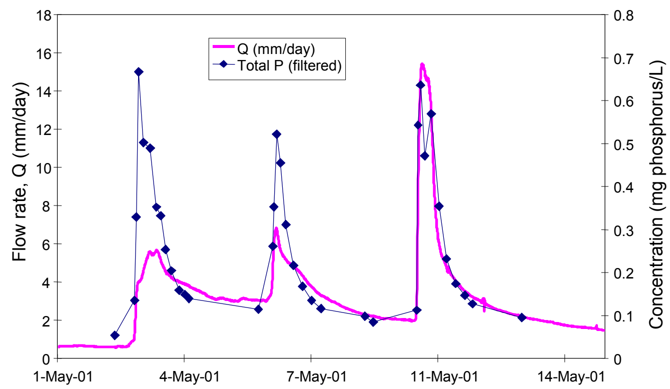

Figure 1-10. Total phosphorus (filtered) at site 4.

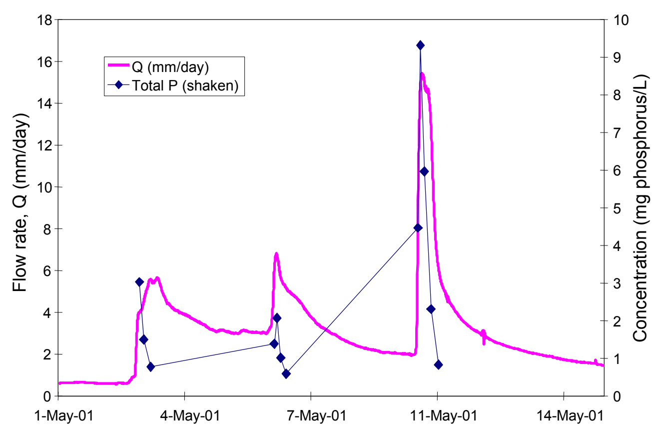

Figure 1-11. Total phosphorus (shaken) at site 4.

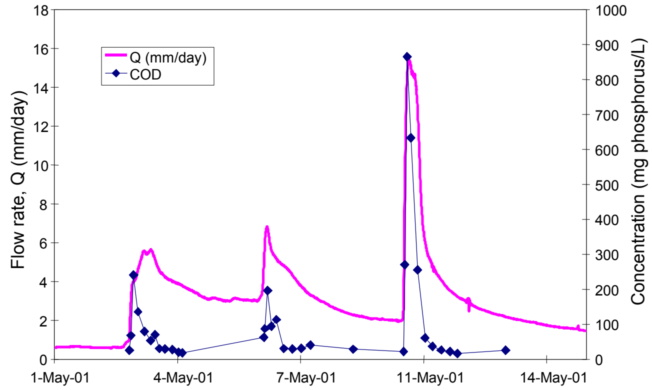

Figure 1-12. COD at site 4.

losses (lb ac^{-1}) are given for both soluble forms and N and P lost with sediment. As shown in the tables, the N losses were dominated by NO_3-N, which ranged from 18 to 58 lb ac^{-1}. Ammonium-nitrogen losses were less that 1 lb ac^{-1}, and soluble organic N loss and N lost with sediment were about the same, in the range of 1 to 1.9 lb ac^{-1}. As a percentage of total N lost in all forms, that lost as NO_3-N was at least 80% of the total for all four sites. As shown in the tables, total soluble P losses were less than 1 lb ac^{-1}, with reactive P in solution making up at least 60% of the total soluble P. The amount of P lost with sediment ranged from about half to more than twice that of total soluble P.

Annual nutrient losses among the three interbasin sites with similar land uses were fairly consistent over the three years of record, as would be expected. Where there were differences, it was usually more related to flow volume differences than concentration differences. Although the interbasin sites had about 15% to 25% more row-crop area than the watershed as a whole, there were not large differences between nutrient losses between the intrabasin sites and the whole watershed. The relative area ratios of 1 to 5.7 to 7.5 (for interbasin sites 2, 3, and 1, respectively) to 68.5 (for the whole watershed, site 4) also did not seem to have an effect on losses. Thus, the attenuation processes that reduce the transport of sediment, and sediment-associated nutrients, as watersheds become larger did not have an effect within the Upper Maquoketa River basin when erosion was low and NO_3-N was the dominant nutrient form lost.

Flow-weighted annual average NO_3-N concentrations (calculated from the data in tables 1-2 through 1-4) ranged from 10.3 to 17.4 mg L^{-1} over the four sites for the three years. The maximum contaminant level (MCL) for NO_3-N in drinking water is 10 mg L^{-1}, which was exceeded much of the time at all four sites. NH_4-N concentrations, which when above 2 mg L^{-1} at normal pH values can be harmful to fishes, never exceeded 1 mg L^{-1} and averaged less than 0.25 mg L^{-1}. Total N concentrations including NO_3-N as well as NH_4-N, soluble organic N, and sediment N were all well above 10 mg L^{-1}, which is more than four times higher than a proposed regional water quality standard of 2.2 mg L^{-1} for flowing water, as discussed earlier. Soluble P concentrations averaged about 0.15 mg L^{-1}, and when sediment P was added, total P concentra-

Table 1-2. Total rainfall, runoff, and losses (lb ac^{-1}) of suspended sediments and nutrients for the Upper Maquoketa watershed in 1999.

	Site 1	Site 2	Site 3	Site 4
Rainfall (inches)	33.6	33.6	33.3	33.3
Runoff (inches)	16.9	10.9	19.6	15.6
NH_4-N	0.8	0.4	0.6	0.2
NO_3-N	40.1	35.0	58.4	36.3
Organic N	2.5	1.3	2.2	1.8
Reactive P	0.5	0.2	0.4	0.3
Total soluble P	0.7	0.3	0.5	0.6
Sediments	362	594	1,373	393
N with sediments	2.1	5.2	11.8	1.5
P with sediments	0.4	0.7	3.1	0.3

Table 1-3. Total rainfall, runoff, and losses (lb ac^{-1}) of suspended sediments and nutrients for the Upper Maquoketa watershed in 2000.

	Site 1	Site 2	Site 3	Site 4
Rainfall (inches)	31.1	31.1	32.7	32.7
Runoff (inches)	11.5	7.9	12.8	12.4
NH_4-N	0.3	0.2	0.6	0.2
NO_3-N	34.5	31.2	48.8	34.1
Organic N	2.5	1.2	2.8	1.8
Reactive P	0.5	0.2	0.3	0.3
Total soluble P	0.6	0.2	0.5	0.4
Sediments	121	133	2,150	789
N with sediments	0.8	0.6	5.3	1.6
P with sediments	0.3	0.2	2.8	0.7

Table 1-4. Total rainfall, runoff, and losses (lb ac^{-1}) of suspended sediments and nutrients for the Upper Maquoketa watershed in 2001.

	Site 1	Site 2	Site 3	Site 4
Rainfall (inches)	35.0	35.0	35.0	35.0
Runoff (inches)	8.9	5.4	8.3	13.1
NH_4-N	0.4	0.1	0.3	0.4
NO_3-N	22.7	17.6	32.9	32.3
Organic N	1.9	0.6	1.2	1.3
Reactive P	0.3	0.1	0.1	0.4
Total soluble P	0.4	0.2	0.2	0.5
Sediments	37	39	237	291
N with sediments	0.1	0.4	1.3	1.9
P with sediments	0.0	0.1	0.3	0.5

tions in stream flow averaged from 0.25 to 0.50 mg L^{-1}. These concentrations are three to seven times higher than a proposed regional water quality standard of 0.076 mg L^{-1}.

Management Practices/Systems

In discussion of nutrient losses, and practices to reduce them, the term "excess nutrients" is often used, with the implication that if there were no excess nutrients, there would be no losses. There are two problems with applying that logic to Corn-Belt row-crop agriculture: (1) under the conditions and assumptions of mass balances being made by Corn Belt states for the corn-soybean rotation, there are no "excess nutrients" (i.e., the amounts of N and P removed from fields on average are more than the inputs, e.g., Libra et al., 2004), and (2) in order that sufficient nutrients are available to the plants to obtain economically optimum crop yields, nutrients must be present in significant amounts during the growing season, and therefore are susceptible to loss with rainfall-runoff and subsurface drainage events that can and do happen at any time.

Corn N needs can be used as an example, where between the grain, stover, and roots, at least 180 lb ac^{-1} of N need to be taken up during the growing season to produce 165 bu ac^{-1}. For high-yielding corn, i.e., 204 bu ac^{-1}, Sawyer et al. (2006) estimate that 275 lb N ac^{-1} would be taken up, with 125 lb ac^{-1} remaining in the stover and roots after harvest. Almost the entire uptake of N is as NO_3-N. Depending on the location and the year, about 18 inches of water is transpired through the corn plant to produce 165 bu ac^{-1}; this is equal to 4 million lb of water per acre. Dividing 180 lb by 4,000,000 lb gives a ratio of NO_3-N to water of 45 mg L^{-1}. Even if only half the N was taken up passively with the mass flow of water into the corn roots at the same NO_3-N concentration as in the soil water, the average concentration in soil water during the growing season would have to be over 22 mg L^{-1} to obtain economically viable yields.

Management practices for the nearly flat, tile-drained areas of Iowa need to be more focused on N because of NO_3-N leaching losses (Baker, 2001, 2003). Management practices for rolling hills, with well-developed surface drainage, need to be more focused on P because of greater potential surface runoff volumes and sediment losses (Baker and Laflen, 1983; Baker, 1987). The Iowa P index addresses this issue (Mallarino et al., 2002).

In summary, the discussion to follow in these workshop proceedings will show that there is potential but also limitations in terms of how much field nutrient losses can be reduced for row-crops with current "tools" in the way of in-field best management practices. These "tools" involve management of fertilizers and manures in the way of rate, timing, and method of application. Tillage and erosion control and improved water management practices are also considered. Off-site practices such as wetlands (for reducing NO_3-N transport) and vegetated filter/buffer strips (for reducing sediment and sediment P transport) will be discussed relative to their considerable potential alone, or when combined with in-field practices to create effective systems, to minimize nutrient losses. Predictions will be made that alternative cropping, in the way of small grains and more and longer sod-based rotations (including cover crops) could have a major impact on reducing nutrient losses; the limitations being mostly economic. The benefits of using field-scale and watershed-scale tools to design more efficient systems of practices based on targeting and site-specific conditions will be presented. And the difficult problem of how to assess the reduction in nutrient losses as a result of implementing new practices/systems on the watershed scale will be addressed. The critical questions of how much nutrient loss reduction is necessary for each nutrient form to meet water quality goals, and who should pay for the implementation of alternative practices when they do not pay for themselves, were not addressed in the workshop. However, they definitely need to be answered and could be the topic for a future workshop.

References

Baker, J. L. 1980. Agricultural areas as nonpoint sources of pollution. In *Environmental Impact of Nonpoint-Source Pollution*, 275-310. M. R. Overcash and J. M. Davidson, eds. Ann Arbor, Mich.: Ann Arbor Science Publishers.

Baker, J. L. 1987. Hydrologic effects of conservation tillage and their importance relative to water quality. In *Effects of Conservation Tillage and Groundwater Quality*, ch. 6. T. J. Logan, J. M. Davidson, J. L. Baker, and M. R. Overcash, eds. Chelsea, Mich.: Lewis Publishers.

Baker, J. L. 1997. The critical nature of infiltration in determining the water quality of agriculture drainage. In *Proc. 3rd USA/CIS Joint Conference on Environmental Hydrology and Hydrogeology-Water: Sustaining a Critical Resource*, 172-181. Minneapolis, Minn.: American Institute of Hydrology.

Baker, J. L. 2001. Limitations of improved nitrogen management to reduce nitrate leaching and increase use efficiency. In *Proc. 2nd International Nitrogen Conference on Science and Policy: Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection. The Scientific World* 1(S2), 10-16.

Baker, J. L. 2003. Potential and limitations of management practices to reduce nutrient losses. In *Proc. Conf. on Total Maximum Daily Load (TMDL) Environmental Regulations II*, 74-79. St. Joseph, Mich.: ASAE.

Baker, J. L. and J. M. Laflen. 1983. Water quality consequences of conservation tillage. *J. Soil Water Conserv*. 38:186-193.

Baker, J. L., M. M. Aqua, P. A. Lawlor, and S. W. Melvin. 2003. Nutrient transport in an agricultural watershed as affected by hydrology. Presented at the 2003 ASAE International Meeting. ASAE Paper No. 030161. St. Joseph, Mich.: ASAE.

Baker, J. L., S. W. Melvin, D. W. Lemke, P. A. Lawlor, W. G. Crumpton, and M. J. Helmers. 2004. Subsurface drainage in Iowa and the water quality benefits and problem. In *Proc. 8th International Drainage Symposium*, 39-50. St. Joseph, Mich.: ASAE.

Baker, J. L., M. J. Helmers, and J. M. Laflen. 2006. Water management practices, rain-fed cropland. In *Environmental Benefits of Conservation on Cropland*, ch. 2. M. Schnef and C. Cox, eds. Ankeny, Iowa: Soil and Water Conservation Society.

Gilliam, J. W., J. L. Baker, and K. R. Reddy. 1999. Water quality effects of drainage in humid regions. In *Drainage Monograph*, ch. 24. Madison, Wisc.: ASA.

Helmers, M. J., P. A. Lawlor, J. L. Baker, S. W. Melvin, and D. W. Lemke. 2005. Temporal subsurface flow patterns from 15 years in north-central Iowa. Presented at the 2005 ASAE International Meeting. ASAE Paper No. 052234. St. Joseph, Mich.: ASAE.

Iowa Ag Statistics. 2003. Iowa Agricultural Statistics. Compiled by Iowa Agricultural Statistics in cooperation with USDA-NASS and Iowa Farm Bureau. Des Moines, Iowa: Iowa Farm Bureau.

Jin, C. X., and G. R. Sands. 2003. The long-term field-scale hydrology of subsurface drainage systems in a cold climate. *Trans. ASAE* 46: 1011-1021.

Johnson, H. P. and J. L. Baker. 1982. Field-to-stream transport of agricultural chemicals and sediment in an Iowa watershed: Part I. Database for model testing (1976-1978). Report No. EPA-600/S3-82-032. Athens, Ga.: U.S. EPA Environmental Research Laboratory.

Johnson, H. P., and J. L. Baker. 1984. Field-to-stream transport of agricultural chemicals and sediment in an Iowa watershed: Part II. Database for model testing (1979-1980). Report No. EPA-600/S3-84-055. Athens, Ga.: U.S. EPA Environmental Research Laboratory.

Kladivko, E. J., J. R. Frankenberger, D. B. Jaynes, D. W. Meek, B. J. Jenkinson, and N. R. Fausey. 2004. Nitrate leaching to subsurface drains as affected by drain spacing and changes in crop production system. *J. Environ. Qual*. 33: 1803-1813.

Libra, R. D., C. F. Wolter, and R. J. Langel. 2004. Nitrogen and phosphorus budgets for Iowa and Iowa watersheds. Technical Information Series Report TIS-47. Iowa City, Iowa: Iowa Geological Survey.

Mallarino, A. P., B. A. Stewart, J. L. Baker, J. A. Downing, and J. E. Sawyer. 2002. Phosphorus indexing for cropland: Overview and basic concepts. *J. Soil Water Cons*. 57: 440-447.

Randall, G. W. 2004. Subsurface drainage flow characteristics during a 15-year period in Minnesota. In *Proc. 8th International Drainage Symposium*, 17-24. St. Joseph, Mich.: ASAE.

Randall, G. W., D. R. Huggins, M. P. Russelle, D. J. Fuchs, W. W. Nelson, and J. L. Anderson. 1997. Nitrate losses through subsurface tile drainage in CRP, alfalfa, and row crop systems. *J. Environ. Qual*. 26: 1240-1247.

Royer, T. V., M. B. David, and L. E. Gentry. 2006. Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: Implications for reducing nutrient loading to the Mississippi River. *Environ. Sci. Technol.* 40: 4126-4131.

Sawyer, J., E. Nafziger, G. Randall, L. Bundy, G. Rehm, and B. Joern. 2006. Concepts and rationale for regional nitrogen rate guidelines for corn. Extension Bulletin PM-2015, Ames, Iowa: Iowa State University.

Schilling, K. E. 2005. Relation of baseflow to row crop intensity in Iowa. *Agric. Ecosys. Environ*. 105: 433-438.

Pp. 19-28 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop.* St. Joseph, Michigan: ASABE.

Drainage Water Management: A Practice for Reducing Nitrate Loads from Subsurface Drainage Systems

2

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Drainage water management (DWM, often referred to as controlled drainage), as defined herein, is the practice in which the outlet from a conventional drainage system is intercepted by a water control structure that effectively functions as an in-line weir, allowing the drainage outlet elevation to be artificially set at levels ranging from the soil surface to the bottom of the drains, as shown in figure 2-1.

Types of structures in common usage are shown in figure 2-2. Water table level is controlled with these structures by adding or removing "stop logs" or by using float mechanisms to regulate the opening/closing of a flow valve. There are many variations in the shapes and sizes of structures. Flashboard structures may either be manually

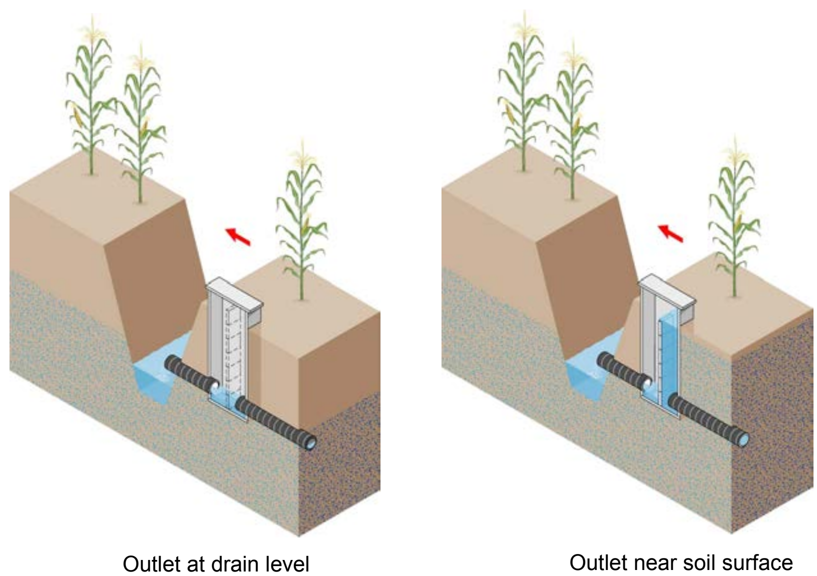

Figure 2-1. Using control structures to manipulate water table levels.

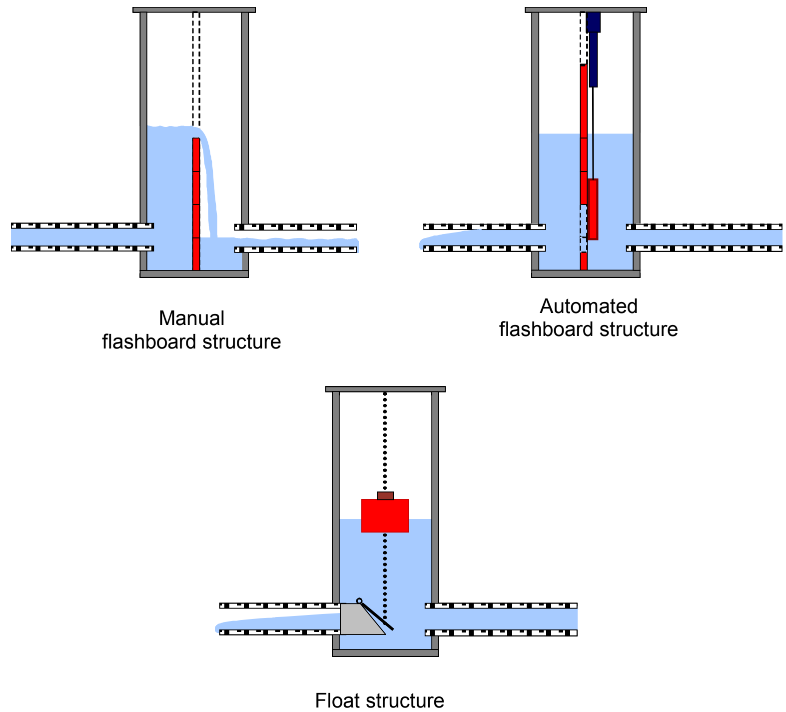

Figure 2-2. Types of water table control structures.

operated or automated to adjust the outlet elevation on fixed dates or in response to rainfall patterns.

Drainage water management practices can target agronomic goals, environmental (water quality) goals, or both. The drainage outlet elevation can be set at or close to the soil surface between growing seasons to recharge the water table, thereby temporarily retaining soil water containing nitrate in the soil profile where it may be subjected to attenuating and nitrate transforming processes, depending on soil temperature and microbiological activity. In addition, it is possible to raise the outlet elevation after planting to help increase water availability to then-shallow plant roots, and to raise or lower it throughout the growing season in response to precipitation conditions. In some soils, water may even be added during very dry periods to reduce crop loss from drought, and this related practice is termed subirrigation. However, the drain spacing for subirrigation may be one-half to one-third the recommended value for drainage, to maintain a water table at a proper depth to reduce deficit crop stress without increasing excess water stress.

Although there have reported instances where DWM has resulted in reduced nitrate concentrations in drain outflow, the general consensus is that the dominant process leading to reductions in nitrate loads is a reduction in drain outflow. With less water leaving the field through the drain pipe, there is potentially less nitrate flowing out of the drain, even with no change in nitrate concentration.

The installation of drainage water management control structures is guided by National Handbook of Conservation Practices (NHCP) Practice Standard 554, Drainage Water Management (NRCS, 2005). Several states have developed local variations of this standard.

Potential

Researchers in North Carolina were among the first to recognize the potential of DWM for reducing N losses from drained lands (Gilliam et al., 1979; Skaggs and Gilliam, 1981). They conducted several field research and demonstration projects to investigate the effectiveness of the method (Gilliam et al., 1978; Doty et al., 1985), developed design guidelines (Gilliam and Skaggs, 1986; Evans and Skaggs, 1989), and demonstrated the application of the method (Evans et al., 1990, 2000). Based on these research and field demonstration projects, conducted in cooperation with USDA-NRCS, DWM was accepted as a BMP by NRCS for reducing N contributions to surface waters. Installation expenses are cost-shared for water quality purposes by the state of North Carolina (Gilliam et al., 1997). The researchers in North Carolina also conducted research to determine the effect of drainage on N loss (Gilliam and Skaggs, 1986; Skaggs and Gilliam, 1981; Skaggs and Chescheir, 2003; Burchell et al., 2005) and developed simulation models to predict those effects (Breve et al., 1997; Luo et al., 2000). In order to evaluate the effects of fall fertilization, consider all forms of mineral and organic fertilizers, and the carryover of N in the various forms of soil and plant organic matter, DRAINMOD-NII was developed to describe a detailed N cycle (Youssef, 2003; Youssef et al., 2004, 2005).

Various researchers in other regions have also found that drainage water management leads to reductions in chemical transport from agricultural fields. In a three-year experiment in Iowa, Kalita and Kanwar (1993) examined the effect of outlet level on crop yield and nitrogen concentration in a DWM system. They observed a reduction in nitrate concentration for all outlet levels, and an increase in crop yield for most. They also found, however, that it was possible to obtain reduced yields by setting the outlet too close to the soil surface during the growing season. Drury et al. (1996) reported a 25% decrease in mean nitrate concentration and a 49% decrease in the total annual nitrate load when drainage water management was implemented on clay loam soil in southwestern Ontario. They did not report the effect on crop yield. Lalonde et al. (1996), working with two-year corn/soybean rotation on a silt loam soil in Quebec, measured nitrate concentration reductions of 76% and 69%, compared to conventional subsurface drainage, for two outlet levels in drainage water management systems. Cooper et al. (1991) reported increased yields ranging from 23% to 58% over three years from establishing a controlled drainage system in Ohio. In their experiment, the control plots were designed as combined drainage and subirrigation system where water was added during most of the growing season. Thus, their results are not necessarily representative of the advantages of moving from conventional drainage to drainage water management. Taken together, however, all these results indicate that drainage water management appears to benefit the environment without adversely affecting yields, if properly managed.

Researchers have reported reductions in nitrate loads due to drainage water management ranging from 14% (Liaghat and Prasher, 1997) to 87% (Gilliam et al., 1979). A conservative estimate by consensus of drainage researchers is that drainage water management can lead to a 30% to 40% reduction in average annual nitrate loads in regions where appreciable drainage occurs in late fall and winter. Measured average annual nitrate-N concentrations from subsurface-drained fields in Illinois ranged from 8 to 19 mg/L depending on cropping practice and the timing of fertilizer application, while average annual nitrate loads ranged from 79 to 115 kg/ha (Algoazany et al., 2005). Based on the 30% estimate, the practice would lead to loading reductions of 24 to 35 kg/ha for those conditions.

Drainage water management systems can be managed to store water in the soil profile and potentially enhance crop yields. In the 2004 crop year, farmers in Illinois reported yield increases of 0.3 to 0.6 MT/ha for corn and 0.2 to 0.4 MT/ha for soybean due to the implementation of drainage water management. However, these are only anecdotal reports; research on the yield benefits of this practice is in the early stages, and may vary by soil and climate. The practice can also be used to benefit wildlife by creating ponded conditions in some fields during the fallow period, providing temporary aquatic habitats for migrating birds.

Currently, there are no good estimates of the extent in the Midwest to which drainage water management systems have been adopted. With the exception of several research and demonstration sites, this practice is a fairly recent introduction to the region, with the majority of systems being installed in the last five years. However, the practice is catching on, partly because of the potential benefits to the environment and partly because of perceived yield benefits.

Related Practices

Drainage water management is just one of several practices involving the design, modification, or operation of subsurface drainage systems to reduce nitrate export. Other practices are at various stages of development and do not have as long a history of implementation. These practices include bioreactors and shallow drainage systems, both of which can be combined with drainage water management.

Bioreactors

Bioreactors are essentially subsurface trenches filled with a carbon source through which water is allowed to flow just before leaving the field to enter a surface water body. The carbon source in the trench serves as a substrate for bacteria that break down the nitrate through the process of denitrification. A bioreactor provides many advantages, such as: (1) it uses proven technology, (2) it requires no modification of current practices, (3) no land needs to be taken out of production, (4) there is no decrease in drainage effectiveness, (5) it requires little or no maintenance, and (6) it can last for up to 20 years. Cooke et al. (2001), Doheny (2002), and Wildman (2002) established relationships between nitrate removal and retention time for laboratory-scale bioreactors with various carbon sources, including woodchips, corn cobs, corn oil, ethanol, or mixtures of these substances. Wildman (2002) and van Driel et al. (2006) measured significant nitrate removal rates in field-scale systems.

Shallow Drainage Systems

Drainage intensity may be defined as the depth of water drained in lowering the water table, initially at the soil surface, by 30 cm in 24 hours. It is a function of hydraulic conductivity, drainable porosity, and drain depth and spacing. In a given soil, different combinations of depth and spacing result in the same drainage coefficient, but they may be different in their water quality response. Experiments are being conducted to determine depth/spacing combinations that optimize productivity with minimum adverse water quality effects. Preliminary results seem to suggest that shallower drains placed closer together produce reduced nitrate loadings when compared to deeper drains placed farther apart (Cooke et al., 2002; Burchell et al., 2005; Sands et al., 2003, 2006).

Important Factors

Drainage water management is best suited for flat, uniform fields with soils that require artificial subsurface drainage. The practice is generally recommended for fields with slopes of 1% or less, but it may be considered for fields with slopes up to 2%. For land slopes greater than 1% to 2%, the increased cost of the drainage water management system may be prohibitive. As a control structure is recommended for each 30 to 45 cm change in field elevation, the cost of a system increases with increasing slope because more structures and drainage mains are required. The practice is also not recommended in instances where elevating the water table would have an adverse effect on adjacent fields.

Under prolonged dry conditions, there may not be enough water (from rainfall) to produce drain outflow. When there is no drain outflow, the elevated outlet is not holding back any water. In this case, drainage water management systems will not offer an advantage over conventional drainage systems (for yield or water quality). Under these conditions, the transport of nutrients through drainage systems is not a significant problem. Under prolonged wet conditions, the proportion of water retained may be small compared to the total outflow; consequently, drainage water management systems may have limited effectiveness.

Limitations

According to 1985 estimates, there are close to 13 million hectares in the Midwest (Iowa, Illinois, Indiana, Ohio, Michigan Minnesota, Missouri, and Wisconsin) that have some degree of subsurface drainage (Pavelis, 1987). Figure 2-3 shows the areas that have the potential to benefit from subsurface drainage based on drainage class (poorly or very poorly drained), hydrologic soil groups (hydrologic soil groups C and D) and slope (less than 2%). Theoretically, drainage water management could be implemented on all of these areas. However, there are practical limitations on a portion of these areas, such as the fact that many existing drainage systems were not designed for drainage water management, thus making retrofitting expensive. In addition, the practice is economically challenging on some slopes greater than 1% to 2%.

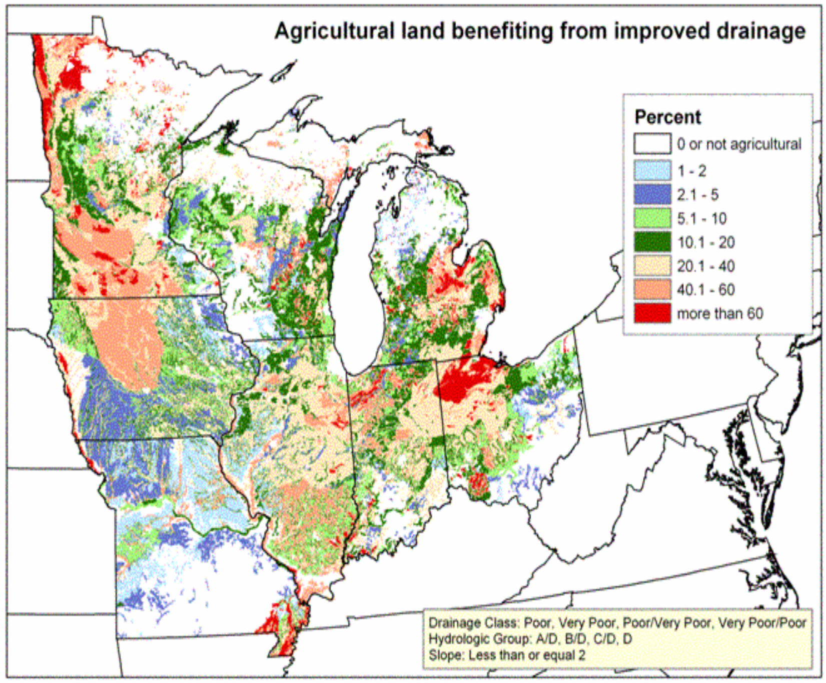

Figure 2-3. Agricultural land in the Midwest with the potential to benefit from drainage water management.

Implementation

Existing drainage systems can be retrofitted for drainage water management by installing control structures at a cost of as little as $100 per hectare. For new systems, additional costs are incurred by designing the drainage systems to optimize the benefits of drainage water management. Typically, drainage systems are designed to minimize the cost of installation. However, such designs do not necessarily maximize the benefits of drainage water management. Shown in figure 2-4 are two possible drainage systems that could be installed on the same field. One design optimizes costs, while the other optimizes the efficacy of drainage water management. In all likelihood, the lower-cost system would be the one selected for installation. Based on an analysis of several fields in Illinois, the average difference in cost, based on average installation costs, is $120 per hectare. Thus, the cost of implementing drainage water management ranges from $100 to $220 per hectare. The lower cost would be applicable to a retrofitted system on a flat field, while the higher figure would apply to a new system on complex topography. If these numbers are combined with the figures for a 30% nitrate load reduction, then the annualized cost for nitrate amelioration with drainage water management systems ranges from $3.00 - $4.20 per kilogram for retrofitted systems on flat fields, to $6.30 - $9.20 per kilogram for new systems on complex topography. Some of this cost may be offset by potential yield increases, but only anecdotal evidence exists regarding yield increase in the Midwest.

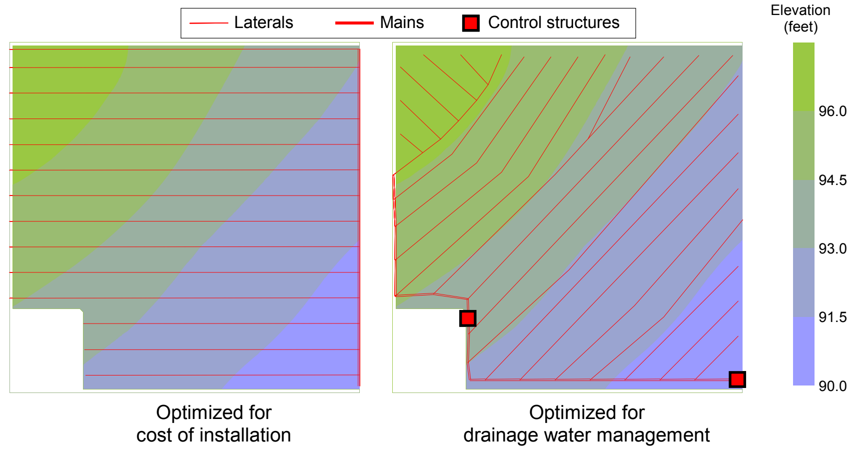

Figure 2-4. Effect of design objective on drainage system layout.

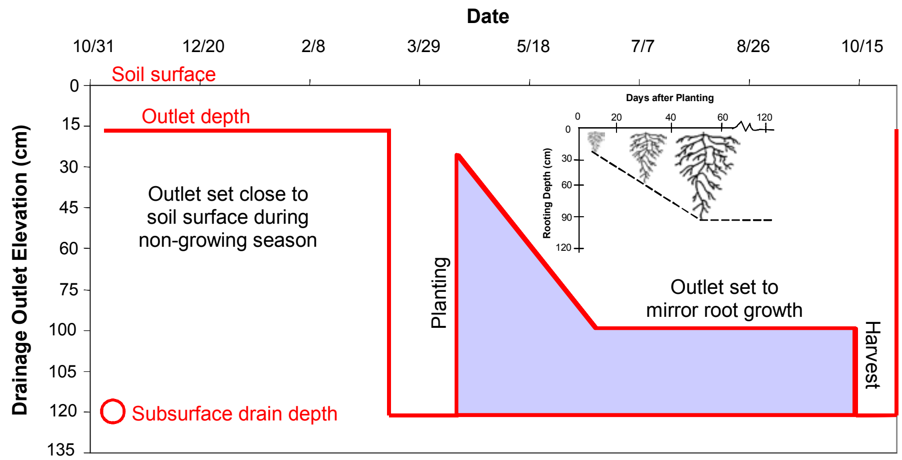

Figure 2-5. Drainage water management system operated for both water quality and yield benefits.

Because drainage water management systems are normally managed during the non-growing season months, there are no crops on the field and thus little potential for yield loss. However, the systems can be used to store water in the soil during the growing season (fig. 2-5), provided there is adequate rainfall and proper management. This water is potentially available for crop consumption and could lead to increase yields. In these instances, if the systems are not managed properly during the growing season, and the water table is allowed to rise into the root zone for extended periods, there is a high risk of reduced yield in very wet years and a moderate risk in normal-rainfall growing seasons.

Long-term computer simulations indicate that the average annual crop yield increase is less than 5%, but it could be substantial in some years. Year-to-year variability depends primarily on growing season precipitation variability and long-term climatic characteristics.

One limitation to determining the efficacy of drainage water management stems from the difficulty in characterizing all the pathways by which water, and by extension the nitrate, leaves a field with an elevated water table. Some of the water may seep laterally or vertically. It is known in some cases that the seepage water gets denitrified, but not known in others. There is also the possibility of increased runoff, which might result in increased sediment and phosphorus transport from the field.

Agricultural Drainage Management Systems Task Force

The Agricultural Drainage Management Systems Task Force (ADMSTF) was formed in 2003 in recognition of the potential for DWM to have an impact on the export of nitrate from drainage systems. This group consists of representatives from universities, USDA-ARS, and USDA-NRCS whose main goal is to "develop a national effort to implement improved drainage water management practices and systems that will enhance crop production, conserve water, and reduce adverse off-site water quality and quantity impacts" (ADMS, 2005). A companion group made up of industry representatives, known as the Agricultural Drainage Management Coalition, has a similar goal. The formation of these two groups has resulted in a greater public awareness of the potential for drainage water management to reduce nitrate transport from subsurface drainage systems.

Since its formation in 2003, the ADMSTF has been working to educate producers, drainage contractors, and conservation professionals about the benefits of drainage water management and to address popular concerns and misconceptions about the practice. The foremost misconception is that when the practice is applied, the drainage outlet is completely closed and no water can flow out of the system. In fact, while the outlet is managed, soil water in excess of that required to elevate the water table to the set outlet level can flow out of the soil profile (fig. 2-1). Other concerns, such as those relating to the impact of DWM on earthworms, potential changes in soil structure, or excessive pressure on and freezing of subsurface drains, are being addressed through research and educational activities.

Summary and Conclusions

Drainage water management has the potential to reduce nitrate loads from subsurface drainage systems by 30% to 40%, mainly by reducing drain outflow volumes. The practice has been proven to be effective in North Carolina, and research is being conducted in several Midwestern states to resolve many questions relating to the practice. In order to assess the benefits of this practice for the Midwest, more information is needed on the crop yield benefits of the practice and how best to manage the systems in the growing season to maximize yields. There is also a need to obtain more information on the water-related properties of many of the soils on which the practice can potentially be implemented. In addition, economic and environmental research is needed to identify and quantify the societal costs of nitrogen enrichment of inland and coastal surface waters. Finally, as with any best management practice, incentives and cost-sharing opportunities for producers must continue to be cultivated to ensure significant adoption of the practice.

References

ADMS. 2005. Agricultural Drainage Management Systems Task Force. Available at: http://extension.osu.edu/~usdasdru/ADMS/ADMSindex.htm.

Algoazany, A. S., P. K. Kalita, J. K. Mitchell, R. A. C. Cooke, and M C. Hirschi. 2005. A long-term monitoring of agricultural chemical transport from a flat tile-drained watershed. ASAE Paper No. 052255. St. Joseph, Mich.: ASAE.

Breve, M. A., R. W. Skaggs, J. E. Parsons, and J. W. Gilliam. 1997. DRAINMOD-N: A nitrogen model for artificially drained soils. *Trans. ASAE* 40(4): 1067-1075.

Burchell II, M. R., R. W. Skaggs, G. M. Chescheir, J. W. Gilliam, and L. A. Amold. 2005. Shallow subsurface drains to reduce nitrate losses from drained agricultural lands. *Trans. ASAE* 48(3): 1079-1089.

Cooke, R. A., A. M. Doheny, and M. C. Hirschi. 2001. Bio-reactors for edge-of-field treatment of tile outflow. ASAE Paper No. 01-2018. St. Joseph, Mich.: ASAE.

Cooke, R., J. Nehmelman, and P. Kalita. 2002. Effect of tile depth on nitrate transport from tile drainage systems. ASAE Paper No. 022017. St. Joseph, Mich.: ASAE.

Cooper, R. L., N. R. Fausey, and J. G. Streeter. 1991. Yield potential of soybeans grown under a subirrigation/drainage water management system. *Agronomy J*. 83(5): 884-887.

Doheny, A. 2002. Amelioration of tile nitrate and atrazine using inline biofilters. Unpublished MS thesis. Urbana, Ill.: University of Illinois.

Doty, C. W., J. E. Parsons, A. W. Badr, A. Nassehzadeh-Tabrizi, and R. W. Skaggs. 1985. Water table control for water resource projects on sandy soils. *J. Soil and Water Conservation* 40(4): 360-364.

Drury, C. F., C. S. Tan, J. D. Gaynor, T. O. Oloya, and T. W. Welacky. 1996. Influence of controlled drainage-subirrigation on surface and tile drainage nitrate loss. *J. Environ. Qual.* 25: 317-324.

Evans, R. O., and R. W. Skaggs. 1989. Design guidelines for water table management systems on coastal plain soils. *Applied Eng. in Agric*. 5(4): 539-548.

Evans, R. O., J. W. Gilliam, and R. W. Skaggs. 1990. Controlled drainage management guidelines for improving water quality. Pub. No. AG-443. Raleigh, N.C.: North Carolina Cooperative Extension Service.

Evans, R. O., J. Paul Lilley, R. W. Skaggs, and J. W. Gilliam. 2000. Rural land use, water movement, coastal water quality. Pub. No. AG-605. Raleigh, N.C.: North Carolina Cooperative Extension Service.

Gilliam, J. W., and R. W. Skaggs. 1986. Controlled agricultural drainage to maintain water quality. *J. Irrig. Drain. Eng*. 112(3): 254-263.

Gilliam, J. W., R. W. Skaggs, and S. B. Weed. 1978. An evaluation of the potential for using drainage control to reduce nitrate loss from agricultural fields to surface waters. Report No. 128. Raleigh, N.C.: University of North Carolina, Water Resources Research Institute.

Gilliam, J. W., R. W. Skaggs, and S. B. Weed. 1979. Drainage control to diminish nitrate loss from agricultural fields. *J. Environ. Qual.* 8(1): 137-142.

Gilliam, J. W., D. L. Osmond, and R. O. Evans. 1997. Selected agricultural best management practices to control nitrogen in the Neuse River basin. Technical Bulletin 311. Raleigh, N.C.: North Carolina State University, North Carolina Agricultural Research Service.

Kalita, P. K., and R. S. Kanwar. 1993. Effect of water table management practices on the transport of nitrate-N to shallow groundwater. *Trans. ASAE* 36(2): 413-422.

Lalonde, V., C. A. Madramootoo, L. Trenholm, and R. S. Broughton. 1996. Effects of controlled drainage on nitrate concentrations in subsurface drain discharge. *Agric. Water Management* 29: 187-199.

Liaghat, A., and S. O. Prasher. 1997. Role of soil and grass strips in reducing nitrate-N pollution in subsurface-drained farmlands: Lysimeter results. *Canadian Water Resources J*. 22(3): 117-127.

Luo, W., R. W. Skaggs, and G. M. Chescheir. 2000. DRAINMOD modifications for cold conditions. *Trans. ASAE* 43(6): 1569-1582.

NRCS. 2005. Drainage water management. Conservation Practice Standard Code 554. Washington, D.C.: USDA Natural Resources Conservation Service. Available at: ftp://ftp-fc.sc.egov.usda.gov/NHQ/practice-standards/standards/554.pdf.

Pavelis, G. A., ed. 1987. Farm drainage in the United States: History, status, and prospects. Miscellaneous Pub. No. 1455. Washington, D.C.: USDA Economic Research Service.

Sands, G. R., L. M. Busman, W. E. Rugger, and B. H. Hansen. 2003. The impact of drainage depth on water quality in a cold climate. ASAE Paper No. 032265. St. Joseph, Mich.: ASAE.

Sands, G. R., I. Song, L. M. Busman, and B. H. Hansen. 2006. Water quality benefits of “shallow” subsurface drainage systems. ASAE Paper No. 062317. St. Joseph, Mich.: ASAE.

Skaggs, R. W., and J. W. Gilliam. 1981. Effect of drainage system design and operation on nitrate transport. *Trans. ASAE* 24: 939-934, 940.

Skaggs, R. W., and G. M. Chescheir. 2003. Effects of subsurface drain depth on nitrogen losses from drained lands. *Trans. ASAE* 46(2): 237-244.

Van Driel, P. W., L. C. Merkley, and W. D. Robertson. 2006. Denitrification of agricultural drainage using wood-based reactors: *Trans. ASABE* 49(2): 565-573.

Wildman, T. A. 2002. Design of field-scale bioreactors for bioremediation of nitrate-N in tile drainage effluent. Unpublished MS thesis. Urbana, Ill.: University of Illinois.

Youssef, M. Y. 2003. Modeling nitrogen transport and transformations in high water table soils. PhD diss. Raleigh, N.C.: North Carolina State University.

Youssef, M. A., R. W. Skaggs, G. M. Chescheir, and J. W. Gilliam. 2004. Field testing of the nitrogen model DRAINMOD-N II for North Carolina soils. In *Proc. 8th International Drainage Symposium*, 171-180. R. A. Cooke, ed. St. Joseph, Mich.: ASAE.

Youssef, M. A., R. W. Skaggs, G. M. Chescheir, and J. W. Gilliam. 2005. The nitrogen simulation model DRAINMOD-N II. *Trans. ASAE* 48(2): 611-626.

Pp. 29-42 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop.* St. Joseph, Michigan: ASABE.

Potential of Restored and Constructed Wetlands to Reduce Nutrient Export from Agricultural Watersheds in the Corn Belt

3

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Abstract. *This chapter addresses the potential of restored and constructed emergent marshes to reduce nutrient export from agricultural watersheds. If wetlands are to serve as long-term "sinks" for nutrients, then reductions in nutrient loads must reflect net storage in the system through accumulation and burial in sediments or net loss from the system, for example due to denitrification. Emergent marshes have significant capacity for denitrification of nitrate and for trapping of particulate nutrients and can be particularly effective at reducing nitrate and phosphorous loads from cultivated fields. The potential of wetlands for water quality improvement depends first on the wetlands intercepting a significant fraction of the nutrient load and second on the wetlands being large enough to adequately treat the load they receive. This depends on the type and magnitude of the intercepted load and on the amount of reduction desired. In the case of nitrate, properly positioned wetlands comprising a few percent of the watershed's contributing area could significantly reduce annual exported nitrate load. The case of phosphorous is more complicated and depends for example on whether the phosphorous loads are associated primarily with suspended particles, which wetlands trap with great efficiency, or with dissolved fractions, for which wetland retention is much more variable. Wetlands are generally less effective at retaining dissolved phosphorous than at removing nitrate. There are opportunities for wetland restoration throughout the Corn Belt and widespread potential for wetlands to intercept agricultural drainage and reduce nutrient export to downstream waters.*

Agricultural nutrient losses to streams are a special concern in the U.S. Corn Belt. This region is characterized by intensive row-crop agriculture (fig. 3-1, top) and by correspondingly intensive use of commercial fertilizer. Corn and soybeans are the two largest acreage crops in the region and account for the vast majority of fertilizer use. Since 1950, total acreage of these two crops has increased by about 50%, primarily due to increases in soybean acreage. Over this same period, commercial fertilizer use has increased dramatically to approximately 10 million metric tons per year (Terry and Kirby, 1997). Agricultural nutrient loads to Corn Belt streams are among the highest in the country and are reflected by significantly elevated stream nutrient concentrations (fig. 3-1, bottom). Nitrate nitrogen concentrations in agricultural streams frequently

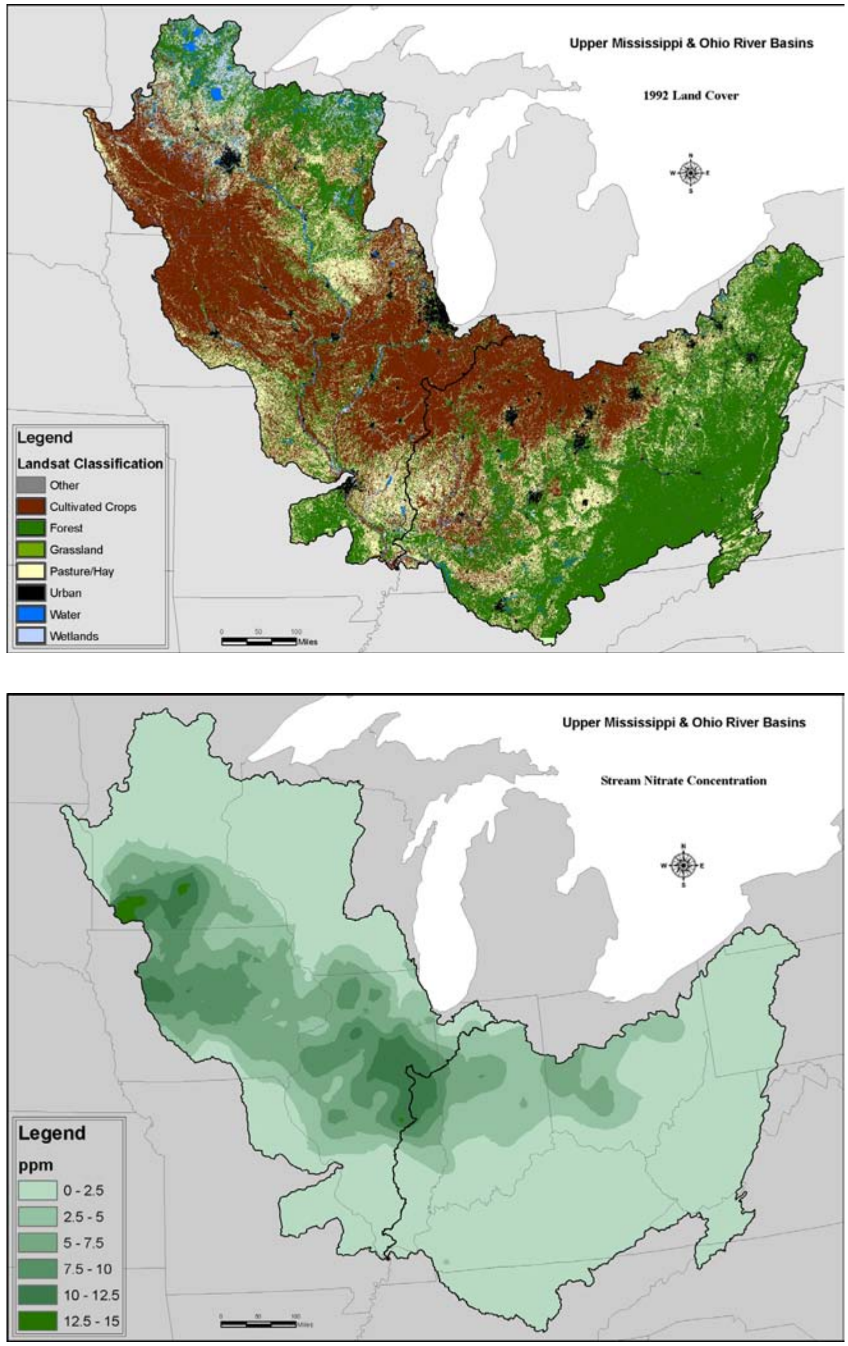

Figure 3-1 (see also inside cover). Land cover based on Landsat data (top) and nitrate concentrations estimated from STORET and state data sets (bottom) for upper Mississippi and Ohio River basins.

exceed the drinking water standard of 10 mg N L^{-1}, and concentrations in tile drainage water are commonly more than double the drinking water standard (Baker et al., 1997, 2004; David et al., 1997; see also chapters 1 and 5). In addition to impacts on water quality within the region, agricultural nutrient loads to Corn Belt streams are considered a primary source of nutrients contributing to hypoxia in the Gulf of Mexico.

Agricultural nutrient loads to surface waters can probably be reduced using a combination of in-field and off-site practices, but the limitations and appropriateness of various alternative practices must be understood in the context of the particular nutrient problem. For example, significant amounts of the fertilizer nitrogen (N) applied to cultivated crops may be lost in agricultural drainage water, primarily in the form of nitrate. In well drained soils, free ammonium applied in fertilizer or derived from mineralization of organic N is converted to nitrate by nitrification. In contrast to ammonium, nitrate is freely mobile and easily transported with infiltrating water to subsurface tile drains. In much of the Corn Belt, the establishment of agricultural drainage networks and the conversion of the natural landscape to annual cropping systems have resulted in increased flow rates and hydraulic loading to streams. In tile-drained landscapes of the Midwest, tile drainage networks are the primary pathway of nitrate transport to surface waters. As a result, grass buffer strips, woody riparian buffers, and many other practices suited to surface runoff have little opportunity to intercept nitrate loads in these areas. In contrast, wetlands sited to intercept tile drainage have the potential to significantly reduce nitrate loads.

Nutrient Transformation and Retention in Wetlands

Wetlands have been shown to be effective in removing a wide variety of water quality contaminants, including suspended solids, nitrogen, and phosphorous (Howard-Williams, 1985; Nixon and Lee, 1986; Kadlec and Knight, 1996; Reddy et al., 1999, 2005). Emergent marshes provide significant potential for denitrification of nitrate and trapping of particulate nutrients and can be particularly effective at reducing nutrient loads associated with agricultural drainage (Braskerud et al., 2005; Crumpton et al., 1995; Crumpton, 2005; Kovacic et al., 2000; Mitsch et al., 2005; Reddy et al., 1999). In general, if wetlands are to serve as long-term "sinks" for nutrients, then reductions in nutrient loads must reflect net storage in the system through accumulation and burial in sediments or net loss from the system, for example through denitrification.

Nitrogen

The processes involved in nitrogen transformation in wetlands are comparable to those in other aquatic systems and soils (Bowden, 1987; Crumpton and Goldsborough, 1998; Howard-Williams, 1985; Reddy and Graetz, 1988). Under anaerobic conditions, NO_3^- can serve as a terminal electron acceptor for the oxidation of organic carbon either through denitrification, resulting in gaseous losses of N_2O or N_2, or through dissimilatory reduction of nitrate to ammonium (Bowden, 1987). Relatively low rates of denitrification and dissimilatory nitrate reduction are observed in natural, unpolluted wetlands (Seitzinger, 1988). However, when wetlands are subjected to significant external nitrate loading, relatively high rates of denitrification can be expected, and with rare exception, denitrification is cited as the primary reason wetlands serve as nitrogen sinks. The effectiveness of wetlands in reducing nitrogen export from agricultural fields will depend on the magnitude and timing of nitrate loads and the capacity of the wetlands to remove nitrate by denitrification.

Phosphorous

In contrast to nitrate, gaseous losses of phosphorous in wetlands (as phosphine) are insignificant. Sediment accretion of bound inorganic phosphorous and unmineralized organic phosphorous is the primary mechanism by which wetlands serve as long-term phosphorous sinks, although a variety of complex processes contribute to shorter-term dynamics of phosphorous uptake and release. In the case of wetlands constructed on former agricultural land, wetlands may initially be net exporters of phosphorous. Depending on cropping practices, these areas can have large accumulations of soil P, some of which can be released under the lower redox conditions of the newly flooded soils (Reddy et al., 2005). A similar problem can occur in wetlands that are allowed to go dry intermittently. When this occurs, there is potential to oxidize the newly accreted, highly organic soils, mineralizing the associated phosphorous. When the wetland is reflooded, this phosphorous is then more likely to be released with the return of lower redox conditions in the reflooded soils (Reddy et al., 1999, 2005). Our understanding of phosphorous loss in wetlands is complicated by the fact that “phosphorous” represents a composite of chemically diverse fractions including particulate, colloidal, and dissolved forms, some of which are very labile and some of which are refractory (Reddy et al., 1999). Water quality studies most commonly measure only total phosphorous and soluble reactive phosphorous.

Greenhouse Gases

A number of studies have researched natural wetlands and peatlands as sources and sinks of greenhouse gases (GHG) (e.g., Bartlett and Harris, 1993). In comparison, less research has been performed on both the short-term and long-term greenhouse gas balance of wetlands restored or constructed for the purposes of nutrient removal (e.g., Tanner et al., 1997). GHG fluxes in emergent marshes capturing agricultural waters are not fully understood and need to be considered in the overall assessment of wetland performance and benefits.

Wetlands can sequester carbon dioxide (CO_2) from the atmosphere through photosynthesis by wetland plants and subsequent carbon storage in above- and below-ground biomass in soil and sediments. The carbon sink capacity of wetland soils can be significant due to low organic matter decomposition rates under saturated (anoxic) conditions and with CO_2 uptake exceeding CO_2 releases from decomposition. The extensive layers of peat underlying many current and former wetlands demonstrate the long-term effectiveness of wetlands as carbon reservoirs. As a carbon sink, wetlands can accrue nearly an order of magnitude more carbon than managed agricultural systems, 1.5 tons of carbon per hectare per year compared to only about 0.2 tons of carbon per hectare per year for agricultural lands (McCarty and Ritchie, 2002).

The efficiency with which wetlands remove carbon is reduced by GHG emissions of CO_2, methane (CH_4), and nitrous oxide (N_2O) (Whiting and Chanton, 2001). The carbon stored in anaerobic wetland soils can be oxidized and released as CO_2 when the area is under dry or drained conditions. In contrast, saturated wetland soils are conducive to CH_4 production through methanogenesis. Methane emissions from anaerobic organic sediments can increase when the sediments are disturbed or through plant-mediated transport (Vretare Strand, 2002). CH_4 production is highest when temperatures are high and nitrate concentrations are low. Research suggests that high nitrate levels inhibit CH_4 production, possibly due to increased redox potential or microbial community competition (Stadmark and Leonardson, 2005). Although N_2O emissions are generally very low in wetlands, there is some concern over increased N_2O emissions in wetlands exposed to high nitrate levels. Relatively few studies have quantified N_2O emissions from wetlands receiving elevated nonpoint-source nitrate loads. These studies confirm that N_2O emissions increase in wetlands at elevated nitrate levels (Paludan and Blicher-Mathiesen, 1996; Stadmark and Leonardson, 2005) but demonstrate that emission rates are very low and N_2O flux accounts for a very small fraction of N removal (Hernandez and Mitsch, 2006; Paludan and Blicher-Mathiesen, 1996; Stadmark and Leonardson, 2005). Hernadez and Mitsch (2006) reported that N_2O efflux accounted for only about 0.3% of total annual N loss in wetlands receiving river flows with elevated nitrate levels. Paludan and Blicher-Mathiesen (1996) did not calculate annual average N_2O flux, but based on their reported flux rates, N_2O efflux represented less than 0.13% of total nitrate removal in a wetland recharged by GW with elevated nitrate levels. N_2O emission rates reported in wetlands receiving agricultural nitrate loads average near 1 μmole N_2O m^{-2} h^{-1} (Hernandez and Mitsch, 2006; Paludan and Blicher-Mathiesen, 1996). These are very similar to rates reported for cultivated crops in the Midwest (1 to 2 μmole N_2O m^{-2} h^{-1}; Parkin and Kaspar, 2006; Grandy et al., 2006) and argue that restoring wetlands on formerly cultivated cropland would have no significant net effect on N_2O emissions.

More information is needed to better understand wetland performance and manage the environmental parameters that can affect carbon storage and GHG emissions in constructed wetlands receiving nutrient-rich agricultural waters. The spatial and temporal variability of GHG fluxes in wetlands is extremely high due to the variation in the environmental factors regulating the microbial processes, such as carbon substrate supply, soil oxidation-reduction status, pH, temperature, electron acceptor availability, hydrologic conditions (e.g., flow, inundation duration, and frequency, etc.), nutrient availability, and presence of vegetation (Altor and Mitsch, 2006; Glass and Gordon, 2005; Stadmark and Leonardson, 2005).

Factors Influencing Wetland Performance

The effectiveness of wetlands in reducing agricultural nutrient loads is influenced by a range of climatological and site-specific factors. Important factors related to wetland inputs include the timing and magnitude of nutrient and hydrologic loads to the wetland, the extent of subsurface tile drainage, the concentrations of nutrients entering the wetland, and the chemical characteristics of nutrients entering the wetland (for example, dissolved versus particulate fractions, nitrate versus ammonium and organic nitrogen, and labile versus refractory forms of phosphorous). Maximum percent reduction in nutrients occurs when residence time is greatest and hydraulic loading rates are low. In the case of nitrate, flood water that transports a large nitrate load rapidly through an otherwise effective wetland may show relatively low percent reductions in nitrate, even though the wetland may be removing a significant mass of nitrate. In the case of phosphorous associated with suspended particles, both percent loss and mass loss rates can be high during periods of high hydrologic loading and short residence times.

In addition to factors related to nutrient and hydrologic inputs, water temperature can have significant effects on nutrient transformation and retention in wetlands, as can the condition of soils and vegetation within the wetland. Carbon and nitrogen transformations are significantly more temperature dependent that those involved with phosphorous retention (Kadlec and Reddy, 2001). Nitrate loss rates may be several times faster during summer months than during the colder winter months, with spring and fall loss rates somewhere between these extremes. Mass reduction is greatest when high nitrate loading coincides with higher temperatures, but in the Corn Belt, nitrate loads generally peak during late winter and spring.

Soils and vegetation can clearly influence wetland performance, particularly in the startup or "adaptation" phase following wetland construction or restoration (Kadlec and Knight, 1996). At a minimum, soils at wetland sites should have low permeability to maintain flooded conditions and should be suitable for the establishment and growth of wetland vegetation. In addition to its obvious habitat value, wetland vegetation significantly influences the water quality performance of wetlands. Wetlands intended to intercept diffuse agricultural nutrient loads will typically be located in lower-lying areas, often on prior converted and farmed wetlands with hydric soils well suited to wetland restoration. Although many soil types can support wetlands, previously drained wetland soils are more likely to provide the required textural properties to guarantee successful wetland establishment.

Potential for Water Quality Improvement

The effectiveness of wetlands for water quality improvement depends on two primary factors. First, wetlands must be positioned to intercept significant nutrient loads if they are to achieve significant load reductions. Second, wetlands must be of sufficient size to allow adequate residence time to treat the loads they receive. For any given location, residence times and load reductions can be increased by increasing the area of wetland relative to the area of the contributing watershed, i.e., the wetland to watershed area (w/w) ratio.

It can be difficult to precisely delineate the contributing watershed area in tile-drained landscapes. When complete and detailed maps of the drainage system are available, the effective area of direct tile drainage can be determined (Kovacic et. al. 2000). Kovacic et al. (2000) determined effective drainage area using a 50 m distance on either side of tile drains. This was based on characteristics of soil permeability (Kurien, 1995). The effective tile drainage area was 50% of the entire watershed area (Kovacic et al., 2000). When detailed drainage maps are not available, it is difficult to calculate the effective area of tile drainage or to differentiate contributing areas of surface runoff and direct tile drainage. In addition, upland areas not directly drained by field tile can contribute to nitrate discharged though drainage networks, and this contribution is difficult to estimate. In the absence of more detailed information, w/w ratio must be determined based on the total area within the surface watershed or drainage district boundaries. The effective area of direct tile drainage is smaller than the area of the surface watershed and, for the same wetland, the w/w ratio will be smaller based on the area of the surface watershed than based on the area of direct tile drainage.

When comparing nutrient removal efficiencies between wetlands based on their w/w ratios, these ratios must be expressed using the same methodology. Over a three-year period from 1995 to 1997, Kovacic et al. (2000) reported annual percent nitrate removal for three Illinois wetlands (A, B, and D) ranging from 33% to 55%, from 37% to 48%, and from 33% to 35%, respectively. The w/w ratios of the wetlands (A, B, and D) were determined to be 4%, 6%, and 3.2%, respectively, based on the effective drainage area estimated using detailed drainage maps. Based on the surface watershed area, the w/w ratios for these same wetlands would be 2%, 3%, and 1.6%. Crumpton et al. (2006) reported annual percent nitrate-N removal of 25%, 68%, and 78% for three Iowa wetlands with w/w ratios of 0.57 %, 2.16%, and 2.25% respectively. Using the same methods to calculate w/w ratios greatly reduces the apparent differences in percent nitrate removal efficiencies between the Iowa and Illinois wetlands. Differences in removal efficiency among wetlands may be related to scale, landscape position, geographic location, loading rates, residence times, concentrations, temperatures, or true differences in w/w ratios.

Performance expectations for wetland restorations must be adjusted for different landscape positions and geographic areas with different patterns of precipitation, volume and timing of surface and subsurface flow, nitrate loading, and temperature. Hydrologic and nitrate loading patterns vary considerably for different landscape positions and different geographic regions of the Corn Belt. The combined effect of variation in land use, precipitation, and runoff means that loading rates to wetlands receiving nonpoint-source loads can be expected to vary by more than an order of magnitude, and will to a large extent determine nitrate loss rates for individual wetlands. Mitsch et al. (2005) developed a model of nitrate retention based on the observed performance of wetlands receiving nonpoint-source nitrate loads either directly from agricultural runoff or from rivers receiving agricultural loads. A nonlinear regression based on annual mass load of nitrate per area of wetland explained 51% of the percent mass nitrate reduction by the wetlands considered. A similar but considerably weaker relationship is found when the analysis is restricted to Corn Belt wetlands receiving seasonally variable water and nutrient loads, i.e., subjected to nonpoint-source loading regimes. Based on 34 “wetland years” of available data (12 wetlands with 1 to 9 years of data each) for sites in Ohio (Mitsch et al., 2005; Zhang and Mitsch, 2000, 2001, 2002, 2004), Illinois (Hey et al., 1994; Kovacic et al., 2000; Phipps, 1997; Phipps and Crumpton, 1994), and Iowa (Crumpton et al., 2006; Davis et al., 1981) examined by

Crumpton et al. (2006), percent mass nitrate removal is much more closely related to hydraulic loading rate (HLR) than to mass loading rate (fig. 3-2). This supports the concept that residence time is the dominant factor controlling percent nitrate removal. A nonlinear function based on HLR [percent mass removal = 103 × (HLR in m $year^{-1})^{-0.33}$] explained 69% of the variability in percent nitrate mass removal in these wetlands compared to only 22% for the best fit function based on mass loading rate [percent mass removal = 118 - 11.07 ln(mass nitrate loading rate in kg N ha^{-1} $year^{-1}$)].

In contrast to percent removal, hydraulic loading rate explains relatively little of the pattern in nitrate mass removal rates. Although total mass removal will obviously be constrained at lower HLRs (because the mass load and potential mass reduction are low at low HLR), mass removal rates vary widely at higher HLRs. Mass nitrate removal rates are considerably more variable than percent nitrate removal among wet-

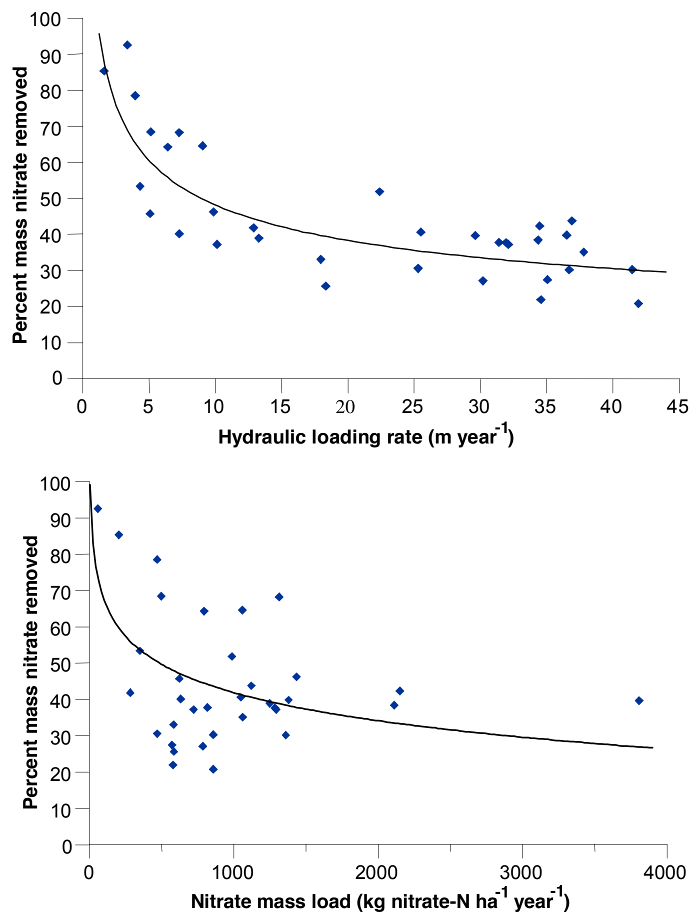

Figure 3-2. Percent mass nitrate removal (top) as a function of hydraulic loading rate (R^2 = 0.69) and (bottom) as a function of mass load (R^2 = 0.22). Adapted from Crumpton et al. (2006).

lands receiving similar hydraulic loading rates. This is to be expected, since mass removal rates are the product of percent removal and mass load and thus include their combined variability. Mass removal rates are affected by hydraulic loading rates, hydraulic efficiency, nitrate concentration, temperature, and wetland condition. Of these, hydraulic loading rate and nitrate concentration are especially important for wetlands intercepting nonpoint-source loads. Hydrologic and nitrate loading patterns vary considerably for different landscape positions and different regions of the Corn Belt. In addition to spatial variation in precipitation (average precipitation declines from southeast to northwest across the Corn Belt), there is tremendous temporal variation in precipitation. The combined effect of these factors means that loading rates to wetlands receiving nonpoint-source loads can be expected to vary by more than an order of magnitude, and will to a large extent determine nitrate loss rates for individual wetlands.

Much of the variability in mass nitrate removal can be accounted for by explicitly considering the effects of HLR and flow-weighted average (FWA) concentration (Crumpton et al., 2006). For the wetlands considered here, it is possible to create a function that calculates mass removal as the product of percent removal [estimated as $103 \times (\text{HLR in m year}^{-1})^{-0.33}$] and mass load [calculated as the product of HLR × FWA nitrate concentration]. Rearranging and accounting for unit conversions, this simplifies to the function [mass nitrate removal in kg ha^{-1} year^{-1} = $10.3 \times (\text{HLR in m year}^{-1})^{0.67} \times$ FWA nitrate concentration in g N m^{-3}]. The relationship can be illustrated by fitting the observed wetland data to a surface plot of this function (fig. 3-3). The isopleths on the function surface illustrate the combinations of HLR and FWA that can be expected to achieve a particular mass loss rate and underscore the benefit of targeting wetland restorations in areas with higher nitrate concentrations. A comparison of the observed and predicted nitrate mass removal demonstrates that the performance of wetlands representing a broad range of loading and loss rates can be reconciled by a model explicitly incorporating hydraulic loading rates and nitrate concentrations (figs. 3-3 and 3-4). The function described above explains 94% of the variability in mass nitrate removal for the wetlands considered here. Because of the variability in FWA nitrate concentrations among these wetlands, percent mass removal is poorly related to absolute mass removal. However, for a given FWA concentration, the highest rates of mass removal are observed at relatively high HLRs, which result in lower rates of percent removal (fig. 3-4). This is an important consideration, since it would be useful to optimize mass load reduction and percent load reduction.

Wetland restoration is a particularly promising approach for heavily tile-drained areas like the U.S. Corn Belt. This region was historically rich in wetlands, and in many areas farming was made possible only as a result of extensive drainage. As a result, there are opportunities for wetland restoration throughout the region, and because of extensive tile drainage systems, there is considerable potential for restored wetlands to intercept tile flow. The greatest benefit of wetlands for mass nitrate reduction will be found in those extensively row-cropped and tile-drained areas of the Corn Belt where the nitrate concentrations and loading rates are highest (fig. 3-1). Crumpton et al.

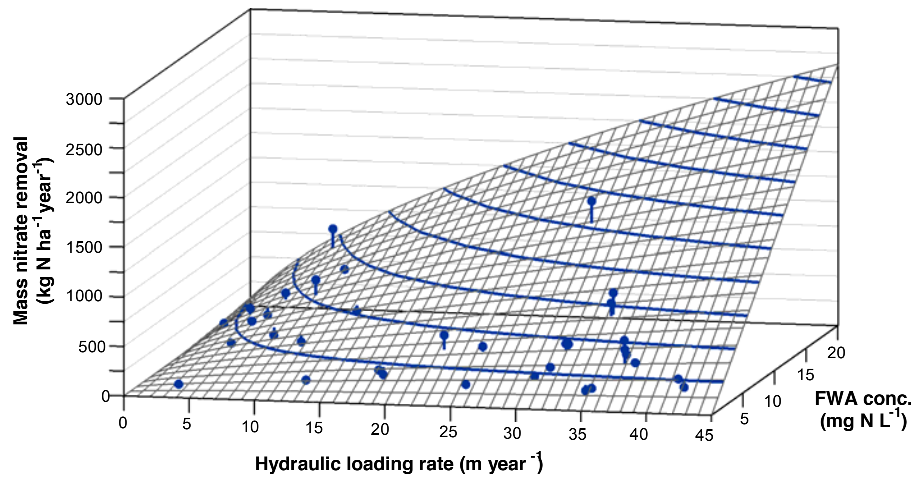

Figure 3-3. Observed (points) and predicted (surface) mass nitrate removal as a function of HLR and FWA nitrate concentrations (R^2 = 0.94). Adapted from Crumpton et al. (2006).

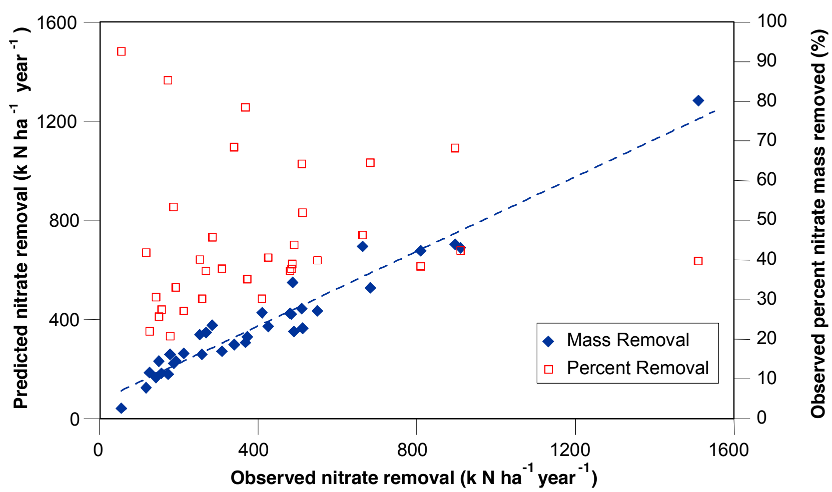

Figure 3-4. Predicted mass nitrate removal (R^2 = 0.94) and observed percent nitrate mass removal versus observed mass nitrate removal.

(2006; also Crumpton, 2005) combined a model of the same general form presented in figure 3-3 with GIS-based estimates of water yield and nitrate concentrations to predict potential nitrate reductions for wetland restorations across the Upper Mississippi and Ohio River basins. That analysis demonstrated significant potential for nitrate reductions if restorations were targeted to those areas of the Corn Belt with the highest nitrate concentrations and loads (fig. 3-5). The actual mass and percent nitrate reductions that can be achieved will depend on our ability to identify and achieve desirable hydraulic and nitrate loading rates.

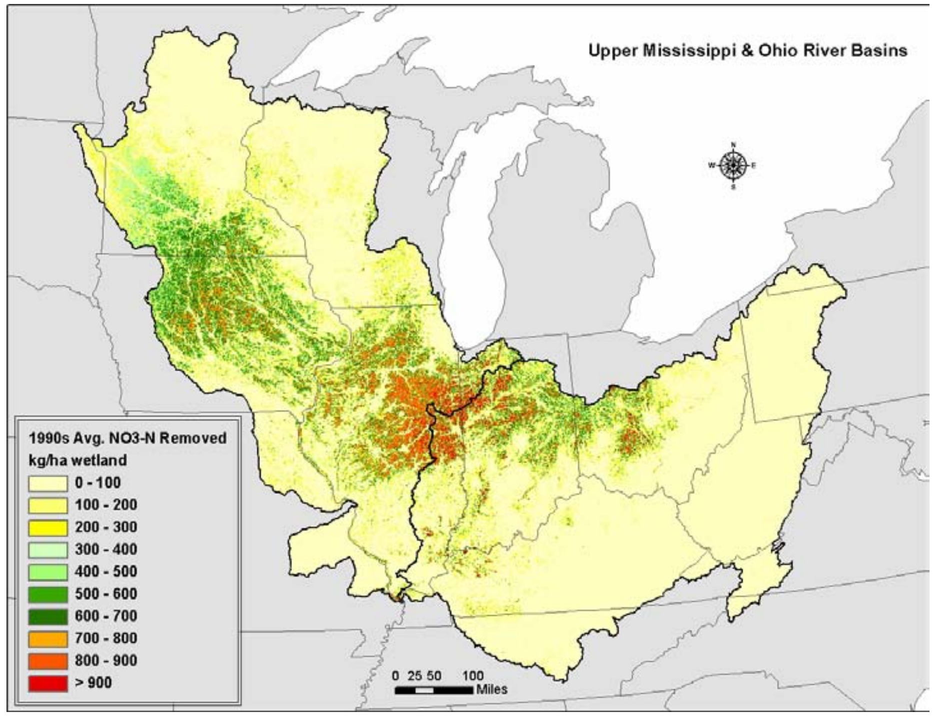

Figure 3-5 (see also inside cover). Estimated average nitrate removal in kg N ha^{-1} of wetland $year^{-1}$ for wetlands with a 2% wetland/watershed ratio. Adapted from Crumpton et al. (2006).

The case of phosphorous is more complicated and depends for example on whether the phosphorous loads are associated primarily with suspended particles, which wetlands trap with great efficiency, or with dissolved fractions, for which wetland retention is much more variable. With the exception of phosphorous associated with suspended solids, wetlands are generally less effective at retaining phosphorous than at removing nitrate (Reddy et al., 1999.)

Factors Influencing the Widespread Use of Wetlands as Nutrient Sinks

Research results over the last couple of decades clearly demonstrate that wetlands can efficiently remove nitrogen and sequester phosphorous and carbon. While the design, operation, and maintenance wetlands need to be better understood, the widespread use of wetlands for nutrient reduction is not limited by science or engineering nor by the availability of suitable land for large-scale wetland restoration (Hey et al., 2004). The main obstacles are related to the scale of effort needed, cost, and policy and regulatory issues.

The primary economic constraint associated with adoption of the practice is the cost associated with wetland restoration and construction and with taking land out of production. These costs vary widely depending on the site characteristics and project size. Land costs are obviously higher for sites located on prime cropland than for those on marginal cropland or pasture, but these costs might be offset by lower construction costs and, at least for nitrate, higher per acre rates of nutrient reduction.

Research Needs

Although most studies report significant nutrient reduction by wetlands, adequate performance data are available for a relatively small number of systems, and there is considerable variability in performance among wetlands, especially in the case of phosphorous. Research is needed to better predict nutrient reduction, carbon sequestration, and greenhouse gas emissions for these systems. Better estimates of hydrologic and nutrient loads to wetlands are needed in order to ensure that wetlands are designed to provide adequate residence time for effective nutrient transformation and retention. Research is needed to determine the effectiveness of wetland restoration in different regions of the Corn Belt with different patterns of precipitation, volume and timing of surface and subsurface flow, nitrate loading, and temperature. This work should develop guidance for effective wetland location and landscape position, effective strategies for wetland management, and effective wetland to watershed ratios for specific regions.

References

Altor, A. E., and W. J. Mitsch. 2006. Methane fluxes from created riparian marshes: Relationship to intermittent versus continuous inundation and emergent macrophytes. *Ecol. Eng.* 28: 224-234.

Baker, J. L., S. K. Mickelson, and W. G. Crumpton. 1997. Integrated crop management and off-site movement of nutrients and pesticides. In *Weed Biology, Soil Management, and Weed Management: Advances in Soil Science*, 135-160. J. C. Hatfield, D. B. Buhler, and B. A. Stewart, eds. Boca Raton, Fla.: CRC Press.

Baker, J. L., S. W. Melvin, D. W. Lemke, P. A. Lawlor, W. G. Crumpton, and M. J. Helmers. 2004. Subsurface drainage in Iowa and the water quality benefits and problem. In *Proc. 8th Intl. Drainage Symposium*, 39-50. R. Cooke, ed. ASAE Publication No. 701P0304. St. Joseph, Mich.: ASAE.

Bartlett, K. B., and R. C. Harris. 1993. Review and assessment of methane emissions from wetlands. *Chemosphere* 26(2): 261-320.

Bowden, W. B. 1987. The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochem.* 4: 313-348.

Braskerud, B. C., K. S. Tonderski, B. Wedding, R. Bakke, A.-G. B. Blankenberg, B. Ulen, and J. Koshiaho. 2005. Can constructed wetlands reduce diffuse phosphorus loads to eutrophic water in cold temperate regions? *J. Environ. Qual.* 34: 2145-2155.

Crumpton, W. G. 2005. Water quality benefits of wetland restoration: A performance-based approach. In *The Conservation Reserve Program - Planting for the Future: Proc. National Conference*, 181-190. USGS Biological Resources Discipline, Scientific Investigations Report 2005-5145. A. W. Allen and M. W. Vandever, eds. Washington, D.C.: U.S. Geological Survey.

Crumpton, W. G., and L. G. Goldsborough. 1998. Nitrogen transformation and fate in prairie wetlands. *Great Plains Research* 8: 57-72.

Crumpton, W. G., J. L. Baker, J. Owens, C. Rose, and J. Stenback. 1995. Wetlands and streams as off-site sinks for agricultural chemicals. In *Clean Water - Clean Environment - 21st Century: Volume I. Pesticides*, 49-53. ASAE Publication No. 295. St. Joseph, Mich.: ASAE.

Crumpton, W. G., G. A. Stenback, B. A. Miller, and M. J. Helmers. 2006. Potential benefits of wetland filters for tile drainage systems: Impact on nitrate loads to Mississippi River subbasins. Project completion report. Washington, D.C.: USDA CSREES.

Crumpton, W. G., G. A. Stenback, B. A. Miller, and M. J. Helmers. 2006. Potential benefits of wetland filters for tile drainage systems: Impact on nitrate loads to Mississippi River subbasins. U. S. Department of Agriculture, CSREES project completion report. Washington, D.C.: USDA CSREES.

David, M. B., L. E. Gentry, D. A. Kovacic, and K. M. Smith. 1997. Nitrogen balance in and export from an agricultural watershed. *J. Environ. Qual.* 26: 1038-1048.

Davis, C. B, J. L. Baker, A. G. van der Valk, and C. E. Beer. 1981. Prairie pothole marshes as traps for nitrogen and phosphorous in agricultural runoff. In B. Richardson (ed.). *Selected Proceedings of the Midwest Conference on Wetland Values and Management*, 152-163., St. Paul, Minn.: The Freshwater Society.

Glass, V., and R. Gordon. 2005. A detailed assessment of gaseous emissions from agricultural treatment constructed wetlands. Climate Change Funding Initiative in Agriculture Final Project Report. Canadian Agri-Food Research Council.

Grandy, A. S., T. D. Loecke, S. Parr, and G. P. Robertson. 2006. Long-term trends in nitrous oxide emissions, soil nitrogen, and crop yields of till and no-till cropping systems. *J. Environ. Qual.* 35: 1487-1495.

Hernandez, M. E., and W. J. Mitsch. 2006. Influence of hydrologic pulses, flooding frequency, and vegetation on nitrous oxide emissions from created riparian marshes. *Wetlands* 26(3): 862-877.

Hey, D. L., A. L. Kenimer, and K. R. Barrett. 1994. Water quality improvement by four experimental wetlands. *Ecol Eng.* 3: 381-397.

Hey, D. L., D. L. Montgomery, L. S. Urban, T. Prato, R. Zarwell, A. Forbes, M. Martell, J. Pollack, Y. Steele. 2004. Flood damage reduction in the upper Mississippi River basin: An ecological means. Chicago, Ill.: The Wetlands Initiative. Available at: www.wetlands-initiative.org/images/UMRBFinalReport.pdf.

Howard-Williams, C. 1985. Cycling and retention of nitrogen and phosphorus in wetlands: A theoretical and applied perspective. *Freshwater Biology* 15: 391-431.

Kadlec, R. H., and R. L. Knight. 1996. *Treatment Wetlands*. Boca Raton, Fla.: Lewis Publishers.

Kadlec, R. H., and K. R. Reddy. 2001. Temperature effects in treatment wetlands. *Water Environ. Research* 73: 543-557.

Kovacic, D. A., M. B. David, L. E. Gentry, K. M. Starks, and R. A. Cooke. 2000. Effectiveness of constructed wetlands in reducing nitrogen and phosphorous export from agricultural tile drainage. *J. Environ. Qual.* 29: 1262-1274.

Kurien, V. M. 1995. Estimation of effective area of influence on random tile drains and monotiles. MS thesis. Urbana-Champaign, Ill.: University of Illinois, Department of Agricultural Engineering.

McCarty, G. W., and J. C. Ritchie. 2002. Impact of soil movement on carbon sequestration in agricultural ecosystems. *Environ. Pollution* 116(3): 423-430.

Mitsch, W. J., J. W. Day, L. Zhang, and R. R. Lane. 2005. Nitrate-nitrogen retention in wetlands in the Mississippi River basin. *Ecol. Eng.* 24: 267-278.

Nixon, S. W., and V. Lee. 1986. Wetlands and water quality: A regional review of recent research in the United States on the role of freshwater and saltwater wetlands as sources, sinks, and transformers of nitrogen, phosphorus, and various heavy metals. Technical Report Y-86-2. Vicksburg, Miss.: Army Corp of Engineers, Waterways Experiment Station.

Paludan, C., and G. Blicher-Mathiesen. 1996. Losses of inorganic carbon and nitrous oxide from a temperate freshwater wetland in relation to nitrate loading. *Biogeochem.* 35(2): 305-326.

Parkin, T. B., and T. C. Kaspar. 2006. Nitrous oxide emissions from corn-soybean systems in the Midwest. *J. Environ. Qual.* 35: 1496-1506.

Phipps, R. G. 1997. Nitrate removal capacity of constructed wetlands. PhD diss. Ames, Iowa: Iowa State University.

Phipps, R. G., and W. G. Crumpton. 1994. Factors affecting nitrogen loss in experimental wetlands with different hydrologic loads. *Ecol. Eng.* 3: 399-408.

Reddy, K. R., and D. A. Graetz. 1988. Carbon and nitrogen dynamics in wetland soils. In *The Ecology and Management of Wetlands*, 307-318. D. D. Hook et al., eds. Portland, Ore.: Timber Press.

Reddy, K. R., R. H. Kadlec, E. Flag, and P. M. Gale. 1999. Phosphorous retention in streams and wetlands: A review. *Crit. Rev. Environ. Sci. Tech.* 29: 83-146.

Reddy, K. R., R. G. Wetzel, and R. H. Kadlec. 2005. Biogeochemistry of phosphorous in wetlands. In *Phosphorous: Agriculture and the Environment*, 263-316. Agronomy Monograph No. 46. Madison, Wisc.: ASA-CSSA-SSSA.

Seitzinger, S. P. 1988. Denitrification in freshwater and coastal marine systems: Ecological and geochemical significance. *Limnol. Ocenaogr.* 33: 702-724.

Stadmark, J., and L. Leonardson. 2005. Emissions of greenhouse gases from ponds constructed for nitrogen removal. *Ecol. Eng.* 25: 542-551.

Tanner, C. C., D. D. Adams, and M. T. Downes. 1997. Methane emissions from constructed wetlands treating agricultural wastewaters. *J. Environ. Qual.* 26(4): 1056-1062.

Terry, D. C., and B. J. Kirby. 1997. Commercial fertilizers 1997. Report prepared by AAPFCO and The Fertilizer Institute, Washington, D.C.

Vretare Strand, V. 2002. The influence of ventilation systems on water depth penetration of emergent macrophytes. *Freshwater Biology* 47: 1005-1097.

Whiting, G. J., and J. P. Chanton. 2001. Greenhouse carbon balance of wetlands: Methane emission versus carbon sequestration. *Tellus* B53(5): 521-528.

Zhang, L., and W. J. Mitsch. 2000. Hydrologic budgets of the two Olentangy River experimental wetlands, 1994-99. In *Olentangy River Wetland Research Park at the Ohio State University, Annual Report 1999*, 41-46. W. J. Mitsch and L. Zhang, eds. Columbus Ohio: Ohio State University.

Zhang, L., and W. J. Mitsch. 2001. Water budgets of the two Olentangy River experimental wetlands in 2000. In *Olentangy River Wetland Research Park at the Ohio State University, Annual Report 2000*, 17-28. W. J. Mitsch and L. Zhang, eds. Columbus Ohio: Ohio State University.

Zhang, L., and W. J. Mitsch. 2002. Water budgets of the two Olentangy River experimental wetlands in 2001. In *Olentangy River Wetland Research Park at the Ohio State University, Annual Report 2001*, 23-34. W. J. Mitsch and L. Zhang, eds. Columbus Ohio: Ohio State University.

Zhang, L., and W. J. Mitsch. 2004. Water budgets of the two Olentangy River experimental wetlands in 2003. In *Olentangy River Wetland Research Park at the Ohio State University, Annual Report 2003*, 39-52. W. J. Mitsch and L. Zhang, eds. Columbus Ohio: Ohio State University.

Pp. 43-58 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop.* St. Joseph, Michigan: ASABE.

Buffers and Vegetative Filter Strips

4

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This chapter describes the use of buffers and vegetative filter strips relative to water quality. In particular, we primarily discuss the herbaceous components of the following NRCS Conservation Practice Standards:

Filter Strip (393)	Alley Cropping (311)
Riparian Forest Buffer (391)	Vegetative Barrier (601)
Conservation Cover (327)	Riparian Herbaceous Cover (390)
Contour Buffer Strips (332)	Grassed Waterway (412)

Placement of most of these practices is illustrated in figure 4-1. Common purposes of these herbaceous components (as defined by the NRCS Conservation Practice Standards) are to:

- Reduce the sediment, particulate organics, and sediment-adsorbed contaminant loadings in runoff.
- Reduce dissolved contaminant loadings in runoff.
- Serve as Zone 3 of a riparian forest buffer.
- Reduce sediment, particulate organics, and sediment-adsorbed contaminant loadings in surface irrigation tailwater.
- Restore, create, or enhance herbaceous habitat for wildlife and beneficial insects.
- Maintain or enhance watershed functions and values.
- Reduce sheet and rill erosion.
- Convey runoff from terraces, diversions, or other water concentrations without causing erosion or flooding (grassed waterway).
- Reduce gully erosion (grassed waterway and vegetative barrier).

The term buffer is used here to generally refer to all eight practice standards noted above. These can be further identified as "edge-of-field" and "in-field" buffers consistent with the terminology used by Dabney et al. (2006). Edge-of-field buffers include filter strips, riparian forest buffers, and riparian herbaceous cover. In-field buffers include conservation cover, contour buffer strips, alley cropping, and grassed waterways. Vegetative barriers could be either in-field or edge-of-field buffers.

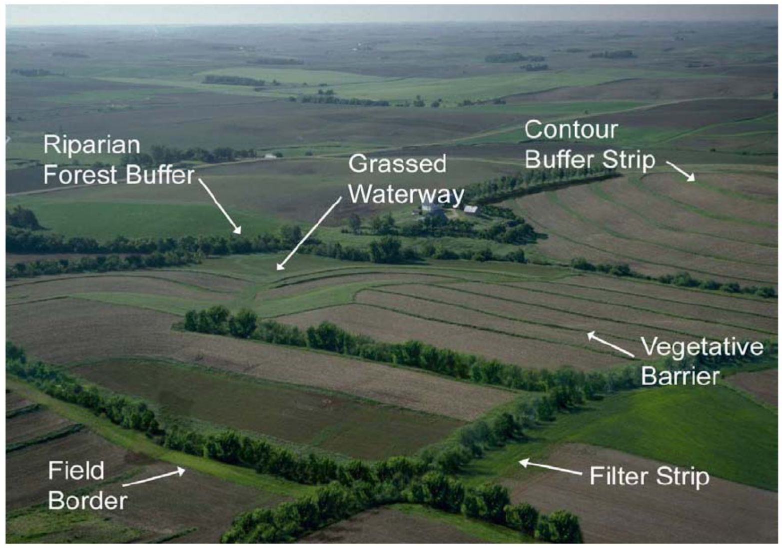

Figure 4-1. Illustration of several vegetative buffer types (photo courtesy of USDA-NRCS).

Processes that influence the environmental impacts provided by these practices include water infiltration, particulate deposition, possible adsorption of soluble pollutants to vegetation and in-place soil, and increased resistance to erosion. Vegetative buffers tend to reduce flow velocities because the vegetation in the buffer provides greater resistance to water flow. This reduction in flow velocity causes deposition of some of the suspended particulates, and the increased resistance to flow can also cause ponding along the upstream edge of the buffer, which promotes infiltration of water and deposition of particulates exiting the field area. Infiltration also takes place within the buffer, which leads to an overall reduction in outflow of water and other contaminants. Together, reduced flow velocity and increased infiltration can offer water quality improvement benefits. Buffers can also promote the uptake of nutrients, denitrification, and assimilation/transformation on the surface of soil, vegetation, and debris. Additionally, there may be a dilution effect on pollutants in the water transported through the buffer due to rainfall interception by the buffer. Another mechanism by which buffers provide water quality improvement is through reduced erosion, since the dense, perennial vegetation generally provides greater resistance to erosion.

Flow conditions vary for the different buffer types. For low flow conditions, the vegetation is expected to remain unsubmerged, but at higher flows the vegetation will be submerged. Of the buffer types described above, grassed waterways are intended to have submerged conditions when functioning in field conditions (Dabney, 2003). As a result, the flow rate entering a buffer system is of primary importance in the functioning of the buffer system. In particular, the flow rate per unit width entering or flowing through the buffer system will affect whether the vegetation is submerged or unsubmerged. The conditions under which vegetation becomes submerged depend on the physical characteristics of the vegetation, including the height, stem density, and stiffness of the vegetation. Dabney (2003) uses specific flow rate (product of flow velocity and depth) to highlight the range of applicability of various buffer systems. Using this method, the specific flow rate range for filter strip type systems is less than approximately 0.22 $ft^2\ s^{-1}$, and the range for grassed waterways is greater than this. Vegetative barriers have specific flow rates that span the range between filter strips and grassed waterways.

Potential Impacts

Surface Processes

Researchers have conducted extensive studies on the pollutant trapping capability of buffers (edge-of-field buffers) where the vegetation has remained unsubmerged. Much of this research has been performed on plot-scale buffer systems. Reported sediment trapping efficiencies have ranged from 41% to 100%, and infiltration efficiencies have ranged from 9% to 100% (Arora et al., 1993; Arora et al., 1996; Barfield et al., 1998; Coyne et al., 1995; Coyne et al., 1998; Daniels and Gilliam, 1996; Dillaha et al., 1989; Hall et al., 1983; Hayes and Hairston, 1983; Lee et al., 2000; Magette et al., 1989; Munoz-Carpena et al., 1999; Parsons et al., 1990; Parsons et al., 1994; Patty et al., 1997; Schmitt et al., 1999; Tingle et al., 1998).

Numerous studies have also examined the nutrient trapping effectiveness of buffers. Dosskey (2001) summarized many of these studies. The buffer trapping efficiency of total phosphorus ranged from 27% to 96% (Dillaha et al., 1989; Magette et al., 1989; Schmitt et al., 1999; Lee et al., 2000; Uusi-Kamppa et al., 2000). The reduction in nitrate-nitrogen (nitrate) ranged from 7% to 100% (Dillaha et al., 1989; Patty et al., 1997; Barfield et al., 1998; Schmitt et al., 1999; and Lee et al., 2000).

As mentioned above, many of the studies on buffer performance have been performed on plot-scale systems. In most of these studies, the ratio of drainage area to buffer area has generally been small, which would be expected to reduce the flow rate per unit width entering the buffer. Thus, this reduced ratio would be expected to reduce the overall loading and loading rate of water and pollutants to the buffer system compared to a case with a greater ratio. In many cases, the ratio of drainage area to buffer area was smaller than might be expected in typical applications. The drainage area to buffer area ranged from 50:1 to 1.5:1 in numerous studies, including those by Arora et al. (1993), Arora et al. (1996), Barfield et al. (1998), Coyne et al. (1995), Coyne et al. (1998), Daniels and Gilliam (1996), Dillaha et al. (1989), Hall et al. (1983), Hayes and Hairston (1983), Lee et al. (2000), Magette et al. (1989), Munoz-Carpena et al. (1999), Parsons et al. (1990), Parsons et al. (1994), Patty et al. (1997), Schmitt et al. (1999), and Tingle et al. (1998).

Of these studies, 50% have a drainage area to buffer area ratio of less than 5:1, whereas a drainage area to buffer area ratio of greater than 20:1 can be expected under most field conditions. For studies with a drainage area to buffer area ratio greater than 10:1 (Arora et al., 1996; Arora et al., 1993; Daniels and Gilliam, 1996; Schmitt et al., 1999; Tingle et al., 1998), the sediment trapping efficiency ranged from 41% to 95%. For a drainage area to buffer area ratio of greater than 10:1, the infiltration ratios ranged from 9% to 98% (Arora et al., 1996; Schmitt et al., 1999). A modeling study showed that higher ratios are expected to produce lower trapping efficiencies (Dosskey et al., 2002). Based on guidelines from the NRCS (1999), the ratio of the drainage area to the buffer area should be 70:1 to 50:1, depending on the RUSLE-R factor in the region. Due to uneven flow distribution, it is likely that the drainage area to a specific region of the buffer will vary with position along the length of the filter. As a result, the drainage area to buffer area ratio will vary, and the areas with the greatest ratio may be contributing the majority of the flow to the system and may need to be considered in the design of a buffer system.

While most studies have been on plot-sized, controlled buffers, the few studies that have investigated unbordered field-scale buffers have shown similar results. Daniels and Gilliam (1996) found that over a range of rainfall events, the buffer reduced sediment loads by 60% to 90%, runoff loads by 50% to 80%, and total phosphorus loads by 50%. The retention of soluble phosphorus was about 20%. The retention of ammonium-nitrogen was 20% to 50%, and the retention of total nitrogen and nitrate was approximately 50%. Sheridan et al. (1999) investigated runoff and sediment transport across a three-zone riparian forest buffer system and monitored the outflow from each zone. Their study showed that runoff reduction in the grass buffer averaged 56% to 72%, and the reduction in sediment transport across the grass buffer ranged from 78% to 83%. They observed no evidence of concentrated flow in the grass buffer portion of their study during the four-year duration of the project, despite a period of high rainfall that included a 100-year, 24-hour storm event. Helmers et al. (2005a) found an average sediment trapping efficiency of 80%.

While the drainage area to buffer ratio captures one source of variability that can affect buffer performance, other variables include condition of the upslope area, degree to which flow concentrates in the upslope area, and the size of the storm event (Lee et al., 2003; Dosskey et al., 2002; Helmers et al., 2002). In some cases, narrow buffers have been shown to provide significant benefits. Narrow buffers (<3 ft.), such as vegetative barriers (in-field and edge-of-field buffers), have been shown to trap significant amounts of sediment (Van Dilk et al., 1996; Blanco-Canqui et al., 2004) and soluble nutrients under conditions where infiltration is increased (Eghball et al., 2000). Gilley et al. (2000) studied the performance of these types of systems under no-till management conditions and found 52% less runoff and 53% less soil loss on plots with grass hedges versus plots without grass hedges. These systems are narrow grass hedges planted on the contour along a hillslope. These hedges normally use stiff-stemmed grasses to reduce overland flow velocity and promote sediment deposition. Grassed hedges are another management practice that has water quality benefits, but their performance will likely be directly tied to how well the vegetation is maintained within the grass hedge. Again, this practice is applicable over a wider range of flow conditions than a buffer, which is intended to intercept shallow overland flow, since grass hedges are designed to control concentrated flow erosion. So, while the drainage area into the buffer is important, the performance of narrow grass hedges highlights that a continuous, well maintained buffer edge may be just as important for maximizing the water quality benefits of these systems.

Research has shown that buffers can remove significant quantities of sediment and nutrients as well as infiltrating a significant portion of the inflow. The reduction in sediment may be generally around 50% for many field settings where the buffer integrity is maintained, but there is likely to be significant variability in the performance of these systems. In general, nutrients that are strongly bound to sediment, such as phosphorus, will have reductions lower than but similar to sediment reductions, but dissolved nutrients will have lower reductions, and their reduction will be closely tied to infiltration. Buffers will likely be less effective for nutrient trapping than for sediment trapping. Daniels and Gilliam (1996) noted that even though buffers are an accepted and highly promoted practice, little quantitative data exist on their effectiveness under unconfined flow-path conditions.

A significant unknown relative to the performance of buffers is how effective they are when flow begins to concentrate and how much of the buffer is effective in treating overland flow. A study by Dosskey et al. (2002) attempted to assess the extent of concentrated flow on four farms in southeast Nebraska and its subsequent impact on sediment trapping efficiency. From visual observations, the researchers estimated an effective buffer area and gross buffer area. The gross buffer area was the total area of the buffer, and the effective buffer area was the area of the buffer that field runoff would encounter as it moved to the stream. Their study showed the effective area, as a percent of the gross area, ranging from 6% to 81%. The modeled sediment trapping efficiency ranged from 15% to 43% for the effective area, compared to 41% to 99% for the gross area. By modeling sediment trapping in a buffer, Helmers et al. (2005b) found that as the convergence of overland flow increases, sediment trapping efficiency is reduced. This concentration of flow, in addition to increasing the flow rate in portions of the buffer that receive runoff, would be expected to adversely affect the overall infiltration and soluble pollutant trapping of the system. Results from these studies show that concentrated flow can reduce the effectiveness of buffers and should be considered in their design. That is, the placement of a buffer may need to be carefully considered so that overland flow is intercepted before it converges or is run through an artificial mechanism to distribute it more evenly for maximum performance. One technique is to use vegetative barriers on the upslope edge of buffers to distribute flow. Another approach is to place vegetative barriers on the contour within the field to minimize the occurrence or magnitude of concentrated flow.

Although grassed waterways (in-field buffers) have been widely used as part of conservation systems, few studies have quantified the reduction in runoff volume and velocity along with sediment delivery through grassed waterways (Fiener and Auerswald, 2003). A study by Briggs et al. (1999) found that grassed waterways reduced the volume of runoff by 47% when compared to non-grassed waterways. Hjelmfelt and Wang (1999) modeled conditions in Missouri for their study. Their data show that a 1,970 ft. grassed waterway with a width of 33 ft reduced the overall volume of runoff by 5%, peak runoff rates by 54%, and sediment yield by 72%.

Another important contribution that grassed waterways and vegetative barriers can provide is protection against gully erosion within agricultural fields. Gully erosion may occur as a result of flow concentration on the landscape. The vegetation in the waterway provides greater resistance to erosion if properly designed. If a waterway can be protected from erosion, then the allowable velocity can be increased. Vegetating the waterway is one form of protection (Haan et al., 1994). In many areas, reducing ephemeral gully erosion can have a significant impact on water quality. Based on studies in 19 states, the USDA (1996) reported that ephemeral gully erosion as a percentage of sheet and rill erosion ranged from 21% to 275%. So, being able to reduce gully erosion would be expected to have a positive impact on downstream water quality, particularly turbidity caused by sediment and phosphorus loss from surface erosion.

Subsurface Processes

While surface water processes are important in evaluating the benefits of buffer systems, they can also intercept shallow groundwater and remove nutrients. Nutrient removal, particularly nitrate removal from shallow groundwater, is one of the common attributes of riparian forest buffers, but clearly not all are equal in this regard. Hill (1996) determined that most riparian forest buffers that remove large amounts of nitrate occur in landscapes with impermeable soil layers near the ground surface. In this setting, nitrate-enriched groundwater from agriculture follows shallow flow paths that increase contact with higher organic matter surface soil and roots of vegetation (Groffman et al., 1992; Hill, 1996). Studies have shown that riparian areas with higher transport rates for subsurface flow (usually with steep terrain and high transmissivities for soils) have the least nitrate attenuation and probably the least denitrification (Jordan et al., 1993).

Denitrification, the microbially mediated reduction of nitrate to nitrogen gases, is an important mechanism for removal of nitrate from groundwater in vegetative buffers (Vidon and Hill, 2004). Denitrification has been measured in a few restored buffers, but in general most of the data come from naturally occurring riparian forests. Denitrification has been measured in riparian and swamp forests in at least 18 different studies, mostly in temperate regions. Not all of the studies were conducted in agricultural watersheds, but there does not seem to be a pattern of the agriculturally impacted riparian areas having higher rates. Rates in the range of 27 to 79 lb N ac^{-1} $year^{-1}$ are not uncommon for these studies, but very low rates in the 0.89 to 4.5 lb N ac^{-1} $year^{-1}$ range are also evident. These studies include a wide variety of systems, ranging from grass buffer areas at field edges to swamp forests. In general, the highest rates were measured from soils of wetter drainage class more highly loaded with N. Nitrogen removal through vegetation assimilation is clearly important (Lowrance et al., 1984), but maintaining assimilation rates requires active management of vegetation.

The capacity of buffers restored on previously cropped soils to remove nitrate is the subject of ongoing studies within the Bear Creek watershed in central Iowa (Simpkins et al., 2002). A focus of these efforts has been to document the capacity of riparian zones to remove nitrate-nitrogen and to elucidate controlling factors. Nitrate-removal efficiency was found to vary between 25% and 100%, with mean nitrate-removal efficiencies ranging from 48% to 85% in shallow groundwater under re-established riparian buffers (Simpkins et al., 2002). The hydrogeologic setting, specifically the direction of groundwater flow and the position of the water table in thin sand aquifers underlying the buffers, is probably the most important factor in determining buffer efficiency (Simpkins et al., 2002). Residence time of groundwater and populations of denitrifying bacteria in the buffer may also be important. Buffer age does not appear to affect removal efficiency. Heterogeneity and larger hydrologic controls will pose challenges to predicting the groundwater quality impacts of future buffers in the watershed.

Factors Impacting Buffer Effectiveness

Buffer Design

Buffers are typically installed with a fixed width. However, due to landscape topography, there are often areas of a buffer that receive greater loading. Bren (1998) proposed using a design procedure in which each element of the buffer has the same ratio of upslope-to-buffer area so that the load to the buffer is constant. Tomer et al. (2003) used terrain-analysis techniques for development of best-management-practice placement strategies, placing buffers according to wetness indices to guarantee that buffer vegetation would intercept overland flow from upslope areas. Since it is unlikely that flow entering the upstream edge of a buffer will be uniformly distributed, it is important to continue to investigate design methods that can maximize the overall effectiveness of the buffer by ensuring that overland flow moves through the buffer. While present buffer designs generally use a fixed-width buffer, consideration should be given to future designs that incorporate variable-width buffers based on the upland contributing area. This may be particularly important where maximizing infiltration is important for reducing soluble pollutant loads to waterbodies.

As with most management practices, there is a time lag with buffers before these systems perform as designed. This time lag depends on how quickly a dense stand of vegetation can be established. There could be much grass growth in a single growing season. However, to ensure long-term performance of the system, it is important to both establish a vigorous and dense stand of vegetation and to maintain the vegetative stand after it is established. The integrity of the buffer system is likely more important than its age in evaluating the effectiveness of the system; thus, some of the benefits could be observed in what may be considered a relatively short time frame.

Site Characteristics

Research has shown that buffers provide water quality benefits, but there is a significant range in the performance of these systems. The performance depends on the field, topographic, and climatic conditions at the site. As discussed above, while there is a significant body of information on the performance of buffers under fairly controlled situations, there is much less information available on the in-field performance of these systems. While it is expected that there will still be significant water quality benefits under these field conditions, it is likely that the performance will be reduced as compared to the results from controlled experiments. In designing buffer systems, the site conditions should be considered to maximize overland flow through the buffer and shallow groundwater interaction with the buffer to take full advantage of the capabilities of the system. While the ratio of drainage area to buffer area and the width of the buffer are factors that can affect the overall performance of the system, research has shown that narrow buffers are also very effective, and some of the most important factors in the performance of the system are the integrity, density, and continuity of the buffer. One of the most important factors to consider in designing or maintaining a buffer is that concentrated flow should be minimized. One method to do this is to ensure that the buffer edges have dense vegetation, which tends to distribute flow.

Since the mechanisms for reducing pollutant transport in buffers ranges from deposition to infiltration, there are numerous factors that influence the physical performance of the buffer regardless of flow concentration. Some of the most sensitive parameters for the hydrologic processes in a buffer include initial soil water content and vertical saturated hydraulic conductivity (Munoz-Carpena et al., 1999). For sediment trapping, some of the most sensitive parameters include the sediment characteristics (particles size, fall velocity, and sediment density) as well as the grass spacing, which affects the resistance to overland flow (Munoz-Carpena et al., 1999). These factors highlight the importance of having a dense stand of vegetation to maximize the pollutant trapping capacity of the buffer.

Soils that have a greater capacity to infiltrate runoff water are likely to have better performance, especially for reducing the mass export of soluble pollutants from surface water runoff. In addition, the sediment trapping capability is greater for larger particles. Thus, when evaluating buffer performance, the eroded (aggregated) sediment size distribution is important. There is a research need for additional data to improve eroded aggregate size distribution predictions as well as for predicting the nitrogen and phosphorus content of each sediment size fraction.

As described previously, the loading, or more specifically the loading rate to the system, will also impact the performance of the system. Some of the variables that influence loading include soil, topography, and management of the upland area. Helmers et al. (2002) found the sediment trapping efficiency to be negatively impacted by the slope of the contributing area, since the higher slopes (10% versus 2%) had greater flow rates entering the buffer system. They also found that as the storm size increased, the sediment trapping efficiency of the buffer decreased. Both of these factors (slope and storm size) influenced the loading, including the flow rate, to the buffer, so as the loading or loading rate increased, the percentage efficiency decreased. However, even though the percent reduction may decrease, the overall mass trapped in the buffer would likely be significant.

Since grassed waterways are designed to convey water off the landscape, such a system must be designed to effectively convey water off the landscape while minimizing channel instability. The hydrology of the site and the soils, particularly in the area of the grassed waterway, need to be considered so that water conveyance is maintained while flow velocities are minimized. While grassed waterways are mainly designed to convey water, as discussed previously, there are also some runoff reduction and direct water quality benefits from grassed waterways. This reduction in runoff will likely be greater under smaller storm and runoff conditions, when the specific flow rate in the grassed waterway is in the range commonly expected for other buffer systems. During larger precipitation events, the grassed waterway will likely function only in a water conveyance capacity.

Limitations on Impact

A large percentage of crop land would benefit from the use of buffers. The scenarios where they would not be expected to have a direct impact on water quality are where there is little runoff and resulting pollutant movement and where the buffer would not intercept shallow groundwater. From a review of the literature, it is evident that buffers provide water quality benefits. However, the effectiveness of buffers will vary significantly depending on the flow conditions in the buffer (e.g., the concentration of flow) as well as the area of the buffer that overland flow will encounter. There is a need to better understand the in-field performance of buffers, where buffer integrity may be comprised by lack of vegetation or features that allow bypass flow to occur through the buffer. Such research would provide much needed information on the performance of this conservation practice under likely common field conditions. This would allow for better evaluation of the range of expected performance. In addition, there are questions about the maintenance required to maximize the performance of the buffer. Most monitoring studies have been short-term in nature, and the long-term performance of buffers with and without some level of maintenance is relatively unknown.

From the review of the literature relative to grassed waterways, it is apparent that only a few studies have quantified the environmental performance of this practice. Differences in grassed waterway design, vegetative conditions, and upland field conditions along with limited data collection make such work difficult. However, the literature also shows that these practices can have a positive impact on water quality and can be effective in reducing peak discharge and sediment yield. Grassed waterways likely improve the quality of the water that enters the channel, and they can also prevent further water quality degradation by reducing ephemeral gully erosion. The available research also indicates positive effects on reducing the volume of runoff. Further investigations in all of these areas are desirable. In particular, there is a need to better understand channel/gully processes, how they contribute to overall delivery of sediment and nutrients to downstream waterbodies, and how practices such as vegetative barriers and grassed waterways can be used to reduce pollutant loading from these mechanisms. While it would be difficult to estimate the direct benefit to water quality improvement on a broad scale, these systems would be expected to be directionally correct. And we know that there is a direct environmental benefit through the reduction in gully erosion with the use of grassed waterways, provided that the waterway is maintained so there is no short-circuiting of flow along the edge of the grassed waterway.

Another area that is in need of future studies is quantifying the percentage of shallow groundwater moving to a particular stream that interacts with the buffer zone. A specific type of landscape in which this might be important is where an extensive subsurface tile drainage system short-circuits subsurface flow through a buffer to streams. Under these conditions, the quantity of shallow groundwater interacting with the root zone of the buffer is likely to be greatly reduced. This effect should be acknowledged in the design, and another conservation practice may be better suited for treating this water. In particular, an edge-of-field practice, such as a wetland, may be more effective in treating the water exiting the subsurface tile lines. In addition, in areas where significant subsurface drainage is present, there may be backslopes on some of the streams or drainage ditches that prevent overland flow from uniformly entering the stream. Instead, the overland flow may flow to a low area and then enter the drain through this pathway, thereby reducing contact with the buffer and the effectiveness of the system. This should be considered when designing the buffer system.

Cost:Benefit Analyses

The costs associated with buffer practices are directly tied to the land that is taken out of production. In some instances, this land could be productive farmland. As such, there is some negative attitude toward installation of these systems. However, a yield reduction in the remainder of the adjacent agricultural land is not expected. Having additional field-scale performance data, particularly where surface water flow concentrates, may improve the acceptance with some producers. Qiu (2003) studied the cost-effectiveness of installing buffers on two small watersheds in Missouri, considering a ten-year evaluation horizon and considering the private costs associated with land opportunity cost and buffer installation cost. For this scenario, the annualized cost of the buffer was \$62.40 ac^{-1} and the annualized benefit was \$73.30, which includes CRP land rental rate and 50% cost share for the installation. For this case, where there was a government subsidy to the producer, there was a net benefit to the producer, so the cost of land taken out of production should be balanced against the value of "green" payments that may offset the cost.

Yuan et al. (2002) studied the cost-effectiveness of various agricultural BMPs in the Mississippi Delta. For their case study, with conventional tillage, they found that edge-of-field buffers reduced sediment yield from 4.5 to 3.7 t ac^{-1} $year^{-1}$ (18% reduction) through the use of filter strips. The approximate cost of sediment reduction for this tillage condition was \$8.5 t^{-1}. When no-till was considered, the reduction in sediment yield due to vegetative filter strips was reduced from 2.2 to 1.6 t ac^{-1} $year^{-1}$ (26% reduction), and the cost of sediment reduction was \$11.8 t^{-1}. Using estimated sediment, total nitrogen, and total phosphorus losses for different tillage practices from chapter 9 of this book and estimated trapping efficiencies for buffers under common field-scale scenarios, the approximate cost per unit reduction in sediment and nutrients is shown in table 4-1. This is a simplified analysis since the cost associated with the practice is just the land rental rate, which was about \$135 ac^{-1} in Iowa in 2005 (ISU Extension, 2005). Other costs would be associated with the buffer, but the major cost would be associated with the land out of production. This type of work highlights the need for establishing what the environmental benefits of these systems are on a field-scale, so that research may be able to help provide a basis for such "green" payments.

The National Conservation Buffer Initiative had a goal of 2 million miles of buffers installed on private land by 2002. Santhi et al. (2001) studied the economic and environmental benefits of this goal and of doubling the size of implementation. Their analysis likely did not consider the overall impacts of concentrated flow on the performance of buffer systems. However, their national estimated reductions in sediment loss, total nitrogen loss, and total phosphorus loss were 15.6%, 10.8%, and 11.7%, respectively, when considering the 2 million mile goal. When the goal was doubled to

Table 4-1. Cost estimates per unit of reduction in sediment, nitrogen, and phosphorus for buffers used in conjunction with two tillage systems.

Treatment System	Loss Estimates[a]	Reduction Range (%)		Pollutant Trapping Range		Annual Operating Cost[b] ($ ac^{-1})	Cost Reduction	
		Low	High	Low	High		Low	High
Sediment	Soil loss (t ac^{-1} year^{-1})			(t ac^{-1} year^{-1})			per ton ($ t^{-1})	
Typical	7.8	40	60	3.1	4.7	6.75	2.2	1.4
No-till	1	40	60	0.4	0.6	6.75	16.9	11.3
Total Nitrogen	N loss (lb ac^{-1} year^{-1})			(lb ac^{-1} year^{-1})			per lb ($ lb^{-1})	
Typical	35.8	30	50	10.7	17.9	6.75	0.6	0.4
No-till	9.7	30	50	2.9	4.9	6.75	2.3	1.4
Phosphorus	P loss (lb ac^{-1} year^{-1})			(lb ac^{-1} year^{-1})			per lb ($ lb^{-1})	
Typical	13.1	30	50	3.9	6.6	6.75	1.7	1.0
No-till	3.1	30	50	0.9	1.6	6.75	7.3	4.4

[a] Loss estimates from chapter 9 of this book.
[b] Assumes 5% of land area in buffer (cost is average land rental rate, $135 ac^{-1}).

4 million miles, the national estimated reductions in sediment loss, total nitrogen loss, and total phosphorus loss were 28.9%, 27.2%, and 25.3%, respectively. While there are significant assumptions in developing these values, this analysis suggests the potential impact that buffer systems might have if 2 million miles or 4 million miles of buffers were installed. Santhi et al. (2001) also estimated the total net cost of these buffers, considering U.S. consumers' loss from reduced supply, program payments to landowners, federal technical assistance cost, and U.S. producers' net gain from higher prices due to the reduced supply. This net cost was then compared to the value of water quality improvements based on studies cited in Ribaudo et al. (1999). From this, Santhi et al. (2001) estimated that the annual net cost of the 2 million mile buffer goal was $793 million and the value of water quality improvements was $3,288 million, for a benefit:cost ratio of 4.1. When they increased the land enrolled in the program to 4 million miles, the cost increased to $1,302 million and the return from water quality improvements was estimated to be $5,650 million, for a benefit:cost ratio of 4.3. They concluded that their analyses showed the buffer programs to be cost-effective.

Interpretive Summary

Practice definition: Buffers and filter strips are areas of permanent vegetation located within and between cropland, grazing land, and disturbed land and the water courses to which they drain. These buffers are intended to intercept and slow runoff, thereby providing water quality benefits. In addition, in many settings, buffers are intended to intercept shallow groundwater moving through the root zone below the buffer.

Site/weather conditions that affect buffer effectiveness: The performance of buffer systems depends on the field, topographic, and climatic conditions at the site. In particular, these factors impact loading to the buffer system. Areas with steeper slopes and fewer in-field conservation practices can be expected to cause greater loading to the buffer. Therefore, the overall performance may be reduced when assessed on the quality of the water exiting the buffer. In addition, more extreme climatic conditions (i.e., greater and more intense precipitation) will also increase loading to the buffer system. However, buffers will still provide a water quality benefit even under more extreme conditions. Depending on site topography, surface water may concentrate prior to being intercepted by the buffer system, which will reduce buffer performance. In designing buffer systems, the potential concentration of surface water runoff should be considered, and to the extent possible, this occurrence should be mitigated by flow redistribution or by intercepting the flow prior to concentration. To maximize buffer performance, loading of water and pollutants should be limited through the use of in-field and edge-of-field conservation practices to maximize contact time with the buffer, and buffers should be properly maintained.

Summary of research findings: Buffers have been found to be most effective in trapping particulate pollutants. In addition, the export of soluble pollutants is expected to decrease when infiltration is maximized. Narrow buffers have also been shown to be effective in reducing the export of particulate pollutants when the integrity of the system is maintained. This highlights that one of the primary functions of buffers is to slow surface water movement, which reduces the export of pollutants, particularly particulate pollutants, and narrow strips of dense grass can function in this capacity and provide water quality benefits (Dabney et al., 2006). Narrow strips could also be used in-field as vegetative barriers to slow pollutant movement in-field and control concentrated flow erosion. To maximize infiltration of runoff, wider buffers or a greater buffer area to source area ratio should be used. Research has found a significant range in buffer performance, with reported sediment trapping efficiencies ranging from 41% to 100% and infiltration efficiencies ranging from 9% to 100%.

Buffers that interact with shallow groundwater moving through the root zone have been found to remove nitrate. Nitrate-removal efficiency has been found to vary between 25% and 100%, with mean nitrate-removal efficiencies ranging from 48% to 85% in shallow groundwater under re-established riparian buffers (Simpkins et al., 2002).

Cost of practice implementation: The costs of buffer systems are associated with the land taken out of production and with planting, establishing, and maintaining the buffers. The costs will vary with location, since land values vary. Qiu (2003) studied the cost-effectiveness of installing buffers on two small watersheds in Missouri, considering a ten-year evaluation horizon and considering the private costs associated with land opportunity cost and buffer installation cost. From this scenario, the annualized cost of the buffer was \$62.40 ac^{-1}.

Potential for water quality improvement: While buffer performance will vary depending on location due to site and climatic factors, research has shown that buffers can have a positive impact on water quality. Research has shown buffers to be most effective in trapping particulate pollutants, but they are also beneficial in reducing the export of soluble pollutants. Buffers are expected to reduce concentrations of nitrogen, phosphorus, and sediment in surface water runoff. In addition, when the buffer's root zone intercepts shallow groundwater, buffers have been shown to reduce nitrate-nitrogen concentrations through plant uptake. The ranges in water quality improvement have been found to vary significantly, but when buffers are designed and maintained properly, they may be expected to trap about 50% of incoming sediment, somewhat less for sediment-bound nutrients, and much less for dissolved nutrients. Nitrate-removal efficiency in shallow groundwater that interacts with the root zone of the buffer has been found to vary, but the mean efficiency may commonly be greater than 50%. However, the percent of groundwater interacting with the root zone of the buffer depends on the geologic and hydrologic conditions of the site and may be limited in cases where subsurface drainage systems short-circuit subsurface flow through the buffer.

In designing a buffer system, the flow contact of either surface water or groundwater with the buffer should be maximized, and the integrity of the vegetation in the buffer should be maintained. While buffers have the potential to provide significant water quality improvement, in-field management needs to be considered along with the implementation of other agricultural best management practices, since buffers best serve as polishers of the water moving through them.

Yuan et al. (2002) studied the cost-effectiveness of various agricultural best management practices (BMPs) in the Mississippi Delta. For their case study, with conventional tillage, they found that vegetative filter strips reduced sediment yield from 4.5 to 3.7 t ac^{-1} $year^{-1}$ (18% reduction). The approximate cost of sediment reduction for this tillage condition was \$8.5 t^{-1}. When no-till was considered, the reduction in sediment yield due to vegetative filter strips was from 2.2 to 1.6 t ac^{-1} $year^{-1}$ (26% reduction), and the cost of sediment reduction was \$11.8 t^{-1}. For a simplified analysis based on Iowa conditions, the cost per ton of sediment reduction ranged from \$1.4 t^{-1} to \$16.9 t^{-1}, the cost per pound of total nitrogen reduction ranged from \$0.4 lb^{-1} to \$2.3 lb^{-1}, and the cost per pound of total phosphorus reduction ranged from \$1.0 lb^{-1} to \$7.3 lb^{-1} (table 4-1).

Extent of area with potential benefit: A large percentage of crop land would benefit from the use of buffers. However, buffers would not be expected to have a direct impact on water quality where there is little runoff and resulting pollutant movement and/or where the buffer would not intercept shallow groundwater. One area in which the water quality benefits may be reduced is in areas where there is significant subsurface drainage such that subsurface flow is short-circuited through the drain lines so that there is minimal interaction with the buffer zone. Some of these areas may also have backslopes on drainage ditches that would likely minimize overland flow through the buffer. Care should be taken to design buffer systems in these locations such that the interaction of surface and ground water with the buffer system is maximized. For example, this may include placing buffers around surface intakes to the subsurface drainage system.

Limitations of adoption: The constraints associated with establishing buffer systems are mainly be associated with the cost of establishing the buffer and the cost to the producer of taking the land out of production. The risks of establishing buffers are that the water quality benefits may be reduced if the buffers are not designed to account for site conditions (i.e., topographic conditions) that minimize the area of the buffer interacting with flow, or the site conditions (e.g., poor soil conditions) that minimize infiltration.

Effect on other resources: Buffers can be expected to have a positive effect on soil and wildlife resources. By converting a portion of cropland to perennial vegetation, we would expect a positive result on soil resources. In addition, the perennial vegetation would provide habitat for wildlife.

Additional research or information needed: There is a need to better understand the in-field performance of buffers, where buffer integrity may be comprised by lack of vegetation or by features that allow bypass flow to occur through the buffer. Such research would provide much needed information on the performance of this conservation practice under likely common field conditions where non-idealized flow may occur. This information would be important for estimating the overall impact of these systems on a watershed scale. There is also a need to evaluate the performance of designs that are specific for water quality improvement. In particular, irregularly shaped buffers that are designed to intercept water as it moves off the source area in a uniform manner should be studied. These may prove to have greater water quality benefits than uniform-width buffers. Finally, there is a need for additional cost:benefit analyses for watersheds to further evaluate the costs and benefits of establishing buffer systems on a watershed scale.

Summary

Buffers and grassed waterways are broadly accepted practices for reducing nutrient runoff from agricultural fields. When properly located, designed, and maintained, buffers may be expected to trap on the order of 50% of incoming sediment, somewhat less for sediment-bound nutrients, and much less for dissolved nutrients. This performance will vary depending on conditions of the buffer and flow through the buffer, and the trapping may be greater than this when flow is nearly uniformly distributed, as has been the case in many plot studies to this point.

The water quality impact will be much lower if the buffer is not properly located, designed, or maintained. In-field management that reduces runoff load and distributes flow evenly along the buffer is important to maximize the effectiveness of the system.

Buffers are cost-effective when considering the water quality benefits. Analysis of the 2 million mile goal indicates a benefit:cost ratio of 4.1; for a 4 million mile goal, the benefit:cost ratio is 4.3.

The accuracy of impact assessments remains limited by lack of research data on watershed-scale effects of buffers and grassed waterways.

References

Arora, K., S. K. Mickelson, J. L. Baker, and D. P. Tierney. 1993. Evaluating herbicide removal by buffer strips under natural rainfall. ASAE Paper No. 932593. St. Joseph, Mich.: ASAE.

Arora, K., S. K. Mickelson, J. L. Baker, D. P. Tierney, and C. J. Peters. 1996. Herbicide retention by vegetative buffer strips from runoff under natural rainfall. *Trans. ASAE* 39(6): 2155-2162.

Barfield, B. J., R. L. Blevins, A. W. Fogle, C. E. Madison, S. Inamdar, D. I. Carey, and V. P. Evangelou. 1998. Water quality impacts of natural filter strips in karst areas. *Trans. ASAE* 41(2): 371-381.

Blanco-Canqui, H., C. J. Gantzer, S. H. Anderson, and E. E. Alberts. 2004. Grass barriers for reduced concentrated flow induced soil and nutrient loss. *SSSA J.* 68: 1963-1972.

Bren, L. J. 1998. The geometry of a constant buffer-loading design method for humid watersheds. *Forest Ecology and Management* 110(1/3): 113-125.

Briggs, J. A., T. Whitwell, and M. B. Riley. 1999. Remediation of herbicides in runoff water from container plant nurseries utilizing grassed waterways. *Weed Tech*. 13(1): 157-164.

Coyne, M. S., R. A. Gilfillen, R. W. Rhodes, and R. L. Blevins. 1995. Soil and fecal coliform trapping by grass filter strips during simulated rain. *J. Soil and Water Conservation* 50(4): 405-408.

Coyne, M. S., R. A. Gilfillen, A. Villalba, Z. Zhang, R. Rhodes, L. Dunn, and R. L. Blevins. 1998. Fecal bacteria trapping by grass filter strips during simulated rain. *J. Soil and Water Conservation* 53(2): 140-145.

Dabney, S. M. 2003. Erosion control, Vegetative. In *Encyclopedia of Soil Science*, 209-213. R. Lal, ed. New York, N.Y.: Marcel Dekker.

Dabney, S. M., M. T. Moore, and M. A. Locke. 2006. Integrated management of in-field, edge-of-field, and after-field buffers. *J. American. Water Resources Assoc*. 42(1):15-24.

Daniels, R. B., and J. W. Gilliam. 1996. Sediment and chemical load reduction by grass and riparian filters. *SSSA J*. 60(1): 246-251.

Dillaha, T. A., R. B. Reneau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint-source pollution control. *Trans. ASAE* 32(2): 513-519.

Dosskey, M. G. 2001. Toward quantifying water pollution abatement in response to installing buffers on crop land. *Environ. Management* 28(5): 577-598.

Dosskey, M. G., M. J. Helmers, D. E. Eisenhauer, T. G. Franti, and K. D. Hoagland. 2002. Assessment of concentrated flow through riparian buffers. *J. Soil and Water Conservation* 57(6): 336-343.

Eghball, B., J. E. Gilley, L. A. Kramer, and T. B. Moorman. 2000. Narrow grass hedge effects on phosphorus and nitrogen in runoff following manure and fertilizer application. *J. Soil and Water Conservation* 55: 172-176.

Fiener, P., and K. Auerswald. 2003. Effectiveness of grassed waterways in reducing runoff and sediment delivery from agricultural watersheds. *J. Environ. Quality* 32(3): 927-936.

Gilley, J. E., B. Eghball, L. A. Kramer, and T. B. Moorman. 2000. Narrow grass hedge effects on runoff and soil loss. *J. Soil and Water Conservation* 55: 190-196.

Groffman, P. M., A. J. Gold, and R. C. Simmons. 1992. Nitrate dynamics in riparian forests: microbial studies. *J. Environ. Quality* 21: 666-671.

Haan, C. T., B. J. Barfield, and J. C. Hayes. 1994. *Design Hydrology and Sedimentology for Small Catchments*. San Diego, Cal.: Academic Press.

Hall, J. K., N. L. Hartwig, and L. D. Hoffman. 1983. Application mode and alternate cropping effects on atrazine losses from a hillside. *J. Environ. Quality* 12(3): 336-340.

Hayes, J. C., and J. E. Hairston. 1983. Modeling the long-term effectiveness of vegetative filters as on-site sediment controls. ASAE Paper No. 832081. St. Joseph, Mich.: ASAE.

Helmers, M. J., D. E. Eisenhauer, M. G. Dosskey, and T. G. Franti. 2002. Modeling vegetative filter performance with VFSMOD. ASAE Paper No. MC02308. St. Joseph, Mich.: ASAE.

Helmers, M. J., D. E. Eisenhauer, M. G. Dosskey, T. G. Franti, J. Brothers, and M. C. McCullough. 2005a. Flow pathways and sediment trapping in a field-scale vegetative filter. *Trans. ASAE* 48(3): 955-968.

Helmers, M. J., D. E. Eisenhauer, T. G. Franti, and M. G. Dosskey. 2005b. Modeling sediment trapping in a vegetative filter accounting for converging overland flow. *Trans. ASAE* 48(2): 541-555.

Hill, A. R. 1996. Nitrate removal in stream riparian zones. *J. Environ. Quality* 25: 743-755.

Hjelmfelt, A., and M. Wang. 1999. Modeling hydrologic and water quality responses to grass waterways. *J. Hydrologic Eng*. 4(3): 251-256.

ISU Extension. 2005. Cash rental rates for Iowa. FM 1851. Ames, Iowa: Iowa State University Extension.

Jordan, T. E., D. L. Correll, and D. E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from cropland. *J. Environ. Quality* 22(3): 467-473.

Lee, K. H., T. M. Isenhart, R. C. Schultz, and S. K. Mickelson. 2000. Multispecies riparian buffers trap sediment and nutrients during rainfall simulations. *J. Environ. Quality* 29(4): 1200-1205.

Lee, K. H., T. M. Isenhart, and R. C. Schultz. 2003. Sediment and nutrient removal in an established multi-species riparian buffer. *J. Soil and Water Conservation* 58: 1-7.

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. *BioScience* 34: 374-377.

Magette, W. L., R. B. Brinsfield, R. E. Palmer, and J. D. Wood. 1989. Nutrient and sediment removal by vegetated filter strips. *Trans. ASAE* 32(2): 663-667.

Munoz-Carpena, R., J. E. Parsons, and J. W. Gilliam. 1999. Modeling hydrology and sediment transport in vegetative filter strips. *J. Hydrology* 214(1/4): 111-129.

NRCS. 1999. Filter strip. National Standard No. 393. Washington, D.C.: USDA Natural Resources Conservation Service.

Parsons, J. E., R. D. Daniels, J. W. Gilliam, and T. A. Dillaha. 1990. Water quality impacts of vegetative filter strips and riparian areas. ASAE Paper No. 902501. St. Joseph, Mich.: ASAE.

Parsons, J. E., J. W. Gilliam, R. Munoz-Carpena, R. B. Daniels, and T. A. Dillaha. 1994. Nutrient and sediment removal by grass and riparian buffers. In *Proc. 2nd Conference on Environmentally Sound Agriculture*, 147-154. K. L. Campbell, W. D. Graham, and A. B. Bottcher, eds. St. Joseph, Mich.: ASAE.

Patty, L., B. Real, and J. J. Gril. 1997. The use of grassed buffer strips to remove pesticides, nitrate and soluble phosphorus compounds from runoff water. *Pesticide Science* 49(3): 243-251.

Qiu, Z. 2003. A VSA-based strategy for placing conservation buffers in agricultural watersheds. *Environ. Management* 32(3): 299-311.

Ribaudo, M. O., D. H. Richard, and M. E. Smith. 1999. Economics of water quality protection from nonpoint sources: Theory and practice. USDA Agric. Econ. Report No. 782. Washington, D.C.: USDA.

Santhi, C., J. D. Atwood, J. Lewis, S. R. Potter, and R. Srinivasan. 2001, Environmental and economic impacts of reaching and doubling the USDA buffer initiative program on water quality. ASAE Paper No. 012068. St. Joseph, Mich.: ASAE.

Schmitt, T. J., M. G. Dosskey, and K. D. Hoagland. 1999. Filter strip performance and processes for different vegetation, widths, and contaminants. *J. Environ. Quality* 28(5): 1479-1489.

Sheridan, J. M., R. Lowrance, and D. D. Bosch. 1999. Management effects on runoff and sediment transport in riparian forest buffers. *Trans. ASAE* 42(1): 55-64.

Simpkins, W. W., T. R. Wineland, R. J. Andress, D. A. Johnston, G. C. Caron, T. M. Isenhart, and R. C. Schultz. 2002. Hydrogeological constraints on riparian buffers for reduction of diffuse pollution: Examples from the Bear Creek watershed in Iowa, USA. *Water Science and Tech.* 45(9): 61-68.

Tingle, C. H., D. R. Shaw, M. Boyette, and G. P. Murphy. 1998. Metolachlor and metribuzin losses in runoff as affected by width of vegetative filter strips. *Weed Science* 46(4): 475-479.

Tomer, M. D., D. E. James, and T. M. Isenhart. 2003. Optimizing the placement of riparian practices in a watershed using terrain analysis. *J. Soil and Water Conservation* 58(4): 198-206.

USDA. 1996. America's private land, a geography of hope. Program Aid 1548. Washington D.C.: USDA Natural Resources Conservation Service.

Uusi-Kamppa, J., B. Braskerud, H. Jansson, N. Syversen, and R. Uusitalo. 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. *J. Environ. Quality* 29(1): 151-158.

Van Dijk, P. M., F. J. P. M. Kwaad, and M. Klapwijk. 1996. Retention of water and sediment by grass strips. *Hydrological Processes* 10:1069-1080.

Vidon, P., and A. R. Hill. 2004. Denitrification and patterns of electron donors and acceptors in eight riparian zones with contrasting hydrogeology. *Biogeochemistry* 71(2): 259-283.

Yuan, Y., S. M. Dabney, and R. L. Bingner. 2002. Cost effectiveness of agricultural BMPs for sediment reduction in the Mississippi Delta. *J. Soil and Water Conservation* 57(5): 259-267.

Pp. 59-71 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop*. St. Joseph, Michigan: ASABE.

Nitrogen Rates

5

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In most crop rotations that include corn, nitrogen (N) applied to the corn phase is a proven and profitable practice. Corn in some rotations requires little to no N input, with first-year corn following established alfalfa as an example. Corn in other rotations requires substantial N input to meet plant requirements, with continuous corn (CC) typically requiring the greatest input. Other rotations or corn phases will be intermediate in N application requirement. With corn in the two most common crop sequences in the Corn Belt, corn following soybean (SC) and CC, if N is not applied, then yield will suffer. If N is not applied on an on-going basis, then over time corn yield will often average around 50 to 60 bu ac^{-1} in CC and 100 to 110 bu ac^{-1} in SC, or less. Consequently, the soil system typically cannot supply the full corn plant N requirement. On average, the yield with no N applied is around 70% in a SC rotation and 55% in CC of the yield obtained at an economic optimum rate. Therefore, supplemental N is needed to reach economic yield potential.

Research measuring corn response to N application has been on-going for over 50 years. Guidelines for suggested N rates based on that research have been derived using economic principles to determine the economic optimum N rate (EONR) rather than maximum yield. Therefore, recommendations are guided by economic return to N application through corn yield increase. The expectation by many is that simply applying N at economic optimum rates will "solve" the issue of nitrate movement from fields in subsurface drainage. However, nitrate losses occur in corn production systems even when no N is applied, and N application at optimum rates increases loss. To date, determination of EONR has not been modified to account for environmental costs resulting from increased nitrate loss to water systems when N is applied, largely due to lack of such cost information and societal decisions on where to partition those costs.

The objectives of this chapter are to review the effect of N application rate for corn on economic return, nitrate in subsurface drainage (tile flow), and potential nitrate reduction.

Economic N Application Rates

Producers should apply N rates that return the most profitable economic yield, where the yield gain from N application will more than pay for the invested N, rather than maximum yield. Nitrogen response trials are conducted where multiple rates of N are applied, and grain yield is measured at each rate. Analysis of that response data allows calculation of site EONR, the rate at which the grain yield increase just pays for the cost of the last increment of applied N (fig. 5-1). Economic net return is the

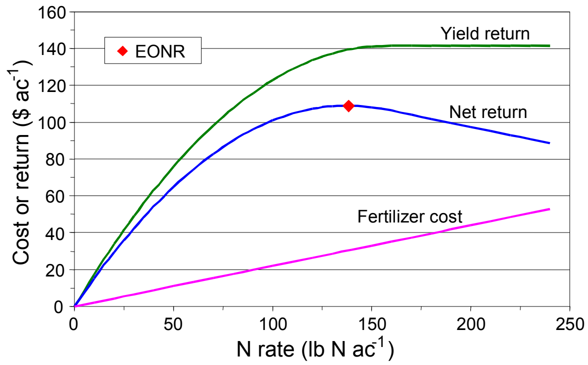
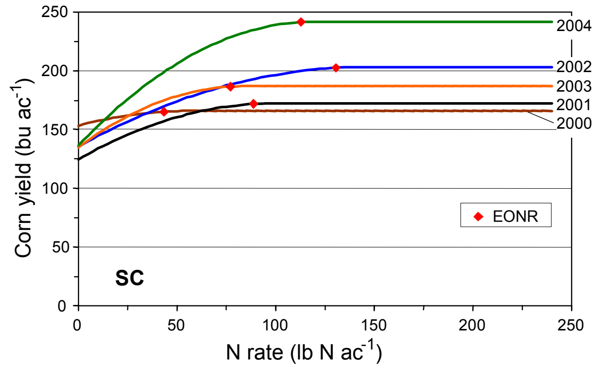


Figure 5-1. Example corn grain yield and fertilizer components of calculated economic net return across N rates from an N response trial, with the economic optimum N rate (EONR) at 0.10 N:corn price ratio ($0.22 lb^{-1} N : $2.20 bu^{-1} corn) indicated by the closed symbol.

difference between the yield gain and N cost. Analysis of response data from many sites is needed to account for typical variation in N response and optimum N across years (fig. 5-2) and locations (fig. 5-3) due to non-controllable factors and to improve determination of the point at which expected maximum economic net return to N (MRTN) occurs (the MRTN approach as described by Nafziger et al., 2004, and Sawyer et al., 2006). The MRTN approach incorporates the uncertainty in yield response to applied N from all sites, uses the diminishing yield increase and maximum response as N rate increases, and provides the point at which the economic net return is maximized across all sites (closed symbols in fig. 5-4). Since the net return is fairly constant at N rates near the MRTN, a range of N rates would be expected to provide similar economic profit (open symbols in fig. 5-4, which are within $1.00 ac^{-1} of the maximum return). This range can provide flexibility in decisions regarding application rate and should provide adequate yield across changing production conditions. Because of the small yield change within the N rate range for maximum profit, rates at the low end of the range will produce greater N use efficiency (more bushels per lb N) and leave less nitrate in the soil for potential loss than rates at the high end of the range. However, the risk of having inadequate N increases.

When N response trials are conducted with corn in different rotations, the MRTN can be calculated for each rotation. Examples are given in figure 5-4 for CC (56 sites) and SC (121 sites) in Iowa for trials conducted approximately the past ten years. In these Iowa trials, the MRTN rate for CC is approximately 175 lb N ac^{-1} and 125 lb N ac^{-1} for SC when the ratio of the N price to corn price is 0.10 ($0.22 lb^{-1} N:$2.20 bu^{-1}). This is a typical difference in economic N rate between these two rotations.

Economic N rates are not necessarily the same across the Corn Belt. Figure 5-5 shows the MRTN rate for CC and SC from recent N response trials conducted in Iowa, Illinois (82 CC sites and 172 SC sites), and Minnesota (68 CC sites and 50 SC sites). Differences can be due to variation in soils, climate, management, and interaction of these factors. These differences must be taken into account as evaluations are made regarding suggested N rates and potential to affect nitrate in drainage water leaving fields.



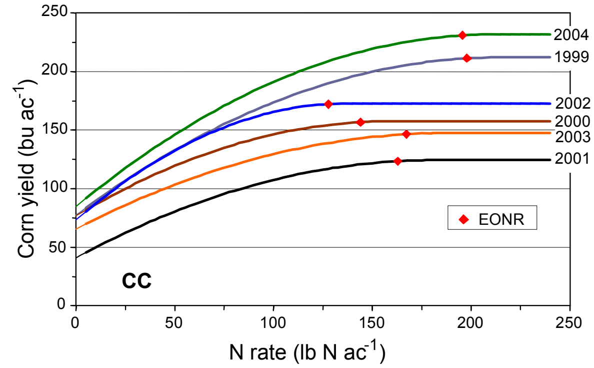


Figure 5-2. Variation in EONR (0.10 price ratio) and yield across years for SC and CC at the same site, Ames, Iowa.

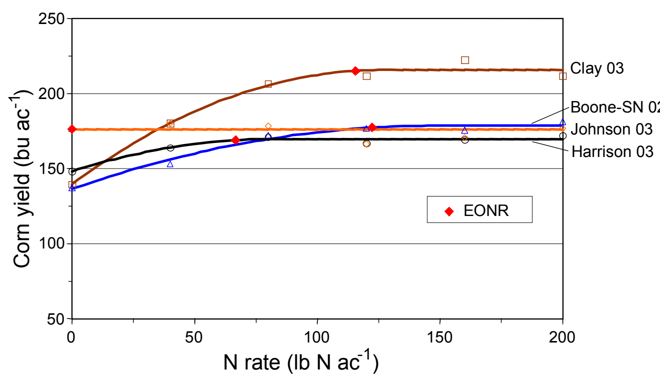


Figure 5-3. Example of variation in response to N and EONR (0.10 price ratio) at different sites in Iowa. Open symbols are measured yield for each N rate.

Economic N rates also change with different relationships between N price and corn price (i.e., the N:corn price ratio, \$ lb^{-1} N : \$ bu^{-1} corn). As shown in figure 5-4, as the N price becomes higher relative to the corn price (i.e., the ratio gets larger), the net return and MRTN rate decrease. In addition, the economic penalty to high N rates

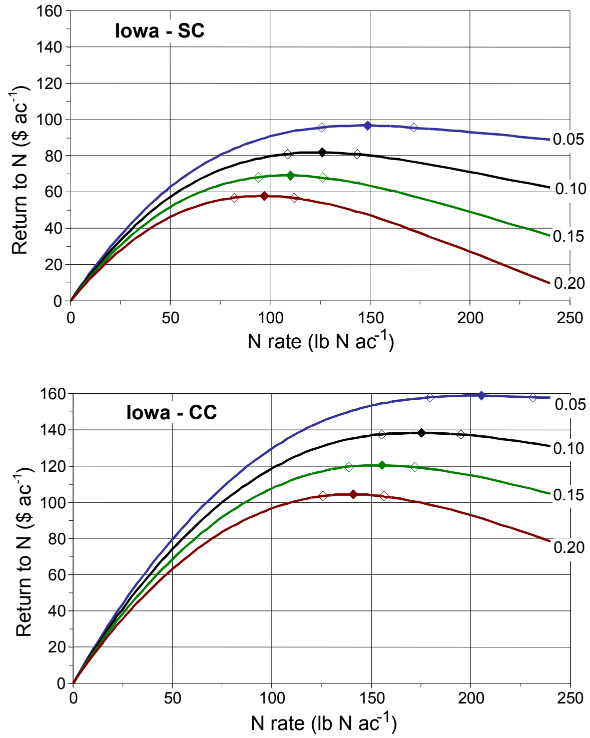


Figure 5-4. Effect of fertilizer N:corn grain price ratio on net return to N (SC and CC rotations in Iowa). The closed symbols correspond to the maximum return to N (MRTN), and the open symbols indicate the range around the MRTN with similar return (within $1.00 ac^{-1} of the maximum return).

above the MRTN increases, as evidenced by the steeper decline in net return as the rate increases above the MRTN. This economic penalty is virtually nonexistent when N is inexpensive (low price ratio), a situation likely recognized by producers and one that may have encouraged high N rates in past years. This situation does not exist today, as N prices have risen substantially. Conversely, there is increased risk of N shortage and severe economic penalty at N rates below the MRTN (fig. 5-4), as evidenced by the rapid decline in net return as N rate declines below the maximum profit range. This is likely the greatest concern for producers: increased production risk and associated severe yield and economic loss due to insufficient N. Incentives for producers to accept increased risk as rates are used at the lower end of the MRTN range could be provided by insurance programs. Another approach is documentation of N adequacy or deficiency with diagnostic tools. Examples include preplant soil testing (PPNT, preplant soil nitrate), in-season soil testing (PSNT, pre-sidedress soil nitrate), plant N stress sensing (hand-held chlorophyll meter, remote aerial color and near-infrared images, pulsed reflective light sensing), and post-season testing (end-of-season stalk nitrate, post-harvest profile nitrate). Continued research on development and refinement of diagnostic tools is needed to improve accuracy and reliability in determining fertilizer N needs.

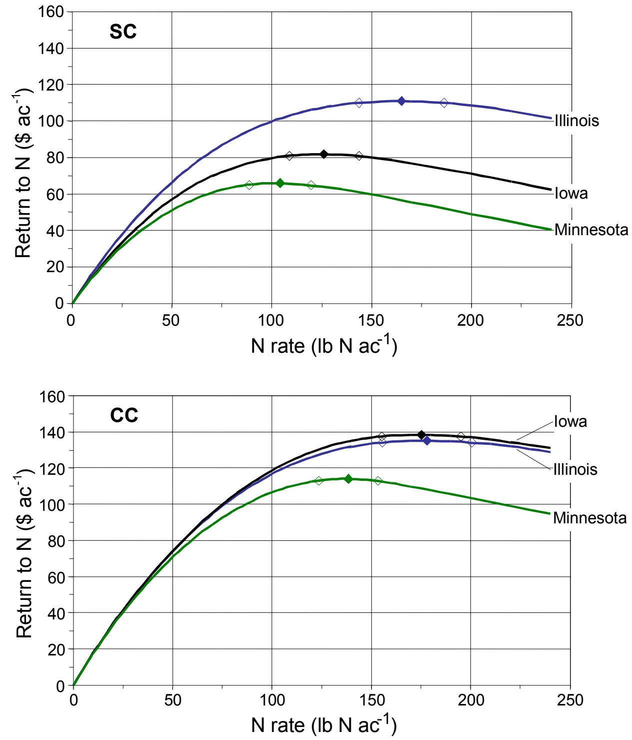


Figure 5-5. Differences between net return to N for SC and CC for various states at a 0.10 N:corn price ratio ($0.22 lb^{-1} N : $2.20 bu^{-1} corn). The closed symbols correspond to the maximum return to N (MRTN), and the open symbols indicate the range around the MRTN with similar return (response data from Illinois courtesy of Emerson Nafziger, University of Illinois).

For sound N management, crop producers should apply the rate of N that provides maximum return to the N investment. This application, however, results in increased soil nitrate, with potential for greater nitrate concentrations moving to water systems. Minimizing nitrate-N concentration or load in drainage water leaving production fields by changing N rate therefore becomes relative to the N rate that provides maximum economic return to N.

Nitrogen Rate and Nitrate-N Losses in Subsurface Drainage

When no N is applied, there is a baseline nitrate-N in subsurface drainage from land cropped to corn or corn in rotation with soybean. This concentration or load varies depending on the climate, soil properties and tile system characteristics, but it often spans the range of 3 to 10 mg L^{-1} or 8 to 20 lb ac^{-1}. As N is applied at increasing rate, the concentration and load of nitrate-N in tile flow increases; examples are shown in tables 5-1 and 5-2 and in figures 5-6 and 5-7, with further examples in Baker et al. (1975), Baker and Johnson (1981), Davis et al. (2000), Jaynes et al. (2001), Kladivko et al. (2004), Jaynes et al. (2004), Clover (2005), and Lawlor et al. (2005). While withholding N application may reduce tile-flow nitrate-N concentrations to less than the USEPA drinking water maximum contaminant level (MCL) standard of 10 mg N L^{-1}, it will not result in concentrations at or less than currently proposed USEPA nutrient ecoregion VI nutrient criteria of 2.18 mg total N L^{-1} for rivers and streams or 0.78 mg total N L^{-1} for lakes and reservoirs (USEPA, 2002).

The change in nitrate in subsurface drainage as N application rate increases is not consistent across locations, but generally increases steadily as N application rate increases (examples in figs. 5-6 and 5-7). Data from some locations show a more rapid increase (curvilinear) as N rate increases, especially well above the EONR. Other locations do not have this trend. While many studies have monitored nitrate in subsurface drainage with a limited number of N rates (due to research cost constraints and interest in multiple practices affecting N loss), there is a scarcity of site data with an adequate number of rates to fully characterize nitrate loss and concurrently determine corn yield response over a long-term period.

It is common to find nitrate-N concentrations in subsurface drainage or discharge from watersheds above the 10 mg N L^{-1} MCL drinking water standard when the EONR or lower rate is applied for corn production (Baker et al., 1975; Baker and Johnson, 1981; Owens et al., 2000; Jaynes et al., 2001; Jaynes et al., 2004; Clover, 2005; Lawlor et al., 2005). In the work of Baker et al. (1975), N applied only to corn at a rate of 100 lb N ac^{-1} in an oat-corn-oat-corn-soybean sequence resulted in an average annual 21 mg nitrate-N L^{-1} in tile flow (site located at Boone, Iowa). Continuing

Table 5-1. Corn production and nitrate loss to tile drainage as affected by rate and time of N application at Waseca, Minnesota, 2000-2003.

N Treatment			Four-Year Average		
Time	N Rate (lb N ac^{-1})	N-Serve	Grain Yield (bu ac^{-1})	Net Return to N[a] (\$ ac^{-1})	Flow-Weighted NO_3-N Conc.[b] (mg L^{-1})
--	0	--	111	--	--
Fall	80	Yes	144	38	11.5
Fall	120	Yes	166	72	13.2
Fall	160	Yes	172	74	18.1
Spring	120	No	180	105	13.7

[a] Corn = \$2.00 bu^{-1}, fall N = \$0.25 lb^{-1}, spring N = \$0.275 lb^{-1}, and N-Serve = \$7.50 ac^{-1}.
[b] Across four SC rotation cycles.

Table 5-2. Corn production and nitrate loss to tile drainage as affected by spring-applied anhydrous ammonia N rate at Filson, Illinois, 2002-2004 (Clover, 2005).

			Change Per 70-lb N Rate Increment			
N Rate (lb N ac^{-1})	Grain Yield (bu ac^{-1})	Tile-Flow NO_3-N[a] (lb ac^{-1})	Yield (bu ac^{-1})	NO_3-N (lb ac^{-1})	Net Loss[b] (\$ ac^{-1})	Net Loss per Unit NO_3-N (\$ lb^{-1})
210	180	61	---	---	---	---
140	169	41	11	20	10	0.52
70	130	30	39	10	68	6.64
0	69	26	61	4	119	29.70

[a] Rotation total from the average across three years of each crop in a SC rotation, i.e., the total amount for the two-year rotation.
[b] Nitrogen at \$0.22 lb^{-1} N and corn grain at \$2.20 bu^{-1}.

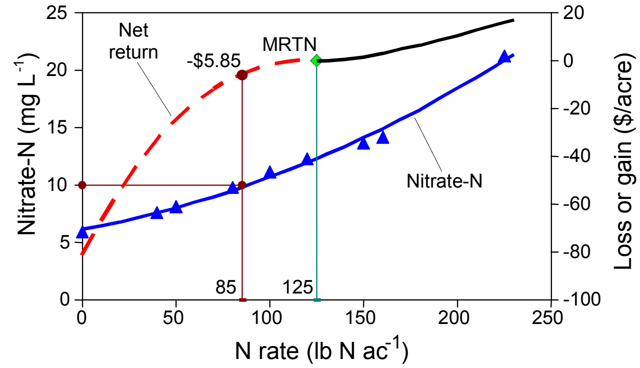


Figure 5-6. Tile-flow nitrate-N annual concentration average in a SC rotation from N rates applied in various years from 1990-2004 at the Gilmore City, Iowa, site (Lawlor et al., 2005) and the net economic gain or loss ($\$0.22$ lb^{-1} N : $\$2.20$ bu^{-1} corn) across N rates for SC in Iowa (Nafziger et al., 2004). The solid section of the net return line represents the gain if N rates are reduced to the maximum return to N (MRTN), and the dashed section represents the loss if N rates are reduced below the MRTN. The indicated economic loss of $5.85 ac^{-1} is for reduction of tile-flow nitrate-N from the MRTN rate to the N rate that results in approximately the 10 mg L^{-1} MCL drinking water standard.

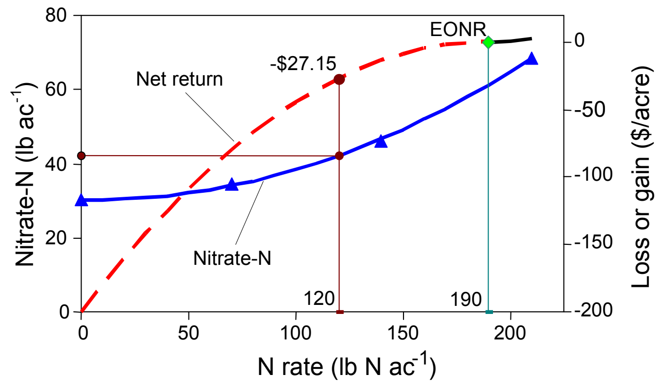


Figure 5-7. Rotation total tile-flow nitrate-N mass load and net economic gain or loss ($0.22 lb^{-1} N : $2.20 bu^{-1} corn) across spring-applied N rates in a SC rotation, average of 2002-2004 at the Filson, Illinois, site (Clover, 2005). The solid section of the net return line represents the gain if N rates are reduced to the site economic optimum N rate (EONR), and the dashed section represents the loss if N rates are reduced below the EONR. The indicated economic loss of $27.15 ac^{-1} is for reduction of tile-flow nitrate-N load from the EONR rate to the N rate that results in an approximate 30% lower load.

research at the site (Baker and Johnson, 1981) with two N rates of approximately 90 and 240 lb N ac^{-1} applied only to corn in a corn-soybean-corn-oat-soybean sequence resulted in an average annual 20 mg nitrate-N L^{-1} (24 lb nitrate-N ac^{-1} year^{-1}) with the low N rate and 40 mg nitrate-N L^{-1} (43 lb nitrate-N ac^{-1} year^{-1}) with the high N rate. Work by Andraski et al. (2000) at a site in Arlington, Wisconsin, with various crop rotations and manure history showed that the soil water nitrate-N concentration (measured in porous-cup samples at 48 in.) was 18 mg L^{-1} at the EONR, was <10 mg L^{-1} when N rates were more than 45 lb N ac^{-1} below the EONR, and was >20 mg L^{-1}

when N rates were more than 45 lb N ac^{-1} above the EONR. Work reported by Randall and Mulla (2001) with depleted ^{15}N ammonium sulfate applied to CC at Waseca, Minnesota, indicated a 17% increase in yield but a 30% higher nitrate-N loss in drainage water with 180 lb N ac^{-1} compared to 120 lb N ac^{-1}. Davis et al. (2000) reported that increasing N rates from 90 to 200 lb N ac^{-1} in CC (Waseca, Minnesota) resulted in a linear increase in nitrate-N loss (0.8 to 22.8 lb nitrate-N ac^{-1} $year^{-1}$). Jaynes et al. (2004) achieved a 30% reduction in nitrate-N concentration in water leaving a central Iowa sub-basin by changing the timing of N application from fall to split spring/sidedress and reducing the N input through use of soil N testing, but the weekly and annual average flow-weighted nitrate-N concentrations were not maintained below the 10 mg L^{-1} drinking water MCL.

If achieving the drinking water standard is a goal for nitrate concentrations in subsurface drainage, it will be difficult to achieve solely with application rate. However, if N is being applied well above rates that produce maximum economic return, then reduction in nitrate loss can be accomplished by reducing rates to those levels (examples in table 5-1 and figs. 5-6 and 5-7). The gain will depend on the specific location, rate change, and production situation.

Nitrate-N concentrations in subsurface drainage are generally greater for CC than for SC due to the frequency of annual N applications. This is especially true when N is over-applied. An over-application of 50 lb N ac^{-1} $year^{-1}$ in a CC system provides greater potential for much higher nitrate losses than an over-application of 50 lb N ac^{-1} every other year in a SC rotation. In addition, soybean can scavenge some of the excess residual N if spring drainage is limited. When N is being applied closer to optimal rates, differences in nitrate-N concentrations in the drainage water between CC and SC will be less and may be minimal. Because nitrate moves in drainage water after soybean harvest, this moderates differences in nitrate loss between the rotations. Data from the Nashua, Iowa, water quality site for 1990-1992 provide an excellent example. The average annual loss (across all tillage systems) was 30 mg nitrate-N L^{-1} (52 lb nitrate-N ac^{-1} $year^{-1}$) with CC and 18 mg nitrate-N L^{-1} (25 lb nitrate-N ac^{-1} $year^{-1}$) with SC, at N rates of 180 lb N ac^{-1} applied each year to corn in CC and 150 lb N ac^{-1} applied every other year to corn in SC (Weed and Kanwar, 1996; Kanwar et al., 1997). Continuing the study site from 1993-1998 with reduced N rates of 120 lb N ac^{-1} in CC and 100 lb N ac^{-1} in SC, the average annual loss was 11 mg nitrate-N L^{-1} (15 lb nitrate-N ac^{-1} $year^{-1}$) with CC and 11 mg nitrate-N L^{-1} (12 lb nitrate-N ac^{-1} $year^{-1}$) with SC (Bakhsh et al., 2005). Another example is the tile-flow data collected by Randall et al. (1997), in which N (based on spring soil sampling) applied in CC compared to SC increased average annual nitrate-N concentrations by approximately 8 mg L^{-1} (from 24 to 32 mg L^{-1}) and increased flux 7%.

While not directly comparing N rates, at a site in southeastern Indiana, Kladivko et al. (2004) found that, over time, decreasing the frequency of N application (moving away from CC to SC after nine years), decreasing the N rate (changing to the SC rotation and changing the N rate over time from an initial 250 to 160 lb N ac^{-1}), and growing a winter cover crop after corn in the SC rotation significantly reduced tile-flow nitrate. Over a 14-year period, the flow-weighted nitrate-N concentration was reduced from approximately 28 to 8 mg L^{-1}. Important characteristics that influenced nitrate-N concentrations and changes over time at the site included relatively shallow tile, low organic matter soil, drainage all winter, and spring-applied anhydrous ammonia fertilizer. Similar results were found in lysimeter studies in Ohio (Owens et al., 1995). When the cropping sequence was changed from CC with an N rate of 300 lb N ac^{-1} to SC with an N rate of 200 lb N ac^{-1} and a winter cover crop, annual flow-weighted nitrate-N concentrations were reduced from about 22 to 12 mg L^{-1}.

In summary, rate of N application and frequency of corn in the cropping sequence are important factors influencing nitrate losses in subsurface drainage. Since losses are greater in a CC system than in a SC system, largely due to annual versus every-other-year frequency of application, it is of greater importance to use the correct amount of N in the CC system than with a SC system if nitrate losses are to be minimized and maximum return to N achieved.

Nitrogen Rate Potential to Reduce Nitrate-N Losses

Since nitrate in subsurface drainage increases with increasing N application rate, there is potential to affect nitrate losses through change in N rate. However, the level of change will be related to the rate comparison and starting rate. In addition, and as mentioned above, the success relative to water quality goals is not likely to be achieved solely through rate adjustment. For instance, at economic optimum application rates for corn production, nitrate-N in tile flow typically exceeds the MCL drinking water standard (examples in table 5-1 and fig. 5-6). Moreover, even if no N is applied, nitrate-N will exceed the proposed EPA nutrient criteria for total N in surface waters (examples in Clover, 2005; Lawlor et al., 2005).

There are also questions regarding costs associated with reducing nitrate losses, and how those costs are to be paid. If N application rates being used are above MRTN rates, then producers can gain economically by reducing rates to those levels (figs. 5-6 and 5-7). They will achieve a net economic positive due to reduced N input and no associated loss in yield. However, if producers are already applying N at MRTN rates, then reduction below those rates will impose an economic penalty through yield loss (tables 5-1 and 5-2 and figs. 5-6 and 5-7). As an example (fig. 5-6), let's say the goal is to reduce tile-flow nitrate-N to 10 mg L^{-1} and the starting N rate is at the MRTN. At the MRTN rate for Iowa SC (125 lb N ac^{-1}), the associated tile-flow nitrate-N is approximately 12 mg L^{-1} (Lawlor et al., 2005). The N rate associated with 10 mg nitrate-N L^{-1} is 85 lb N ac^{-1}. The net economic loss due to an N rate reduction from 125 to 85 lb N ac^{-1} is \$5.85 ac^{-1}. In another example, where corn yield and tile-flow nitrate is more responsive to N application (fig. 5-7), moving from the site EONR of 190 lb N ac^{-1} to a 120 lb N ac^{-1} rate (an associated 30% reduction in tile-flow nitrate load from 61 to 42 lb nitrate-N ac^{-1}), the net economic loss is \$27.15 ac^{-1}.

Since yield response decreases with increasing N rate, the cost in yield penalty for reduced N input is less near the MRTN rate than near zero N. Therefore, cost per unit of nitrate-N reduction in drainage water becomes much larger as N rate declines below the MRTN and approaches zero (table 5-2 and fig. 5-7). For the Filson, Illinois, site, the first 70 lb N rate increment (from 210 to 140 lb N ac^{-1}) costs \$0.52 per unit of nitrate-N load reduction, but the last 70 lb N rate increment (from 70 lb N ac^{-1} to zero N) costs \$29.70 per unit of nitrate-N load reduction (table 5-2).

These examples illustrate the significant risk and economic constraints that face producers if they are asked to reduce N application to rates below maximum net return. If N rates in both examples given above were reduced to zero, then the economic losses would be \$81.75 ac^{-1} and \$200.10 ac^{-1}, both of which are unacceptable. These examples also clearly show that potential reduction in nitrate in subsurface drainage, and costs for potential reductions, varies significantly across the Corn Belt.

Summary

Nitrate in subsurface drainage is responsive to N application rate. Increasing the rate of N applied for corn results in greater nitrate concentrations in subsurface drainage water. While rates that produce maximum net economic gain through yield return to N will moderate nitrate-N, the resulting concentrations can approach but usually will be greater than acceptable in relation to the USEPA drinking water MCL standard, and definitely above proposed water quality criteria. Growing corn in rotation, for example every other year with soybean, reduces nitrate in subsurface drainage due to lower corn N fertilization requirement and less frequent application.

Economic and water quality gains can be achieved by reducing N rates if producers are applying N at rates above those needed for maximum net economic return. However, water quality gains achieved by reducing rates below those for maximum economic return will result in economic loss due to reduction in corn grain yield greater than that offset by N input reduction. If such restrictions are placed on N application rates as part of reaching a goal in regard to gulf hypoxia or local nitrate in surface waters, then it will be important to consider mechanisms to reimburse producers for lost income. It is also important to recognize that corn N fertilization requirements, potential for reducing nitrate concentrations in subsurface drainage, and costs for potential nitrate reductions vary significantly across the Corn Belt and must be accounted for in predictions of nitrate loss improvement and associated cost estimates when considering water quality driven changes in N inputs.

Interpretive Summary

Practice Recommended

- Apply N to corn at rates that produce maximum profit.

Important Factors

- Profitability for producers.
- In corn production systems, nitrate is lost in tile-flow drainage even if no fertilizer N is applied, often in the 3 to 10 mg nitrate-N L^{-1} range.
- Nitrate-N concentration in subsurface drainage generally increases in a continuous relationship with increasing N rate.
- Application of N above optimal rates reduces economic return and further increases nitrate losses.
- Optimal rates of N must account for previous crop and for N inputs from ma-

nure, ammoniated phosphate fertilizers, starter fertilizers, and N fertilizers applied in weed and feed herbicide applications.

- Preplant and in-season soil and plant diagnostic tests are decision aids that can improve N rates.
- The potential for reducing nitrate-N concentration or load in drainage water by changing N application rate should be evaluated relative to that at rates providing maximum economic return to N and for associated producer risks.
- Reducing N rates below optimum results in economic losses to the producer because the value of lost yield is not offset by reduced N costs.
- Nitrate losses are usually higher for continuous corn than for corn rotated with soybean, small grains, and alfalfa.

Limitations

- Even with application of no fertilizer N to corn, nitrate-N concentrations in subsurface drainage are above the currently proposed EPA nutrient ecoregion VI surface water quality criteria for total N.
- Application of N near rates that provide maximum economic return usually results in tile flow having nitrate-N concentrations above the EPA drinking water MCL, often in the range of 10 to 20 mg nitrate-N L^{-1} for SC and 15 to 30 mg nitrate-N L^{-1} for CC.
- In Iowa studies, to lower the nitrate concentration to 10 mg nitrate-N L^{-1} in tile drainage with a SC rotation, the N rate applied to corn had to be reduced by 40 lb N ac^{-1} below the rate providing maximum economic return; this reduction would have an associated net loss of \$5.85 ac^{-1}.
- In an Illinois study with a SC rotation, to reduce the total nitrate-N load by 30% (relative to that at optimal N application) in tile drainage, the N rate had to be reduced by 70 lb N ac^{-1} below the economic optimum rate, with an associated net loss of \$27.15 ac^{-1}.
- The "cost" (in yield loss) per unit of nitrate-N reduction in tile flow becomes much larger as N rates decrease below the optimum rate.
- As N rates are reduced below the maximum economic return rate, production variability and risk increase due to uncertainties in the N needs of corn for any given year and location.

Potential

- Nitrogen rate reduction will directly benefit producers when current application rates are above optimum. Reduction to optimal rates will also reduce nitrate losses. While there is uncertainty in the actual N application rate for corn in specific geographic areas, and hence the possible incidence of over-application, it can be projected that adjusting N rates from a 40 lb over-application down to economic optimal rates will decrease nitrate concentration in subsurface drainage water by about 20% to 25% from fields with such over-application.
- Optimal N rates for corn, associated nitrate levels in subsurface drainage, and the potential to gain improvement in nitrate losses through optimizing N rates varies across the Upper Mississippi River sub-basin and needs to be accounted

for in water quality programs addressing N application rates.

- Crop rotations that include fewer years with corn consequently reduce the frequency of application and the total N rate, resulting in lower nitrate concentrations in subsurface drainage.
- To achieve desired water quality goals, other in-field or out-of-field practices will need to be implemented, as change in N application rates or application at optimal rates to all corn production fields will not alone "solve" nitrate loss issues.

Future Research Needs

- More research using adequate N rate increments and concurrently measuring nitrate loss in subsurface drainage is needed to better quantify that relationship.
- Research is needed to provide a better understanding of reasons for variation in optimal N rates across the Upper Mississippi River sub-basin.
- Research on development and refinement of tools such as soil N tests, plant tests, and plant sensors is needed to determine more accurately fertilizer N needs and thus reduce risk of under- or over-fertilization.

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References

Andraski, T. W., L. G. Bundy, and K. R. Brye. 2000. Crop management and corn nitrogen rate effects on nitrate leaching. *J. Environ. Qual*. 29: 1095-1103.

Baker, J. L, K. L. Campbell, H. P. Johnson, and J. J. Hanway. 1975. Nitrate, phosphorus, and sulfate in subsurface drainage water. *J. Environ. Qual*. 4: 406-412.

Baker, J. L, and H. P. Johnson. 1981. Nitrate-nitrogen in tile drainage as affected by fertilization. *J. Environ. Qual*. 10: 519-522.

Bakhsh, A., R. S. Kanwar, and D. L. Karlen. 2005. Effects of liquid swine manure application on NO_3-N leaching losses to subsurface drainage water from loamy soils in Iowa. *Agric. Ecosyst. Environ*. 109: 118-128.

Clover, M. W. 2005. Impact of nitrogen management on corn grain yield and nitrogen loss on a tile drained field. MS thesis. Urbana, Ill.: University of Illinois.

Davis, D. M., P. H. Gowda, D. J. Mulla, and G. W. Randall. 2000. Modeling nitrate nitrogen leaching in response to nitrogen fertilizer rate and tile drain depth or spacing for southern Minnesota, USA. *J. Environ. Qual*. 29: 1568-1581.

Jaynes, D. B., T. S. Colvin, D. L. Karlen, C. A. Cambardella, and D. W. Meek. 2001. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. *J. Environ. Qual*. 30: 1305-1314.

Jaynes, D. B., D. L. Dinnes, D. W. Meek, D. L. Karlen, C. A. Cambardella, and T. S. Colvin. 2004. Using the late spring nitrate test to reduce nitrate loss within a watershed. *J. Environ. Qual*. 33: 669-677.

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.

Kladivko, E. J., J. R. Frankenberger, D. B. Jaynes, D. W. Meek, B. J. Jenkinson, and N. R. Fausey. 2004. Nitrate leaching to subsurface drains as affected by drain spacing and changes in crop production system. *J. Environ. Qual*. 33: 1803-1813.

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 subsurface drainage. ASAE Paper No. 052025. St. Joseph, Mich.: ASAE.

Nafziger, E. D., J. E. Sawyer, and R. G. Hoeft. 2004. Formulating N recommendations for corn in the corn belt using recent data. In *Proc. N.C. Ext.-Ind. Soil Fertility Conf.*, Vol. 20: 5-11. Brookings, S.D.: Potash and Phosphate Institute.

Owens, L. B., W. M. Edwards, and M. J. Shipitalo. 1995. Nitrate leaching through lysimeters in a corn-soybean rotation. *SSSA J*. 59: 902-907.

Owens, L. B., R. W. Malone, M. J. Shipitalo, W. M. Edwards, and J. V. Bonta. 2000. Lysimeter study of nitrate leaching from a corn-soybean rotation. *J. Environ. Qual*. 29: 467-474.

Randall, G. W., D. R. Huggins, M. P. Russelle, D. J. Fuchs, W. W. Nelson, and J. L. Anderson. 1997. Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems. *J. Environ. Qual*. 26: 1240-1247.

Randall, G. W., and D. J. Mulla. 2001. Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices. *J. Environ. Qual*. 30: 337-344.

Sawyer, J., E. Nafziger, G. Randall, L. Bundy, G. Rehm, and B. Joern. 2006. Concepts and rationale for regional nitrogen rate guidelines for corn. PM 2015. Ames, Iowa: Iowa State University Extension.

USEPA. 2002. Nutrient water quality criteria, summary table for the nutrient criteria documents. Washington, D.C.: U.S. Environmental Protection Agency. Available at: www.epa.gov/waterscience/criteria/nutrient/ecoregions/. Accessed 19 January 2007.

Weed, D. A. J., and R. S. Kanwar. 1996. Nitrate and water present in and flowing from root-zone soil. *J. Environ. Qual*. 25: 709-719.

Pp. 73-85 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop.* St. Joseph, Michigan: ASABE.

Nitrogen Application Timing, Forms, and Additives

6

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Wet, poorly drained soils throughout North America and Europe are often artificially drained with subsurface tile systems to remove excess (gravitational) water from the upper 1 to 1.2 m soil profile. Improved crop production that often results from drainage is in large part due to better physical conditions for field operations and a deeper unrestricted root zone for greater crop rooting, nutrient uptake, and yields. Removal of excess water by drainage lessens the potential for anaerobic conditions and consequently reduces the potential for nitrate to be lost from the soil profile by the process of denitrification. The combination of greater soil organic matter N mineralization with increased aerobic soil conditions, less N lost via denitrification, and increased transport of subsurface water results in higher nitrate concentrations in the receiving surface water bodies. Watersheds containing similar production systems and soils without subsurface drainage generate lower nitrate concentrations because anaerobic conditions exist more frequently. Under anaerobic conditions, denitrification predominates, resulting in nitrate losses as N gas to the atmosphere as well as economic losses to the farmer because of reduced available N.

Factors influencing nitrate content in subsurface waters draining from agricultural production landscapes can be divided into two categories: noncontrollable and controllable. Precipitation, including variation in annual amount, temporal distribution within a year, and extreme daily events, provides noncontrollable factors that have the greatest impact on nitrate loss. Controllable factors are those management practices that crop producers use to improve the yield and profitability of their enterprise. Time of N application, N fertilizer product, and nitrification inhibitors play a significant role in minimizing nitrate loss, especially under wetter and warmer fall, winter, and spring conditions (Dinnes et al., 2002).

Time of N Application

Agronomically and environmentally, spring applications are frequently superior to fall application because less loss of N occurs in the time between application and N uptake by the crop. However, many U.S. corn growers, especially in the northern part of the Corn Belt, desire to apply N in the fall because they usually have more available time and field conditions are more suitable for application. Early planting of corn as soon as the soils are tillable in the spring is desirable for highest yields and profit. Consequently, if a farmer wishes to separate spring N fertilizer application from preemergence herbicide application, the window of opportunity for spring N application

Table 6-1. Effect of N rate and time of application on nitrate-N losses to subsurface drainage and corn yield in Minnesota (adapted from Randall and Mulla, 2001).

N[a]		Annual Loss of	Five-Year Yield Average	
Rate ($lb\ ac^{-1}$)	Time	Nitrate-N in Drainage ($lb\ N\ ac^{-1}\ year^{-1}$)	Yield ($bu\ ac^{-1}$)	Net Return ($\$\ ac^{-1}$)
0	0	7	66	--
120	Fall	27	131	100
120	Spring	19	150	135
180	Fall	34	160	143
180	Spring	26	168	154

[a] Ammonium sulfate applied to continuous corn about 1 November or 1 May.

becomes very narrow (Randall and Schmitt, 1998). Risk of soil compaction and extended periods of rainy weather can also be deterrents to spring application of N.

In an extensive review of N application timing, Bundy (1986) concluded that fall N application is an acceptable option on medium to fine-textured soils where winter temperatures retard nitrification. However, under these conditions, fall-applied N is usually 10% to 15% less effective than spring-applied N. A recent Iowa study (Kyveryga et al., 2004) reported more rapid nitrification of fall-applied anhydrous ammonia in soils with pH >7.5, which influenced the amount of nitrate lost by denitrification or leaching during spring rainfall. They suggested that economic and environmental benefits of delaying application of fertilizer N may be greater on high pH soils than in lower pH soils. In Europe, N applied in autumn, either as mineral fertilizer (Goss et al., 1993) or as animal manure (Thompson et al., 1987) is very vulnerable to leaching in the winter.

Nitrogen was applied as ammonium sulfate in the fall (early November) and spring (late April) for continuous corn to determine the effect of N application time and rate on nitrate losses to subsurface drainage and corn yields on a Canisteo clay loam, glacial till soil in Minnesota (Randall and Mulla, 2001). Over the five-year study period, corn yields from the late fall application averaged 8% lower (146 vs. 159 bu ac^{-1} $year^{-1}$) than with spring application (table 6-1). Moreover, annual losses of nitrate-N in the tile drainage water averaged 36% higher (30 vs. 22 lb ac^{-1} $year^{-1}$) with fall application compared to spring application. It is interesting to note that less nitrate was lost in the drainage water for the 180 lb spring-applied treatment than for the 120 lb fall-applied treatment; yet greater yields (37 bu ac^{-1}) and net return ($54 ac^{-1}) were obtained for the spring treatment.

A long-term corn-soybean rotation study comparing late-October application of ammonia with and without N-Serve, and a spring preplant application without N-Serve showed distinct yield, economic, and environmental advantages for spring application, but not in all years (table 6-2). Across the 15-year period, corn yields averaged about 10 bu ac^{-1} greater for the fall N + N-Serve (nitrapyrin) and spring N treatments compared with fall N without N-Serve (Randall et al., 2003b; Randall and Vetsch, 2005b). In addition, compared with fall application of N without N-Serve, nitrate-N losses in the drainage water were reduced by 14% and 15% (Randall et al., 2003a; Randall and Vetsch, 2005a), economic return to N was increased by $9 and $19 ac^{-1}, and N recovery in the grain was increased by 8% and 9% for fall N + N-Serve and spring N, respectively. However, corn yields were significantly affected by the N treatments in only seven of 15 years. In those seven years, when April, May, and/or June were wetter than normal, average corn grain yield was increased by 15 and 27 bu ac^{-1} and average economic return was increased by \$22.50 and \$51.00 ac^{-1} for the fall N + N-Serve and spring N treatments, respectively. In summary, the 15-year data suggest that applications of ammonia in the late fall + N-Serve or in the spring preplant were better management practices. However, when spring conditions were wet, especially in May and June, spring application gave substantially greater yield and profit than the fall N + N-Serve treatment. Therefore, fall N + N-Serve application is considered to be economically more risky than a spring preplant application of ammonia.

Anhydrous ammonia applied at 110 lb N ac^{-1} without N-Serve in late October after soybean harvest was compared with ammonia applied midway between the rows in late April across four different tillage systems (no-till, strip-till, spring field cultivate, and chisel plow plus field cultivate) in 1997-1999 (Vetsch and Randall, 2004). Yields were not different between fall and spring-applied N in 1997 or 1998 (table 6-3). The effect of wet spring conditions was evident in 1999 when corn yields were 36 bu ac^{-1} lower for fall-applied N. An interaction between tillage system and time/placement of N was not found, indicating that the effect of fall vs. spring application was the same for all tillage systems in each year.

A four-year (2000-2003) study conducted on Nicollet, Webster, and Canisteo soils in Iowa found NO_3-N concentrations in subsurface drainage from a corn-soybean rotation to not be different between fall and spring application of aqua ammonia, either with or without N-Serve, under slightly dry to normal precipitation conditions (Lawlor et al., 2004) (table 6-4). Although timing and method of N application may be important, the authors concluded that applying the correct amount of N was perhaps the most important factor.

Split application of N should theoretically result in increased N efficiency and reduced nitrate losses because of greater synchronization between time of application and crop uptake. However, evidence in the literature to support this concept is mixed. Baker and Melvin (1994) reported losses of nitrate-N to be higher for split application compared to a spring preplant application with continuous corn. Losses with split application for the corn-soybean rotation were lower in the year of application but tended to be higher in the subsequent year when soybean followed corn. In another Iowa study, Bjorneberg et al. (1998) concluded that combining a split N fertilizer management strategy based on the pre-sidedress nitrate soil test (PSNT) with no-tillage practices can have positive environmental benefits without reducing corn yields in a corn-soybean rotation. Jaynes et al. (2004) reported nitrate reductions of 30% in drainage water in the last two years of a four-year Iowa study when the in-season N rate of a split-application strategy was determined by the late spring nitrate test (LSNT); however, the four-year average corn yield was slightly lower (3%) but not statistically different for the LSNT-based N rate compared to the non-limiting N rate (200 lb N ac^{-1}).

A split application of ammonia with 40% applied preplant (55 lb N ac^{-1}) and 60% applied sidedress (80 lb N ac^{-1}) at the V8 corn growth stage was compared with late October and spring preplant applications of ammonia (135 lb N ac^{-1}) (table 6-5). In

Table 6-2. Corn yield and economic return to N program as affected by time of anhydrous ammonia application and N-Serve at Waseca, 1987-2001 (adapted from Randall and Vetsch, 2005a, 2005b and Randall et al., 2003a, 2003b).[a]

	Time of Application		
Parameter	Fall	Fall + N-Serve	Spring
15-year avg. yield (bu ac^{-1})	144	153	156
15-year avg. economic return over fall N ($ ac^{-1} $year^{-1}$)[b]	--	$9.30	$18.80
7-year avg. yield (bu ac^{-1})[c]	131	146	158
7-year avg. economic return over fall N ($ ac^{-1} $year^{-1}$)[b]	--	$22.50	$51.00
15-year flow-weighted NO_3-N concentration in tile drainage from the corn-soybean rotation (mg L^{-1})	14.1	12.2	12.0
15-year N recovery in the corn grain (%)[d]	38	46	47

[a] Rate of N was 135 lb ac^{-1} $year^{-1}$ for 1987-1993 and 120 lb N ac^{-1} $year^{-1}$ for 1994-2001.
[b] Based on corn = $2.00 bu^{-1}, fall N = $0.25 lb^{-1} N, spring N = $0.275 lb^{-1} N, and N-Serve = $7.50 ac^{-1}.
[c] Only those seven years when a statistically significant yield difference occurred among treatments.
[d] N recovery = (N content in grain - N content in grain from 0 lb check) / fertilizer N rate.

Table 6-3. Corn yield as affected by time/placement of anhydrous ammonia at Waseca (adapted from Vetsch and Randall, 2004).

	Yield (bu ac^{-1})		Three-Year
Time/Placement	1997-1998	1999	Average
Fall, near row	188	145	174
Spring, between rows	188	181	186
LSD (0.10):	NS	5	3

Table 6-4. Average annual flow-weighted NO_3-N concentration in subsurface drainage from a corn-soybean rotation in Iowa as affected by time of N application, N-Serve, and N rate (2000-2003) (adapted from Lawlor et al., 2004).

Nitrogen Treatment			Four-Year Average
Time	Rate (lb N ac^{-1})	N-Serve	Flow-Weighted NO_3-N (mg L^{-1})
Fall	150	No	14.2
Fall	150	Yes	16.2
Fall	225	No	18.1
Spring	150	No	15.4
Spring	150	Yes	17.7
Spring	225	No	24.4
		LSD (0.05):	3.0

Table 6-5. Corn production and nitrate loss as affected by time of anhydrous application and N-Serve at Waseca, 1987-1993 (adapted from Randall et al., 2003a, 2003b).

Nitrogen Treatment		Seven-Year Average			Flow-Weighted
Time	N-Serve	Corn Yield (bu ac^{-1})	N Recovery[a] (%)	Economic Return to N[b] ($ ac^{-1})	NO_3-N Concentration in Tile Drainage[c] (mg L^{-1})
Fall	No	131	31	34	16.8
Fall	Yes	139	37	43	13.7
Spring	No	139	40	47	13.7
Split	No	145	44	56	14.6
LSD (0.10):		4			

[a] N recovery = (N content in grain - N content in grain from 0 lb check) / fertilizer N rate.
[b] Based on corn = $2.00 bu^{-1}, fall N = $0.25 lb^{-1}, spring N = $0.275 lb^{-1}, N-Serve = $7.50 ac^{-1}, and application cost = $4.00 ac^{-1} $time^{-1}$.
[c] Across the four-cycle corn (1990-1993) - soybean (1991-1994) rotation.

this seven-year period, grain yields were significantly greater (6 bu ac^{-1}) for the split-applied treatments, resulting in slightly greater N recovery in the grain and economic return to N compared to the fall and spring treatments (Randall et al., 2003b). However, flow-weighted nitrate-N concentration in the tile drainage across the four-cycle corn-soybean rotation (1990-1993) for the split N treatment was also slightly higher than for the spring N and fall N + N-Serve treatments (Randall et al., 2003a). Intuitively, one could rationalize suggesting lower rates of N when split-applied in a manner similar to this study. But to our knowledge, there are no other corn yield data that support the recommendation to reduce N rate below the preplant recommended rate in this production system. Perhaps the difference between an optimal single-application preplant N rate of ammonia and a split application rate is so small that field experiments cannot distinguish yield or water quality differences.

Split application is an N management strategy that will likely gain momentum in the next five to ten years. Growers are looking for combinations of preplant techniques (rates, sources, and placement methods) and sidedress techniques (in-season diagnostic tools to determine optimum N rate, time of application, and placement) that optimize N use efficiency (NUE), improve profitability, and minimize N losses. Localized placement of some N near the seed at planting has stimulated greater early corn growth and has resulted in positive yield responses, particularly in research conducted in very reduced tillage systems. Others are looking for the ideal proportion of preplant N vs. sidedress N to both optimize return on investment and/or to facilitate in-season diagnostic methods to determine optimum sidedress N rates. Remote sensing techniques, perhaps in conjunction with other diagnostic tools and/or climate models, may provide the necessary information to fine-tune in-season application techniques. These techniques would guide the application of spatially variable rates of N throughout the field and could help determine the optimum application window for sidedress application. At this time, these technologies appear to be much more feasible and dependable under irrigated conditions because the N can be applied with the irrigation water and moved down into the active root zone for quick uptake. Given the complex interactions between soils, weather, cropping systems, N sources, application equipment, etc., that affect the outcomes, research will continue to address these questions in an effort to determine those strategies with the greatest potential for providing economic and environmental success.

As the literature clearly indicates, however, sidedressing N does not necessarily reduce nitrate losses to drainage water. Nitrate losses in the drainage water are generally lower in the year of sidedress application unless fall rainfall is excessive, but due to greater potential carryover in the soil, nitrate tends to leach from the profile the following spring when precipitation exceeds evapotranspiration (ET) and soils are saturated. However, if the preplant or planting time N rate can be optimized in combination with applying a more precise sidedress rate, determined by in-season diagnostic methods, the total rate applied using this split N strategy should optimize NUE and profitability and may reduce nitrate losses below those found with current split-application strategies.

To estimate the extent of fall-applied N in the Corn Belt, state extension soil fertility specialists and state fertilizer associations were contacted to solicit estimates of the percent of each state's annual fertilizer N amount that is applied in the fall. The estimates are: Illinois = 25% to 30%, Indiana = 5% to 10%, Iowa = 25% to 30%, Michigan = <5%, Minnesota = 60% to 65%, Missouri = 15% to 20%, Ohio <5%, and Wisconsin = 10%. Total corn acreage in 2005 for these states was 12.1, 5.9, 12.8, 2.2, 7.3, 3.1, 3.4, and 3.8 million acres, respectively (NASS, 2005). Based on these data, an estimated 25% (12.9 million acres) of the 50.6 million acres of corn in this eight-state area receives N in the fall. States with the largest amount of fall-applied N are Minnesota (4.56 million acres), Iowa (3.52 million acres), and Illinois (3.28 million acres). Not only are these states major corn producers, they are also major contributors of nitrate to the Mississippi River. Thus, changing N application from the fall to spring or split applications could have a significant impact on nitrate loss in these three states, but may have limited impact in terms of the larger Mississippi River/Gulf of Mexico hypoxia issue.

Nitrification Inhibitors

Nitrification inhibitors are sometimes added to ammonium fertilizers (anhydrous ammonia and urea) to retard or slow the conversion of ammonium to nitrate after fertilizer application. N-Serve has been the most commonly used nitrification inhibitor in the U.S. and has been a component in many N research studies. The length of time that N-Serve remains active in the soil before it degrades largely determines its efficacy. The period of inhibition depends primarily on when N-Serve is applied, soil temperature, and soil pH. In Minnesota, when N-Serve is applied with anhydrous ammonia in late October (soil temperatures at the 6-inch depth average about 50°F and soils are frozen from early December through late March), inhibition activity continues into May. When N-Serve is applied in mid- to late April, inhibition can continue into June. Warm soil temperatures and high-pH soils speed the degradation process, thus shortening the inhibition activity period.

Many studies have shown that nitrification inhibitors, such as N-Serve, are effective in delaying conversion of ammonium to nitrate when N is fall-applied (Hoeft, 1984), but use of nitrification inhibitors with fall-applied N has not given consistent crop yield responses. Bundy (1986) concluded that nitrification inhibitors can improve the effectiveness of fall-applied N, but spring N is more effective than fall N applied with an inhibitor when conditions favoring N loss from fall application develop.

Anhydrous ammonia was applied at a rate of 135 lb N ac^{-1} in four treatments [late fall, late fall + N-Serve, spring preplant, and split (40% preplant + 60% sidedress)] to drainage plots in Minnesota from 1987 through 1993. Subsurface tile drainage did not occur in 1987 through 1989 due to very dry conditions. Flow-weighted nitrate-N concentrations across the four-year corn-soybean rotation flow period (1990-1993) averaged 16.8, 13.7, 13.7, and 14.6 mg L^{-1} for the four treatments, respectively (table 6-5). Yields were increased significantly in the very wet years by the addition of N-Serve to the fall application.

Table 6-6. Corn grain yield as affected by fall and spring application of N-Serve with anhydrous ammonia at Waseca, 1994-1999 (adapted from Randall and Vetsch, 2005b).

Time of Application	Six-Year Average Yield (bu ac^{-1})	
	With N-Serve	Without N-Serve
Fall	161	171
Spring	172	176

A six-year study comparing fall vs. spring application of N-Serve with ammonia (120 lb N ac^{-1}) showed a statistically and economically significant 10 bu ac^{-1} yield response to N-Serve applied in the fall (table 6-6). The 4 bu ac^{-1} yield increase to spring-applied N-Serve was not statistically significant and was considered economically neutral (Randall and Vetsch, 2005b). However, a yield response to spring-applied N-Serve occurred in years when June rainfall was excessive. Because the above data do not suggest a consistently significant and economical response to N-Serve applied in the spring, and because excessive June rainfall cannot be predicted at the time of spring ammonia application, adding N-Serve to spring-applied ammonia is not considered to be an effective practice in Minnesota.

The interaction between time of N application and N-Serve in the above study was significant for nitrate-N concentration in the drainage water in three of six years during the corn phase and in two of six years during the soybean phase. Annual nitrate-N concentrations were reduced 2 to 4 mg L^{-1} when N-Serve was added to fall-applied N but were increased 1 to 3 mg L^{-1} when N-Serve was added to spring-applied N. These increased concentrations of nitrate-N in the drainage water with spring-applied N-Serve are similar to the results with split-applied N (spring + sidedress) shown in table 6-5.

N-Serve added to spring-applied urea for continuous corn in Ohio reduced nitrate losses in drainage water from lysimeters (Owens, 1987). A three-year drainage study in Illinois showed significant differences among fall, spring, and sidedress application of N to corn on tile flow, nitrate-N concentration, and loss in corn and in soybean the following year (R. G. Hoeft, personal communication, 2005). However, the addition of N-Serve to fall-applied N did not affect either nitrate-N concentration or loss in the drainage water or corn yield.

Response to N-Serve appears to be particularly dependent on time of N application. Quesada et al. (2000) reported the agronomic and economic effects of N-Serve applied with ammonia in the spring during a ten-year period in Iowa. Grain yield responses occurred with N-Serve in one year for continuous corn but did not occur for corn in rotation with soybean. The Minnesota data for N-Serve shown in tables 6-2 and 6-4 suggest that applying N-Serve with anhydrous ammonia in late October when soil temperatures are at or below 50°F is economically beneficial on the Canisteo and associated glacial till soils. Corn yields were increased by 9 bu ac^{-1} and net economic return was increased by \$9.30 ac^{-1}. Moreover, NO_3-N losses in tile drainage water were reduced by 14%. These data further suggest that N-Serve applied with ammonia in the spring would not likely be beneficial in reducing nitrate losses to tile drainage or in increasing yields and profitability.

N Source and Time of Application

The N source used must also be considered when selecting the proper time of application. Studies on a Webster clay loam in Minnesota in 1981 and 1982 compared fall application of anhydrous ammonia and urea at 75 and 150 lb N ac^{-1}, with and without N-Serve, to spring application of the same products and N rates. Two-year average second-year corn yields shown in table 6-7 indicate: (1) broadcast and incorporated urea was inferior to anhydrous ammonia when fall-applied, and (2) spring application of urea was superior to fall application. Although no nitrate loss data were collected in this study, it is quite likely that nitrate losses into drainage water from fall-applied urea would be similar to those from fall-applied ammonium sulfate shown in table 6-1.

A subsequent study on Nicollet and Webster glacial till soils in southern Minnesota compared late-October application 100 lb N ac^{-1} of urea (4 in. deep band) and anhydrous ammonia with and without N-Serve to spring preplant urea and anhydrous ammonia. Three-year average yields show advantages for spring application of 33 bu ac^{-1} for urea and 14 bu ac^{-1} for ammonia (table 6-8). Nitrogen recovery in the corn plant ranked: spring ammonia = spring urea > fall ammonia > fall urea. The effect of N-Serve in this study was minimal. Yield responses to the spring treatments were greatest in 1998, when April and May were warm and late May was wet, and in 1999 when the fall of 1998 was warm and April and May of 1999 were very wet. Significant yield differences were not found in 1997 when the fall of 1996 was cold and the spring of 1997 was cool and dry.

Similar findings for fall-applied urea have been observed in a long-term Iowa study (A. P. Mallarino, personal communication, 2005). Corn yields averaged across 17 years

Table 6-7. Corn yield as influenced by N source, time of application, and N-Serve at Waseca, 1981-1982 (unpublished data).

Nitrogen Treatment		Yield (bu ac^{-1})	
Source	N-Serve	Fall Application	Spring Application
None	--	104	
Urea	No	157	164
Urea	Yes	155	167
Anhydrous ammonia	No	162	168
Anhydrous ammonia	Yes	170	173

Table 6-8. Corn yield and N recovery in the whole plant as influenced by time of application and N source at Waseca, 1997-1999 (unpublished data).

Nitrogen Management			Three-Year Average	
			Yield	N Recovery[a]
Time	Source	N-Serve	(bu ac^{-1})	(%)
--	None	--	112	
Fall	Urea	No	152	43
Fall	Urea	Yes	158	47
Fall	Anhydrous ammonia	No	168	60
Fall	Anhydrous ammonia	Yes	170	63
Spring preplant	Urea	No	185	76
Spring preplant	Anhydrous ammonia	No	182	84
		LSD (0.10):	8	

[a] N recovery = (N content in grain - N content in grain from 0 lb check) / fertilizer N rate.

for the 240 lb N rate were 13 bu ac^{-1} greater with urea when applied in the spring compared with the fall. In the last four years, the yield advantage for spring-applied urea was 16 bu ac^{-1}. Moreover, the 160 lb spring rate yielded 10 bu ac^{-1} more than the 240 lb fall rate.

Controlled-release N fertilizers such as ESN produced by Agrium, where a polymer coating on each urea granule controls the release of urea to the surrounding soil matrix, and slow-release N fertilizers have potential for generating greater corn yields and reduced losses of nitrate compared with urea, especially in situations where N loss potential is high (sandy soils, plentiful spring rainfall, fall application, etc.). The authors are not aware of any published research on these new, developing N sources illustrating their effect on corn yields and nitrate losses to drainage water in the Midwest. Because of their potential to increase NUE, research is needed in this area.

Although we have not discussed manure applications in this chapter (see chapter 8), approaches to making application timing decisions should be similar to those with N fertilizers. In general, animal manures with high levels of first-year N availability (i.e., a high ratio of ammonium N to organic N) should be spring-applied for best NUE and lowest potential for nitrate loss. Manures with a greater organic N content and lower first-year N availability can be fall-applied with less potential for crop yield or nitrate loss. Results from a four-year study in Minnesota showed no difference in nitrate losses to subsurface drainage from late fall-applied dairy manure slurry compared with spring-applied urea when applied at the same rate of estimated crop-available N for continuous corn (Randall et al., 2000). Adding N-Serve to manure slurry can be quite expensive (a label rate of 2 qt N-Serve ac^{-1}), and yield results generally have not supported the practice. Manure applied after corn for soybean is not thought to cause increased nitrate losses if the application rate is appropriate (50 bu ac^{-1} soybeans take up about 200 lb N ac^{-1}) and the manure is applied late in the fall or in the spring.

As the U.S. depends on more off-shore produced fertilizer N because of higher U.S. natural gas prices and older, less efficient production facilities, we can expect to see: (1) a shift away from anhydrous ammonia to urea and urea-ammonium nitrate (UAN) solution, and (2) higher prices for N. Because urea and UAN are considered to be agronomically less dependable than ammonia under moderate to high-loss-potential conditions, improved management strategies must be employed to gain greater NUE and profitability with these N sources. This provides opportunities for comprehensive research programs supporting improved N management, particularly on split application of N, controlled/slow release N fertilizers, nitrification inhibitors, remote sensing to assess in-season plant N status for prediction of supplemental N needs, and N-efficient hybrid genetic traits.

Overall Conclusion

This chapter, summarizing much of the published research, clearly shows that best management practices (BMPs) for application timing, N forms, and additives such as N-Serve can reduce nitrate losses to subsurface drainage water. But two questions need to be asked: (1) Will BMPs be quickly and universally implemented, especially in those areas where N losses associated with these practices are most prevalent? And

(2) Is the nitrate reduction with BMPs of significant magnitude to accomplish society's goals? History has indicated that BMP implementation can be slow unless incentive and/or disincentives are offered. The current U.S. Farm Bill does little to encourage adoption of these BMPs. Furthermore, the data suggest that BMPs will not reduce nitrate losses to the level needed/expected on a regional basis, and perhaps not even on a local basis. Thus, in addition to these BMPs, farm policy changes leading to longer crop rotations and diversification involving legumes and perennials (resulting in N source reduction) coupled with landscape modification, i.e., strategically placed wetlands and cover crop establishment, will be needed in the Upper Mississippi River basin to meet society's goals for reducing nitrate losses to water resources.

Interpretation/Extrapolation Summary

Time of N Application

Site conditions: Warm and wet conditions in the spring (April-June) in the northern regions or late fall and spring (March-May) in the central to southern regions are conducive to substantial loss of fall-applied N. Losses by denitrification and/or leaching range from nil under dry conditions to more than 50% under very wet conditions.

Research findings: Spring application of N is superior to fall application in most cases. Under "very limited or no" N loss conditions, differences between fall and spring application are not significant on medium to fine-textured soils. No clear or consistent evidence shows split or sidedress applications to be superior to spring preplant anhydrous ammonia from a water quality or corn yield perspective on medium and fine-textured Corn Belt soils. For UAN, split application (preplant and sidedress) is desirable as it reduces the risk of loss when conditions are wet prior to the V10 corn growth stage. Data showing this are limited, however.

Water quality improvement: Minnesota data suggest an average 15% reduction of leaching loss in drainage water with spring application of ammonia compared to a late-October application when soil temperatures are at or below 50°F. Nitrate losses in drainage water from fall-applied N throughout the Corn Belt could range between nil to 25% depending on time of fall application (early vs. late), fall and winter soil temperatures, and spring rainfall. Benefits of spring and split applications of N would be greatest in Minnesota, Iowa, and Illinois where the extent of fall-applied N is largest.

Cost: On a pound-for-pound product basis, spring-applied N may cost up to \$5 to \$10 ac^{-1} $year^{-1}$ more than fall N. However, spring-applied N rates may be able to be reduced without a yield penalty compared to N rates applied in the fall. With spring application, the N rate should not be adjusted downward to achieve a cost savings if the N rate recommendations are based on calibration data from spring and split applications of N.

Extent of area: We estimate that 25% (12.9 million acres) of the 50.6 million acres of corn in the Corn Belt presently receives fall N. All of those acres could benefit from spring or split applications of N.

Limitations for adoption of spring N: The current mindset or tradition of fall anhydrous ammonia application among growers and suppliers will be slow to change in the absence of incentive or disincentive programs. Supplier infrastructure, although this is currently changing, will cause spring supply and storage issues and will require equipment changes and substantial capital.

Impact on other resources: Incorporation of broadcast urea and UAN to limit volatilization or surface runoff losses could enhance soil erosion (negative impact). Crop yields will likely become less variable, thus reducing the potential for lower yields and profitability (positive impact).

Research needs: Determine the optimum combination of preplant and sidedress N applications for greatest yield, practicability, and economic return, and lowest nitrate losses. Determine whether lower N rates can be used with split-application technologies to maintain yield and reduce nitrate losses below those for preplant N application. Evaluate the role of in-season diagnostic tools on improving the efficacy of sidedress applications and improving N use efficiency. Develop models and decision aids by monitoring in-season climatic factors (daily temperature and precipitation) and characterizing soil properties thoroughly within each of the above research efforts.

Nitrification Inhibitors

Site conditions: Conditions affecting the effectiveness of nitrification inhibitors for reducing nitrate losses are essentially the same as those for "time of application."

Water quality improvement: Minnesota data obtained on calcareous, poorly drained, glacial till soils suggest an average nitrate leaching loss reduction of 14% when N-Serve is used with anhydrous ammonia in late October compared to not using N-Serve in the fall. Leaching losses were not influenced by spring application of N-Serve. Nitrate leaching losses were not affected by fall-applied N-Serve on well drained soils in Minnesota or in the Illinois and Iowa studies.

Cost: Annual cost of \$7.50 ac^{-1} for a reduction of 3.5 lb nitrate-N ac^{-1} (range is 0 to 9 lb nitrate-N ac^{-1}).

Extent of area: Percent of corn acres in the Corn Belt that could benefit from fall N-Serve is maybe 15% at the most, depending on when fall application occurs. This percentage will decline as anhydrous ammonia loses market share. Use of N-Serve with urea and UAN is unlikely.

Limitation for adoption: Barriers include old chemistry and inconsistent, weather-related results, as well as the extra cost. New inhibitors and controlled-release forms of urea are needed that reduce nitrate loss, reliably supply crop-available N, and are inexpensive.

Impact on other resources: Nitrification inhibitors do not affect other resources. Crop yields may be improved if the inhibitor reduces nitrate losses, but yields are not reduced by use of an inhibitor.

Research needs: Evaluate efficacy of new inhibitors and slow-release products for both corn production and environmental purposes.

Source of N

Current situation: Urea and urea-ammonium nitrate solution (UAN) are gaining a greater portion of market share at the expense of anhydrous ammonia. These forms of

N are most suitable for spring and in-season application, thereby facilitating the conversion from fall application to spring application.

Research findings: Urea and UAN are acceptable sources of N for optimum crop production when spring preplant-applied and split-applied. Fall-applied urea has performed poorly.

Water quality improvement: Water quality is generally not affected by fertilizer N source as long as the N is applied using best management practices. However, specific situations involving large rainfall/leaching events shortly after N application could result in greater nitrate losses from UAN than from ammonia or urea due to the nitrate component of UAN.

Cost: Costs among the fertilizer N sources will vary depending on season, dealership, demand, supply, etc. The price difference among sources generally ranges from $0.05 to $0.10 per pound, with UAN being most expensive and anhydrous ammonia the cheapest. However, combining spring UAN application with pre-emergence herbicide application reduces fuel consumption by eliminating one field pass per growing season and can aid in "burndown" herbicide efficacy.

Extent of area: No limitation other than supplier's source inventory.

Limitations for adoptions: Two primary limitations exist. From the supplier's perspective, the distribution system and storage will present significant challenges. Substituting urea and UAN for ammonia will result in a huge volume change. From the grower's and supplier's perspectives, application equipment is a limitation. Distribution infrastructure, storage facilities, and application equipment will need to be purchased, requiring significant additional expense to overcome these limitations.

Impact on other resources: Increased erosion potential associated with the incorporation of urea containing fertilizers. Agrotain, a urease inhibitor, could be added to urea to greatly reduce volatilization of the surface-applied and non-incorporated N, but this would add an extra cost.

Research needs: Evaluate controlled-release and slow-release fertilizers and their impact on the economic and environmental aspects of corn production. Determine the effect of various livestock manures and their rate and time of application on nitrate losses.

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References

Baker, J. L. and S. W. Melvin. 1994. Chemical management, status and findings. In *Agricultural Drainage Well Research and Demonstration Project - Annual Report and Project Summary*, 27-60. Des Moines, Iowa: Iowa Department of Agriculture and Land Stewardship and Iowa State University.

Bjorneberg, D. L., D. L. Karlen, R. S. Kanwar, and C. A. Cambardella. 1998. Alternative N fertilizer management strategies effect on subsurface drain effluent and N uptake. *Applied Eng. in Agric*. 14(5): 469-473.

Bundy, L. G. 1986. Review: Timing nitrogen applications to maximize fertilizer efficiency and crop response in conventional corn production. *J. Fert. Issues* 3: 99-106.

Dinnes, D. L., D. L. Karlen, D. B. Jaynes, T. C. Kaspar, J. L. Hatfield, T. L. Colvin, and C. A. Cambardella. 2002. Nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. *Agron. J.* 94: 153-171.

Goss, M. J., K. R. Howse, P. W. Lane, D. G. Christian, and G. L. Harris. 1993. Losses of nitrate-nitrogen in water draining from under autumn-sown crops established by direct drilling or mouldboard ploughing. *J. Soil Sci*. 44: 35-48.

Hoeft, R. G. 1984. Current status of nitrification inhibitor use in U.S. agriculture. In *Nitrogen in Crop Production*, 561-570. R. D. Hauck, ed. Madison, Wisc.: ASA, CSSA, and SSSA.

Jaynes, D. B., D. L. Dinnes, D. W. Meek, D. L. Karlen, C. A. Cambardella, and T. S. Colvin. 2004. Using the late spring nitrate test to reduce nitrate loss within a watershed. *J. Environ. Qual*. 33: 669-677.

Kyveryga, P. M., A. M. Blackmer, J. W. Ellsworth, and R. Isla. 2004. Soil pH effects on nitrification of fall-applied anhydrous ammonia. *SSSA J*. 68: 545-551.

Lawlor, P. A., J. L. Baker, S. W. Melvin, and M. J. Helmers. 2004. Nitrification inhibitor and nitrogen application timing effects on yields and nitrate-N concentrations in subsurface drainage from a corn-soybean rotation. ASAE Paper No. 042273. St. Joseph, Mich.: ASAE.

NASS. 2005. Available at: www.nass.usda.gov. Washington, D.C.: USDA National Agricultural Statistics Service.

Owens, L. B. 1987. Nitrate leaching losses from monolith lysimeters as influenced by nitrapyrin. *J. Environ. Qual*. 16: 34-38.

Quesada, J. P., R. Killorn, and A. M. Dierdickx. 2000. Response of corn grown in two crop rotations to different N rates and nitrapyrin. In *Agronomy Abstracts*, 274. Madison, Wisc.: ASA.

Randall, G. W., and M. A. Schmitt. 1998. Advisability of fall-applying nitrogen. In *Proc. 1998 Wisconsin Fertilizer, Aglime, and Pest Management Conf*., 90-96. Madison, Wisc.: University of Wisconsin, Department of Soil Science.

Randall, G. W., and D. J. Mulla. 2001. Nitrate-N in surface waters as influenced by climatic conditions and agricultural practices. *J. Environ. Qual*. 30: 337-344.

Randall, G. W., and J. A. Vetsch. 2005a. Nitrate losses in subsurface drainage from a corn-soybean rotation as affected by fall vs. spring application of nitrogen and nitrapyrin. *J. Environ. Qual*. 34(2): 590-597.

Randall, G. W., and J. A. Vetsch. 2005b. Corn production on a subsurface-drained mollisol as affected by fall versus spring application of nitrogen and nitrapyrin. *Agron. J.* 97: 472-478.

Randall, G. W., T. K. Iragavarapu, and M. A. Schmitt. 2000. Nutrient losses in subsurface drainage water from dairy manure and urea applied for corn. *J. Environ. Qual*. 29: 1244-1252.

Randall, G. W., J. A. Vetsch, and J. R. Huffman. 2003a. Nitrate losses in subsurface drainage from a corn-soybean rotation as affected by time of nitrogen application and use of nitrapyrin. *J. Environ. Qual*. 32: 1764-1772.

Randall, G. W., J. A. Vetsch, and J. R. Huffman. 2003b. Corn production on a subsurface-drained mollisol as affected by time of nitrogen application and nitrapyrin. *Agron. J.* 95: 1213-1219.

Thompson, R. B., J. C. Ryden, and D. R. Lockyer. 1987. Fate of nitrogen in cattle slurry following surface application or injection to grassland. *J. Soil Sci*. 38: 689-700.

Vetsch, J. A., and G. W. Randall. 2004. Corn production as affected by nitrogen application timing and tillage. *Agron. J.* 96: 502-509.

Pp. 87-104 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop*. St. Joseph, Michigan: ASABE.

7

Agronomic and Environmental Implications of Phosphorus Management Practices

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Phosphorus (P) is an essential nutrient for growth of plants and aquatic organisms. Fertilizer or manure P application to land is often necessary to achieve or maintain optimal levels of crop production. However, P applications seldom are needed in high-testing soils, and excessive applications can result in increased P delivery to water resources. Excessive nutrient levels can result in eutrophication of surface freshwater resources. The movement of P from agricultural land to water bodies is a complex process involving several source factors, transport factors, and multiple delivery pathways. Phosphorus moves into surface water attached to particulate matter eroded from the land and as dissolved P in surface runoff or subsurface tile drainage. It can also move into groundwater mainly as dissolved P.

Widespread animal production in the Upper Mississippi River watershed, mainly in the Corn Belt, results in significant manure application to many agricultural fields. Applications of fertilizer or manure P in excess of P removal with crop harvest have resulted in sharp soil-test P (STP) increases in many areas of the region during the last few decades (PPI, 2001). Recently applied P is particularly prone to loss, and the loss is affected by factors such as the form of P applied, the time since application, the placement method, and precipitation events soon after application. The factors contributing to P loss from agricultural land to surface waters are commonly grouped as source factors (site and management) and transport factors. This chapter focuses on selected source factors related to fertilizer and manure P management relevant to both crop production and risk of increased P delivery from fields to water resources.

Phosphorus in Soils and Sources

Soils of the north central region of the U.S. typically contain 300 to 1000 ppm of total P. This range results from differences in long-term soil forming processes and management practices. Only a small portion of the total P is readily available to plants. A small fraction of soil P is dissolved in the soil solution in the orthophosphate form, which is the form taken up by plants. As the plant depletes orthophosphate in the soil solution, dissolved P is replenished from P in a soil pool (sometimes referred to as labile P) in which P is held by a variety of relatively weak bonds to mineral particles and organic matter. The majority of soil P is in a stable pool (sometimes referred to as non-labile P) in which it is strongly held to mineral particles or is combined in mineral compounds of low solubility, mainly iron (Fe) and aluminum (Al) phosphates in acid or weathered soils and calcium (Ca) phosphates in calcareous soils, and in recalcitrant organic compounds. Stable P is considered unavailable to plants in the short term, although it becomes available over time at a very slow rate. The degree and strength to which P is bound in soils are largely determined by the amount and types of Al, Ca, and Fe compounds present and by other soil properties such as pH, organic matter, clay mineralogy, and the amount of P currently present in the soil.

Most P fertilizers used in crop production are composed of water-soluble P compounds. The most commonly used granulated fertilizers are ammonium phosphates, while fluid P fertilizers may include ammonium phosphates, potassium phosphate, and ammonium polyphosphates. There is little use of acidulated calcium phosphates (superphosphates), which once were the most common sources of commercial P fertilizers. The total P content and P forms in manure applied to fields vary greatly with the species, animal age, diet, and storage method. Some manures may have up to 80 to 100 lb P_2O_5 per ton (some poultry manures, for example), whereas others may contain 5 to 10 lb P_2O_5 per ton or less (such as liquid swine manure from lagoons or solid cattle manure). The proportion of organic, inorganic, and immediately soluble P in manure also varies greatly. For example, more than 80% of the P of liquid swine manure is in inorganic and soluble forms, while the rest is present as organic P. Therefore, liquid swine manure P reactions and P availability to plants in soils is near that of fertilizer P. On the other hand, solid manure from beef and dairy cattle can have less than 50% inorganic P, with the rest in relatively more stable organic forms. Estimates of manure P that becomes available to the first crop after application range from 60% to 100% in the north central region (J. Peters et al., NCR-13 Regional Soil Testing and Plant Analysis Committee, unpublished, 2004).

Application of P fertilizer and some manures (mainly poultry and liquid swine manures) causes a fast and sharp increase in soluble P in the soil at the point of application. Chemical equilibrium is rapidly reestablished as much of the added P is adsorbed to soil particles or precipitates as compounds of lower water solubility. The increase in soil-soluble P is less evident and more gradual over time for other manures. Over time, some of the P in the weakly retained soil P pools is converted into more stable mineral and organic forms. Therefore, the immediate result of fertilization and manure P applications is to increase the capacity of the labile P pool to replenish solution and stable soil P pools. The net long-term result depends on several soil chemical and mineralogical properties, P removal by crops, P movement to deeper soil layers, and P loss with soil erosion, surface runoff, or subsurface drainage.

Water-extractable manure P is not a good indicator of P available to a crop. The more labile inorganic and organic P forms in manure can become readily available for crops shortly after being in contact with soil, and the soluble P may be retained to a different degree by soil constituents. Reducing the total P concentration of animal manures would effectively reduce the amount of P applied to fields and should reduce the risk of P loss. Use of phytase enzyme to increase the digestibility of phytate P in swine and poultry rations is becoming common in large feeding operations. This practice can reduce total P in manure by up to 40% when mineral P supplementation is reduced accordingly. Phytase addition to diets does not consistently affect the proportion of soluble P in manure (Baxter et al., 2003; Angel et al., 2005).

Soil-Test Phosphorus Levels for Crop Production

Soil P tests have been developed based on knowledge of the chemical forms in which P exists in soils and empirical work to assess how the tests correlate with crop growth and P uptake by crops in the field. Interpreting a soil-test value requires an understanding of the impacts of the extractant, method of soil sampling, and sample handling on the test result and also of the intended use for the result. Accurate interpretations of soil-test results and appropriate fertilizer recommendations require that the relationship between the amount of a nutrient measured by a given soil test and the crop response to the added nutrient be known through field calibrations for different crops, soils, and growing conditions.

Soil P tests for agronomic use employ dilute strong or weak acids, complexing ions, and/or buffered alkaline solutions. The Bray P-1 and Mehlich-3 tests, and the Olsen test to a lesser degree, are routinely used in the north central region. The Bray P-1 test was developed for use in the acid to neutral soils of the region, and the Olsen (or sodium bicarbonate) test was developed primarily for use on calcareous soils. Regional research has shown that the Bray P-1 test is not reliable in many calcareous soils. The Mehlich-3 extractant is being rapidly adopted in the region because it is suitable for a wider range of pH and soil properties than the other tests and also can be used for extraction of other nutrients. With the exception of calcareous soils, these tests are highly correlated, but the actual quantities measured can differ greatly. For example, in acid or near-neutral soils, the Olsen test usually measures 50% to 70% of the P measured by the Bray P-1 and the colorimetric version of the Mehlich-3 test (Mallarino, 1997). The inductively coupled (ICP) method of determining extracted P results in more measured P than the traditional colorimetric method, and different interpretations are needed for the Mehlich-3 extractant when these two determination methods are used (Mallarino, 2003a). As an example, figure 7-1 shows relationships between the relative yield increase of corn and STP measured by various soil tests across several Iowa soils. The response curve is used to divide STP levels into five categories: very low, low, medium or optimum, high, and very high or excessive. In general, there are only small differences across states of the region regarding recommended optimal STP levels for similar crops grown on relatively similar soils of Illinois (Hoeft and Peck, 2001), Iowa (Sawyer et al., 2002), Minnesota (Rehm et al., 2001), and Wisconsin (Kelling et al., 1998).

A thorough understanding of crop response to P and factors such as sampling date, sampling depth, and both method and time of nutrient application is needed to interpret STP results correctly and to provide P application recommendations. Other factors (such as climate, plant population, levels of other nutrients, and crop cultivars) often influence crop growth, P uptake and removal with harvest, and the P application rate needed to maintain optimal crop production. However, these factors usually have little impact on the optimal STP levels for crops. The influence of these factors is often very important for nutrients that are highly mobile in the soil (such as the nitrate form of N) but less important for nutrients with stronger retention by the soil (such as P).

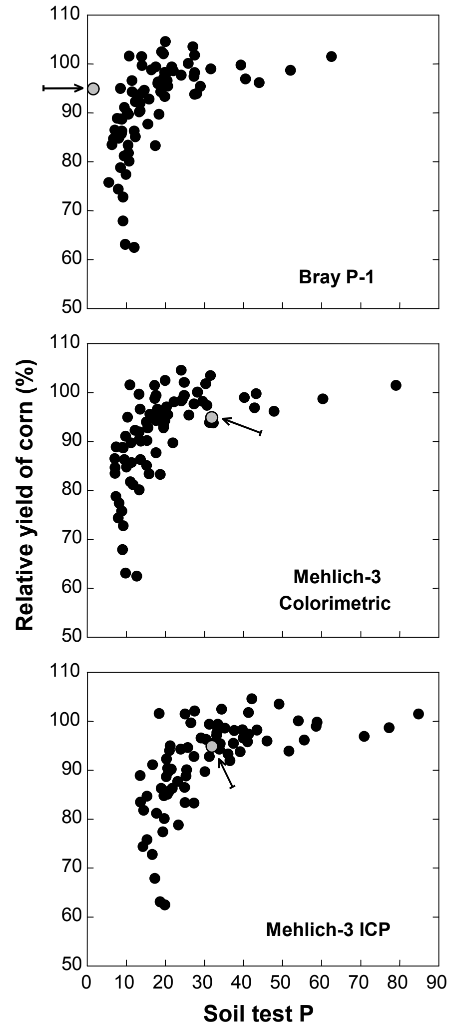

Figure 7-1. Example of the relationship between corn yield and soil-test P measured by three P tests commonly used in the Upper Mississippi River region (adapted from Mallarino, 2003a). The gray points and arrows indicate results for a highly calcareous soil.

Soil Sampling for Phosphorus for Agronomic Purposes

For a soil testing program to be effective, besides proper soil-test calibration and laboratory quality control, soil samples should be collected in a cost-effective manner, should accurately represent the nutrient level in the area of interest, and the sampling depth should be the same as the depth used for developing the soil-test calibrations. Sampling is a critical component of the soil-testing process because it usually represents the largest single source of error in soil testing. Many factors that vary both spatially and temporally influence nutrient concentrations in soils.

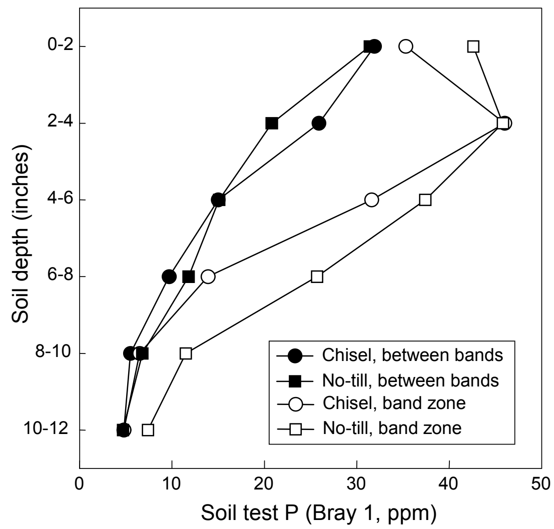

Figure 7-2. Mean soil-test P across five sites after several years of no-till or chisel-plow tillage and deep-band P fertilization using similar application rates for band and inter-band zones (adapted from Mallarino and Borges, 2006).

Soil-test P variation with depth results from a combination of soil-forming and management factors and the differential mobility of nutrients in soils. Nutrients such as P with relatively low mobility tend to accumulate near the application point or zone of crop residue accumulation. The tillage system and the application method greatly influence vertical and lateral P stratification. The significant vertical stratification of P in pastures and no-tilled soils is well known. However, as shown in figure 7-2, vertical and lateral stratification also exists in fields managed with chisel-plow tillage, and subsurface banding can significantly reduce STP concentration near the soil surface. Therefore, the consistent proper depth for soil sampling is an important consideration. Soil samples should be collected from the soil depth that results in the soil-test values best correlated with nutrient sufficiency for crops, which can be known only through soil-test field calibration research. Soil samples for estimating plant-available P and other nutrients with relatively low mobility in soil often are collected from the top 6 to 8 inches of soil. Although shallower soil sampling for P sometimes is recommended in some parts of the country for soils managed with no-tillage or pastures, the available field calibration research (or the lack of it) in the northern region does not justify establishing differential sampling depth recommendations at this time. However, a shallower sampling depth for pastures often is justified and recommended for other reasons (such as to determine lime needs).

Variation in landscape position and soil parent material can cause large changes in soil texture, organic matter, drainage, and other properties over a field and can result in large spatial (lateral) STP variability. These properties may affect STP directly through their influence on the amount of plant-available P or indirectly through crop yield and P removal with harvest. Variability caused by long-term history of manure or fertilizer application and other soil or crop management practices overlays the variability associated with soil-formation factors. Proximity to livestock confinement areas, feed storage areas, and field boundaries are additional examples of historical factors causing large variability in many fields. Small-scale variability usually predominates in fields with long histories of cropping and fertilizer or manure applications, especially when nutrients are applied in bands. The challenge in these situations is to determine cost-effective methods to delineate sampling areas within a field and the number of cores needed for each composite sample to account for small-scale variability.

A variety of systematic and zone sampling approaches have been developed to measure STP. The development of affordable global positioning technology, geographic information systems, and variable-rate application equipment has led to widespread use of site-specific soil sampling approaches in the region. These approaches typically are used to generate a soil fertility map to serve as an input to computer-controlled equipment for applying varying rates of one or more materials. One such approach is zone sampling, by which field areas with more homogeneous properties than the field as a whole are delineated. Differences in landscape position, soil mapping unit, remotely sensed soil and crop canopy properties (such as soil color, soil electrical conductivity, and canopy color or growth patterns), and grain yield measured with yield monitors are examples of factors often used to define management zones. Another approach involves systematic grid sampling, where soil-test patterns in a field are determined by means of a dense and systematic sample collection. A grid size of approximately 2.5 acres is common in the region. Small-scale variability of P is so high in some fields that accurate within-field soil fertility mapping is practically and economically impossible. Many producers and crop consultants believe that the cost of dense grid sampling can be reduced by taking only a few cores for each composite sample, and often take as few as four to five cores per sample. However, research in the region demonstrates that at least 10 to 15 cores per sample should be collected from most fields independently of the sampling approach used in order to have reasonable confidence in the soil-test results. Much uncertainty still exists regarding how to best perform site-specific soil sampling and generate accurate soil fertility maps.

Phosphorus Application Methods

Phosphorus applications can be tailored to match crop needs and minimize excessive soil P accumulation by use of soil testing and estimates of P removal with harvest. Phosphorus removal by crops varies greatly among species and with the plant part harvested, and extension services of most states provide tables with average values. Soils with naturally high STP levels are rare in the north central region, and most high-testing soils result from historical P applications in excess of crop removal. Several long-term experiments have been used to provide recommendations. For example, in Iowa (Mallarino et al., 1991; Webb et al., 1992; Dodd and Mallarino, 2005) and Minnesota (Randall et al., 1997), long-term research showed that annual fertilizer P rates of 30 to 50 lb P_2O_5 ac^{-1} $year^{-1}$ maintained near-optimum STP levels (16 to 20 ppm as Bray P-1) and corn-soybean grain yields. This long-term research also shows that additional P application for row-crop production may not be needed for 10 to 15 years in soils that have STP four to five times higher than optimal levels for crops, except for small starter fertilizer rates in some conditions.

The timing of P application relative to planting a crop is not a critical issue for the predominant crops and soils of the region. This is because P has relatively low mobility in soils and the soils of the region have low to moderate capacity for retaining added P in unavailable forms. Therefore, P can be applied at planting time or in advance (weeks or months) of planting without a significant loss of efficiency. Several studies in Iowa (J. R. Webb and A. P. Mallarino, unpublished) and Minnesota (Randall et al., 1997) have shown that annual or bi-annual P applications for corn-soybean rotations have approximately similar efficiencies. Moreover, similar efficiencies of broadcast and band fertilizer P for no-till crops in Iowa have been partly explained by broadcast P application several months (in the fall) before planting (Bordoli and Mallarino, 1998; Borges and Mallarino, 2000). In addition, manure P application in advance of planting time may increase the efficiency of applied P with any tillage system because of usually slow P release from organic P forms.

Fertilizer placement options for crops have been evaluated for many years in the north central region. Theoretical reasons suggest increased efficiency of banded P in some conditions compared with the ubiquitous broadcast application. These include reduced P retention by soil constituents in forms unavailable to plants (which involve processes independent of plants or plant growth) and increased plant P uptake through a variety of processes as a result of placing a fertilizer band in the root zone. Reviews by Randall and Hoeft (1988) and Bundy et al. (2005) provide excellent summaries of published research. Much effort has focused on corn. Although placement options exist for other crops, the area planted is smaller, banding generally is not used or recommended (such as for soybean), or surface broadcast application is the only practical approach (such as for forages) for applying P, unless fertilizer is incorporated into the soil before crop establishment. Reviews of early work indicate that grain crop responses to P placement are less frequent at high STP levels than at low STP levels. At low STP levels and low P application rates, planter-band applications (mainly applied 2 inches beside and 2 inches below the seeds) usually maximize corn response to P compared to the broadcast placement method when the rates are similar. Research since the early 1990s has placed more emphasis on deep banding, in-furrow starter N-P-K or N-P fertilization, and surface-band fertilizer applications.

The placement of P or K fertilizer below the depth typically achieved with broadcast or planter-band application has been evaluated as a method of avoiding reduced nutrient availability due to stratification, particularly in no-till and ridge-till systems. While substantial evidence of nutrient stratification exists (e.g., Robbins and Voss, 1991; Rehm et al., 1995; Randall et al, 2001; Mallarino and Borges, 2006), reports of significant detrimental effects on crop yield are few. Early work by Farber and Fixen (1986) compared broadcast, deep band, fall-applied surface strip, and planter-band P applications for late-planted corn and found that the “2 by 2 inch” planter-band application was superior to the other placement options across three tillage systems. Work in Iowa has shown no advantage of deep P placement. A comparison of deep-band P

(5 to 7 inches deep) with broadcast and planter-band P (placed 2 inches besides and below the seed) placements for corn and soybean managed with no-till (Bordoli and Mallarino, 1998; Borges and Mallarino, 2000) and ridge-till (Borges and Mallarino, 2001; Borges and Mallarino, 2003) tillage systems showed no differences among the placement alternatives for various soils and STP ranges, although deep-band K often was better for both crops. However, deep P banding reduces P accumulation at or near the soil surface (fig. 7-2), and similar results have been observed for injected liquid swine manure.

Starter fertilization involves low rates of nutrient mixtures placed near or in the seed furrow with the planters and is commonly used in corn production. Most starter fertilizers contain N, P, and K, and mechanisms of crop response to starter are not always clear. Research in the north central region has shown frequent corn response to starter N, P, and K (Ritchie et al., 1996; Scharf, 1999; Lamond et al., 2001; Bermudez and Mallarino, 2002; Mallarino, 2003b; Niehues et al., 2004). Many researchers, such as Vetsch and Randall (2002), concluded that responses to N-P-K starter mixtures were not due to a consistent response to a single nutrient. Iowa work with no-till corn after soybean in high-testing soils (Mallarino, 2003b) showed that N explained the response to starter fertilizer in the three responsive sites of a total of eight fields. In the responsive fields, the primary N rate (110 to 160 lb N ac^{-1}) was injected across all treatments at the V5-V6 corn growth stage. Recent research (Niehues et al., 2004) suggests that response to sulfur (S) may also partly explain response to S-containing starter mixtures in the region.

Starter fertilization often increases corn yield in low-testing soils because crops respond to nutrient addition regardless of the placement method. However, at higher soil fertility levels, the response to starter is less frequent, and probably due to a placement effect that enhances early plant growth or helps overcome occasional limitations to early nutrient uptake imposed by the management system or climate. Table 7-1 shows a summary of results of experiments in the region. Some of these experiments and others (Lamond, et al., 2001; Niehues et al., 2004) reported responses to starter mixtures in soils testing low to high in P and/or K. However, Bermudez and Mallarino (2002), Mallarino (2003b), and Kaiser et al. (2005) found no significant response to starter P or K in Iowa high-testing soils when the starter was applied in addition to recommended broadcast P-K rates for corn-soybean rotations. Research indicates a greater likelihood of response to starter for continuous corn than for corn after soybean

Table 7-1. Frequency and size of no-till corn yield response to N-P-K or N-P starter fertilizer in several states of the Upper Mississippi River region (adapted and update from Bundy et al., 2005).[a]

Location	Reference	Response Frequency	Response (bu ac^{-1})
Illinois	Ritchie et al. (1996)	8 of 9 trials	14 average
Iowa	Buah et al. (1999)	7 of 9 trials	4 to 18
Iowa	Bermudez and Mallarino (2002)	5 of 7 trials	2 to 8
Iowa	Mallarino (2003b)	3 of 8 trials	5 average
Iowa	Kaiser et al. (2005)	1 of 2	15
Missouri	Scharf (1999)	6 of 6 trials	13 average
Wisconsin	Bundy and Widen (1992)	8 of 12 trials	15 average

[a] Soils tested medium, optimum, or higher in P and K according to local interpretations.

in environments with a short growing period where an acceleration of plant growth can translate into higher yield (Farber and Fixen, 1986; Bundy and Widen, 1992; Bundy and Andraski, 1999) and for some corn hybrids than for others.

The rate and placement of starter fertilizers can influence their performance. Higher starter rates are needed to optimize production in low-testing soils than in other soils when the starter is the sole nutrient source (Kaiser et al., 2005). Work with seed-placed starter indicates that application rates must be limited to avoid seedling damage and reduced plant populations. Nitrogen and K rather than P are the rate-limiting factors, and recommendations for seed placement typically indicate that the N plus K_2O in the fertilizer should not exceed 10 lb ac^{-1} of these nutrients. However, the safe application rate is highly affected by soil moisture content and the source of N and K. Because of these limitations, use of in-furrow N-P-K or N-P fertilizers often does not provide enough P to maximize crop response in low-testing soils.

Studies in the region suggest that manure application does not influence corn response to starter fertilizer strongly or consistently. Factors such as the importance of rapid early-season growth in realizing yield potential, soil drainage, and possibly soil-test level may influence response in manured systems. Motavalli et al. (1989) evaluated starter fertilization for corn silage on a soil with excessively high STP in northern Wisconsin. The starter increased yield in one of three years, but there was no interaction between manure and starter fertilization. Bundy and Andraski (1999) found that manure application did not significantly influence starter response on high-testing Wisconsin soils.

Surface-band P fertilizer applications usually have been evaluated as a starter fertilizer placement option. Teare and Wright (1990) found that a surface band of an N-P fertilizer increased yield across a range of corn hybrids. Surface band or dribble starter treatments were not as effective as seed or side-placed placements in Illinois studies (Ritchie et al, 1996). However, Lamond et al. (2001) and Niehues et al. (2004) found that surface dribble treatments produced similar yield response to banded starter, or differences were small and inconsistent. Little is known about potential implications of applying these small fertilizer rates to the soil surface at or near planting time for P loss with surface runoff.

Precision agriculture technologies available to producers or custom fertilizer and manure applicators facilitate application of P at rates adequate for different parts of a field. Dense-grid soil sampling from many fields of the Midwest has shown very large within-field spatial variability of STP. Variable-rate application of fertilizer P is common, and some custom applicators are beginning to apply manure at variable rates. Research in Illinois (Anderson and Bullock, 1998) and Iowa (Wittry and Mallarino, 2004) has shown that grid or zone soil sampling methods combined with variable-rate application based on STP often do not increase crop yield compared with traditional methods. Mallarino and Schepers (2005) suggested that use of current P fertilizer recommendations that encourage STP build-up in low-testing soils combined with very high small-scale STP variation may explain the lack of yield response differences between uniform and variable-rate fertilization methods. However, Iowa research showed that application according to spatial variability minimizes P application to high-testing areas and reduces STP variability within fields (fig. 7-3).

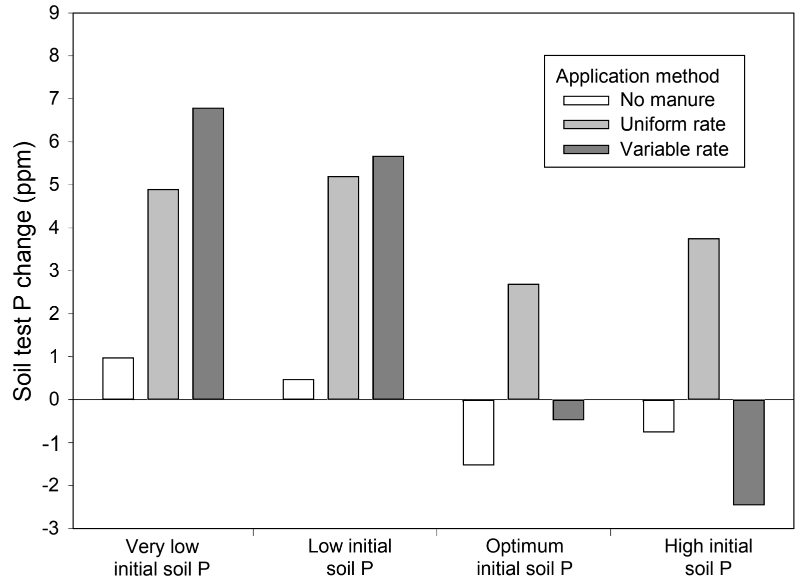

Figure 7-3. Effect of uniform application and soil-test P based variable-rate application of liquid swine manure on soil test P change within a field for various initial soil-test P interpretation classes (adapted from Mallarino and Schepers, 2005).

Environmental Implications of Phosphorus Management for Crop Production

Most P management practices discussed above have implications in relation to risk of P delivery from agricultural fields to water resources and environmental P management. Phosphorus delivery from fields depends on complex interactions between source and transport factors that are considered in P indices developed by most states of the region. In this section, we briefly discuss the most relevant issues of source factors. Source factors that affect P delivery to surface waters include soil P level and management practices such as the time and method of P application, although tillage practice and cropping system are also often considered source factors.

Soil-Test P Level and Sampling Depth

The potential for dissolved and particulate P loss through soil erosion, surface runoff, and subsurface drainage increase as soil P increases. Soil P is one of the factors useful to assess risk of P delivery to surface water. It may be measured by agronomic soil tests such as Bray P-1, Olsen, and colorimetric or ICP versions of the Mehlich-3 and also by environmental soil P tests that measure water-extractable P or presumed algal-available P (such as the Fe-oxide impregnated filter paper test, or estimated soil P saturation). The results of the agronomic and environmental P tests are generally well correlated in the north central region (Atia and Mallarino, 2002; Andraski and Bundy, 2003). Several studies have found that concentrations of dissolved, bioavailable, and particulate P in runoff increase linearly as STP increases (Andraski and Bundy, 2003; Allen et al., 2006). In some cases, however, P concentration in runoff may increase more rapidly at very high STP levels compared with lower levels (Klatt et al., 2002). Another consideration is that the total P concentration in sediment is higher than in the eroded soil; this P enrichment occurs due to selective removal during the erosion process of organic material and fine soil particles that are higher in P than the average for the soil. Studies of relationships between various STP and P loss with subsurface drainage show little P increase in water until a certain STP value (usually referred to as change point), after which P loss usually increases linearly. A study with subsurface tile drainage systems at three Iowa locations (Klatt et al., 2002) indicated a change point of approximately 60 ppm by the Olsen test or 100 ppm by the Bray P-1 test (6 inch sampling depth), which is four to five times larger than optimal STP levels for most crops of the region.

Ideally, soil samples collected for environmental purposes should reflect the depth of the soil-water mixing zone that contributes to P loss. Phosphorus accumulation at or near the application point is well known, and unless the P is incorporated into the soil, fertilization or manure application results in high P levels in the mixing zone of soil and runoff, especially for no-till and forage fields. This affects soil-test results and has implications for P loss in runoff. Tillage and deep P banding reduces STP stratification, but significant stratification exists with the use of implements such as chisel plows and field cultivators (fig. 7-2). Interpretation of agronomic soil tests is generally based on a sampling depth of 6 to 8 inches. Research in the region has shown inconsistent results concerning the benefit of a shallower sampling depth for prediction of both crop yield response to P and dissolved P loss with surface runoff in stratified no-till and pasture fields. Although a shallow sampling depth sometimes improves relationships between STP and runoff P, often differences are very small (Andraski and Bundy, 2003; Vadas et al., 2005). These results, together with practical complications of implementing different sampling depths in production agriculture, have resulted in the use of agronomic tests and sampling depths for P loss assessments. Furthermore, Wisconsin research has shown that reasonable predictions of STP stratification are possible for the purpose of assessing risk of P loss (Bundy and Good, 2004).

Soil-Test P Spatial Distribution

Spatial variability of soil P within a field needs to be considered in assessing risk of P loss. High concentrations of P in some field areas, mainly because of uneven manure application, can strongly affect soil test results. Sites of old farmsteads often have high STP as well. In pastures, grazing animals tend to deposit more manure near feeding areas, shaded areas, water sources, and fences and gates, resulting in relatively high soil P levels in these areas. Global position systems and variable-rate fertilization provide an opportunity to apply P only where it is needed within a field and to reduce STP variation. Although use of this technology usually does not increase yield significantly or consistently, Iowa research showed that application according to STP minimizes P application to high-testing areas and reduces STP variability within fields. Moreover, Mallarino (2003c) showed that variable-rate P application could be practically implemented based on P index ratings for field zones, not just based on STP. There are also variable runoff and erosion areas based on soils and topography.

Phosphorus Source and Application Timing and Method

An increase of the P application rate often increases risk of P loss independently of the STP level. In fact, research based on simulated rainfall shows no relationship between runoff P and STP for runoff events immediately after P application. Water passing over the soil surface interacting with recently applied manure or fertilizer P is highly concentrated in P, much of it as dissolved P. The concentration of runoff P shortly after application usually increases linearly as the rate of P application increases, although exponential increases are possible, and incorporation of the P into the soil tends to reduce P concentrations (figs. 7-4 and 7-5). Although the timing of P application may not have a major impact for crop production in the region, it can greatly impact P loss from fields in various ways. The risk of recently applied P loss is higher when the application is made in periods of high probability of intense rainfall, to water-saturated or snow-covered soil, to sloping ground, and to flood-prone areas. Time also influences risk of P loss in another way. Iowa research (fig. 7-4) shows that a runoff event 10 to 15 days after fertilizer or manure application can reduce total and dissolved P concentrations in runoff to less than 50% as compared to rain within 24 hours of manure application to soil having corn or soybean harvest residues and that is not incorporated (results not shown for P loads indicated approximately similar trends). Ongoing research indicates that this effect varies for different manures and is higher for liquid swine manure. Added P that reacts with the soil is less prone to losses with surface runoff. Therefore, when the P is not injected or incorporated, applying P during periods with a low probability of runoff events can substantially reduce the risk of P loss. The probability of runoff in this region is typically greatest in late winter and spring, a period that includes snowmelt, high rainfall, and little soil cover.

Research is showing inconsistent differences between fertilizer and liquid swine manure sources concerning P loss with surface runoff after surface applications. While some research suggests that water-extractable soluble manure P may be a good indicator of short-term P loss potential when manure is applied to the soil surface (Kleinman et al., 2002; Maguire et al., 2005), other research in the north central region indicates this is not be the case especially across several manure types and field conditions (Haq et al., 2006). Moreover, water-extractable manure P is not a good indicator of long-term P loss. Daverede et al. (2004) showed slightly larger runoff P concentrations for liquid swine manure than fertilizer (fig. 7-5). Ginting et al. (1998) showed that total P loss from plots receiving beef manure was either similar or lower than from plots receiving no manure. In addition, Bundy et al. (2001) showed that total P load in runoff from simulated rainfall was significantly lower where dairy manure was surface applied than in a control treatment where manure was not applied. Recent Iowa research (Haq et al., 2006) showed that P loss during runoff events shortly after surface application of various P sources was highest for fertilizer, lowest for poultry and beef manure, and intermediate for liquid swine manure. Several factors, some not well identified at this time, can explain this result. Manure typically has less inorganic and soluble P than fertilizers, adds organic matter (sometimes it includes bedding), and often in-

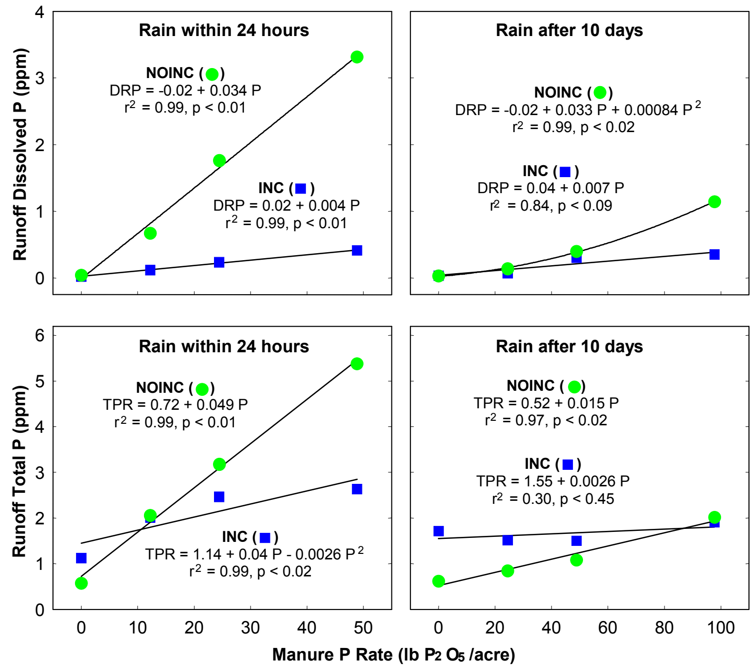

Figure 7-4. Effect of liquid swine manure incorporated (INC) or not incorporated (NOINC) into the soil and time of simulated rainfall on dissolved reactive and total P concentrations in runoff (B. L. Allen, A. P. Mallarino, and J. L. Baker, Iowa State University, unpublished).

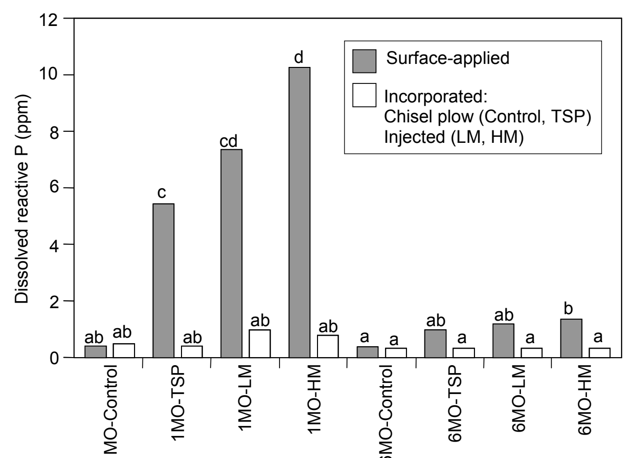

Figure 7-5. Mean dissolved reactive phosphorus (DRP) concentration in runoff as affected by time of rainfall simulation one month (1MO) and six months (6MO) after P application; P source (control, TSP = triple superphosphate, LM = low swine manure rate, and HM = high swine manure rate); and application method (surface-applied and incorporated, where the control and TSP were chisel-plowed and LM and HM were injected). The P rates applied were 110 lb P_2O_5 ac^{-1} for TSP, 68 to 80 lb P_2O_5 ac^{-1} for LM, and 135 to 161 lb P_2O_5 ac^{-1} for HM. Values that have the same letters are not significantly different (*P* < 0.1). Adapted from Daverede et al. (2004).

creases water infiltration, all of which can result in different runoff volume and runoff P concentration after application compared with fertilizer.

We mentioned above that applying P annually or once every two years is similarly effective for most crops of the region, as long as the application rates are similar. However, a biannual application system increases the instantaneous application rate. Because research usually shows a linear relationship between P rate and P loss with runoff, high and more widely spaced P application strategies may increase P loss. However, there is little evidence that applying the same amount of P in infrequent applications at higher rates with care and appropriate methods results in more long-term potential for P runoff loss than annual applications with proportionally lower rates of application. Moreover, infrequent N-based applications of manure benefit farmers, as this strategy allows them to meet the full N need of crops such as corn grown in rotation and reduce the need for supplemental N fertilization.

Incorporation of applied P, deep banding of P fertilizer, or injection of liquid manure generally reduce the rate of P build-up near the soil surface and both short-term and long-term risk of P loss with surface runoff. However, runoff P loss may not be reduced when the P incorporation into the soil involves tillage or the aggressive injectors often used to apply liquid manure. The increased soil erosion risk associated with the incorporation or injection of manure or fertilizer needs to be considered. On highly erodible land, the P rate and the degree of soil and crop residue disturbance by application or tillage equipment largely determines the option of least risk. These concerns emphasize the need for a comprehensive tool, such as the P index, that considers both source and transport factors to assess risk of P loss from fields.

Interpretive Summary

Practices recommended: Apply P fertilizer rates that optimize crop yield based on soil testing and crop P removal. When applying N-based manure, use P index ratings as a planning tool to avoid excessive soil P build-up, and choose methods and timing of application that reduce the risk of P loss with surface runoff and subsurface drainage.

Important considerations: Soil P testing is an imperfect tool, but it is very useful to guide P application for crop production. The soil sampling and testing methods recommended in the region for crop production generally are adequate for environmental P management. However, soil-test P interpretation classes used for crop production are not appropriate for environmental P management.

Optimization of manure nutrients use (especially N) and farm profitability may result in soil-test P build-up in animal and corn production systems. Use of the P index is a valuable tool to avoid excessive soil P build-up and risk of P loss.

Subsurface P placement methods reduce P accumulation at or near the soil surface and have potential to reduce P loss in sites with high risk of erosion and surface runoff compared with surface application methods. However, they are more costly and seldom increase crop yield further than other methods, and their impact on reducing P loss is dependant on their effect on increasing risk of erosion and the probability of a runoff event shortly after application. Therefore, guidelines for their use should be flexible to avoid economic penalties to producers.

Limitations: Even with P application rates determined according to soil testing and crop removal and without exceeding optimal soil-test P levels for crops, rates of P delivery to surface water resources may be unacceptably high in conditions with very high erosion and surface runoff.

Cost-effective soil sampling methods for crop production may not appropriately describe spatial soil-test P variability to reduce P loss in critical field areas.

Limiting manure application to P-based rates limits manure N use and may reduce farm profitability.

There is limited knowledge of the interactions between P source (fertilizer and manure types) and both timing and method of application for short-term and long-term P loss with runoff.

Potential: Fertilizer P rate reduction or elimination directly benefits crop producers when soil-test P values are above optimum levels for crops and reduces the potential for P loss.

Elimination of fertilizer and manure P application during periods of snow-covered or frozen sloping soil reduces the potential for P loss.

Application of N-based manure rates for crops in conjunction with the P index is a reasonable way of encouraging utilizing manure nutrients while reducing the risk of soil P build-up and P loss from fields.

Additional information needed: More research is needed to evaluate impacts of P placement methods on both short-term and long-term P loss from fields and on soil erosion rates. Additional needs include:

Better understanding of the impact on P loss of the time between surface P application and a runoff event and of the probability of a runoff event shortly after application.

Research on the effect of the proportion of soluble P in animal manures and P loss with surface runoff shortly after a surface application.

Research to further develop and learn use of cost-effective tools for assessing within-field variation of soil-test P and applying P more accurately to reduce risk of under- or over-fertilization.

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References

Allen, B. L., A. P. Mallarino, J. G. Klatt, J. L. Baker, and M. Camara. 2006. Soil and surface runoff phosphorus relationships for five typical USA Midwest soils. *J. Environ. Qual*. 35: 599-610.

Anderson, L. L., and D. G. Bullock. 1998. Variable-rate fertilizer application for corn and soybean. *J. Plant Nutr*. 21: 1355-1361.

Andraski, T. W., and L. G. Bundy. 2003. Relationships between phosphorus levels in soil and in runoff from corn production systems. *J. Environ. Qual*. 32: 310-316.

Angel, C. R., W. J. Powers, T. J. Applegate, N. M. Tamim, and M. C. Christman 2005. Influence of phytase on water-soluble phosphorus in poultry and swine manure. *J. Environ. Qual*. 34: 563-571.

Atia, A. M., and A. P. Mallarino. 2002. Agronomic and environmental phosphorus testing for soils receiving swine manure. *SSSA J*. 66: 1696-1705.

Baxter, C. A., B. C. Joern, D. Ragland, J. S. Sands, and O. Adeola. 2003. Phytase, high-available-phosphorus corn, and storage effects on phosphorus levels in pig excreta. *J. Environ. Qual*. 32: 1481-1489.

Bermudez, M., and A. P. Mallarino. 2002. Yield and early growth responses to starter fertilizer in no-till corn. *Agron. J*. 94: 1024-1033.

Bordoli, J. M., and A. P. Mallarino. 1998. Deep and shallow banding of phosphorus and potassium as alternatives to broadcast fertilization for no-till corn. *Agron. J*. 90: 27-33.

Borges, R., and A. P Mallarino. 2000. Grain yield, early growth, and nutrient uptake of no-till soybean as affected by phosphorus and potassium placement. *Agron. J*. 92: 380-388.

Borges, R., and A. P. Mallarino. 2001. Deep banding phosphorus and potassium fertilizers for corn managed with ridge tillage. *SSSA J.* 65: 376-384.

Borges, R., and A. P. Mallarino. 2003. Broadcast and deep-band placement of phosphorus and potassium for soybean managed with ridge tillage. *SSSA J*. 67: 1920-1927.

Buah, S. S. J., T. A. Polito, and R. Killorn. 1999. No-tillage corn hybrids response to starter fertilizer. *J. Prod. Agric*. 12: 676-680.

Bundy, L. G., and T. W. Andraski. 1999. Site-specific factors affecting corn response to starter fertilizer. *J. Prod. Agric*. 12: 664-671.

Bundy, L. G., and L. W. Good. 2004. The Wisconsin phosphorus index. Madison, Wisc.: University of Wisconsin, Department of Soil Science. Available at: http://wpindex.soils.wisc.edu. Accessed 12 Feb. 2007.

Bundy, L. G., and P. C. Widen. 1992. Corn response to starter fertilizer: Planting date and tillage effects. *Better Crops* 76: 20-23.

Bundy, L. G., T. W. Andraski, and J. M. Powell. 2001. Management practice effects on phosphorus losses in runoff in corn production systems. *J. Environ. Qual*. 30: 1822-1828.

Bundy, L. G., H. Tunney, and A. D. Halvorson. 2005. Agronomic aspects of phosphorus management. In *Phosphorus: Agriculture and the Environment*, 685-727. ASA Monograph Series No. 46. A. Sharpley and T. Sims, eds. Madison, Wisc.: ASA-CSSA-SSSA.

Daverede, I. C., A. N. Kravchenko, R. G. Hoeft, E. D. Nafziger, D. G. Bullock, J. J. Warren, and L. C. Gonzini. 2004. Phosphorus runoff from incorporated and surface-applied liquid swine manure and phosphorus fertilizer. *J. Environ. Qual*. 33: 1535-1544.

Dodd, J. R., and A. P. Mallarino. 2005. Soil-test phosphorus and crop grain yield responses to long-term phosphorus fertilization strategies for corn-soybean rotations. *SSSA J*. 69: 1118-1128.

Farber, B. G., and P. E. Fixen. 1986. Phosphorus response of late planted corn in three tillage systems. *J. Fert. Issues* 3: 46-51.

Ginting, D., J. F. Moncrief, S. C. Gupta, and S. D. Evans. 1998. Interaction between manure and tillage system on phosphorus uptake and runoff losses. *J. Environ. Qual*. 27: 1403-1410.

Haq, M. U., A. P. Mallarino, and B. L. Allen. 2006. Runoff phosphorus loss immediately after surface application of fertilizer and swine, poultry, or beef manure. *Agron. Abs.* CD-ROM. Madison, Wisc.: ASA-CSSA-SSSA.

Hoeft, R. G., and T. R. Peck. 2001. Soil testing and fertility. In *Illinois Agronomy Handbook 2001-2002*, 78-116. Circ. No. 1360. Urbana, Ill.: University of Illinois Cooperative Extension Service.

Kaiser, D. E., A. P. Mallarino, and M. Bermudez. 2005. Corn grain yield, early growth, and early nutrient uptake as affected by broadcast and in-furrow starter fertilization. *Agron. J*. 97: 620-626.

Kelling, K. A, L. G. Bundy, S. M. Combs, and J. B Peters. 1998. Soil test recommendations for field, vegetable, and fruit crops. Publ. No. A2809. Madison, Wisc.: University of Wisconsin Extension.

Klatt, J. G., A. P. Mallarino, and B. L. Allen. 2002. Relationships between soil phosphorus and phosphorus in surface runoff and subsurface drainage. An overview of ongoing research. In *Proc. North-Central Extension-Industry Soil Fertility Conf.*, Vol. 18, 183-189. Ames, Iowa: Iowa State University Extension.

Kleinman, P. J. A., A. N. Sharpley, B. G. Moyer, and G. F. Elwinger. 2002. Effect of mineral and manure phosphorus sources on runoff phosphorus. *J. Environ. Qual*. 31: 2026-2033.

Lamond, R. E., W. B. Gordon, B. J. Niehues, and C. J. Olsen. 2001. Evaluation of starter fertilizer formulations and placements for conservation tillage production systems. In *Proc. North-Central Extension-Industry Soil Fertility Conf.*, Vol. 17. 137-143. Brookings, S.D.: Potash and Phosphate Institute.

Maguire, R. O., J. T. Sims, and T. J. Applegate. 2005. Phytase supplementation and reduced-phosphorus turkey diets reduce phosphorus loss in runoff following litter application. *J. Environ. Qual.* 34: 359–369.

Mallarino, A. P. 1997. Interpretation of soil phosphorus tests for corn in soils with varying pH and calcium carbonate content. *J. Prod. Agric*. 10: 163-167.

Mallarino, A. P. 2003a. Field calibration for corn of the Mehlich-3 soil phosphorus test with colorimetric and inductively coupled plasma emission spectroscopy determination methods. *SSSA J*. 67: 1928-1934.

Mallarino, A. P. 2003b. Foliar and starter fertilization: Are they needed to supplement primary fertilization? In *Proc Integrated Crop Management Conf.*, 113-120. Ames, Iowa: Iowa State University Extension.

Mallarino, A. P. 2003c. Using the Iowa phosphorus index for agronomic and environmental management of fertilizer and manure phosphorus. In *Proc Integrated Crop Management Conf.*, 121-126. Ames, Iowa: Iowa State University Extension.

Mallarino, A. P., and R. Borges. 2006. Phosphorus and potassium distribution in soil following long-term deep-band fertilization in different tillage systems. *SSSA J*. 70: 702-707.

Mallarino, A. P., and J. S. Schepers. 2005. Role of precision agriculture in phosphorus management practices. In *Phosphorus: Agriculture and the Environment*. A. Sharpley and T. Sims, eds. Monograph Series No. 46. Madison, Wisc.: ASA.

Mallarino, A. P., J. R. Webb, and A. M. Blackmer. 1991. Corn and soybean yields during 11 years of phosphorus and potassium fertilization on a high-testing soil. *J. Prod. Agric*. 4: 312-317.

Motavalli, P. P., K. A. Kelling, and J. C. Converse. 1989. First-year nutrient availability from injected dairy manure. *J. Environ. Qual*. 18: 180-185.

Niehues, B. J., R. E. Lamond, C. B. Godsey, and C. J. Olsen. 2004. Starter nitrogen fertilizer management for continuous no-till corn production. *Agron. J.* 96: 1412-1418.

PPI. 2001. Soil test levels in North America. Summary update. Tech. Bull. 2001-1. PPI/PPIC/FAR. Norcross, Ga.: Potash and Phosphate Institute.

Randall, G. W., and E. G. Hoeft. 1988. Placement methods for improved efficiency of P and K fertilizers: A review. *J. Prod. Agric*. 1: 70-79.

Randall, G. W., T. K. Iragavarapu, and S. D. Evans. 1997. Long-term P and K applications: I. Effect on soil test incline and decline rates and critical soil test levels. *J. Prod. Agric*. 10: 565-571.

Randall, G. W., J. A. Vetsch, and T. S. Murrell. 2001. Soybean response to residual phosphorus for various placements and tillage practices. *Better Crops* 85: 12-15.

Rehm, G. W., G. W. Randall, A. J. Scobbie, and J. A. Vetsch. 1995. Impact of fertilizer placement and tillage system on phosphorus distribution in soil. *SSSA J*. 59: 1661-1665.

Rehm, G., M. Schmidt, J. Lamb, and R. Elliason. 2001. Fertilizer recommendations for agronomic crops in Minnesota. Publ. BU-6240-S. St. Paul, Minn.: University of Minnesota Extension.

Ritchie, K. B., E. G. Hoeft, E. D. Nafziger, L. C. Gonzini, J. I. Warren, and W. L. Banwart. 1996. Starter fertilizers for no-till corn. *Better Crops* 80: 6-9.

Robbins, S. G., and R. D. Voss. 1991. Phosphorus and potassium stratification in conservation tillage systems. *J. Soil Water Conserv*. 46: 298-300.

Sawyer, J. E., A. P. Mallarino, R. Killorn, and S. K. Barnhart. 2002. General guide for crop nutrient recommendations in Iowa. Publ. PM-1688 (Rev.). Ames, Iowa: Iowa State University Extension.

Scharf, P. C. 1999. On-farm starter fertilizer response in no-till corn. *J. Prod. Agric*. 12: 692-695.

Teare, J. D., and D. L. Wright. 1990. Corn hybrid-starter fertilizer interaction for yield and lodging. *Crop Sci*. 30: 1298-1303.

Vadas, P. A., A. P. Mallarino, and A. McFarland. 2005. The importance of sampling depth when testing soils for their potential to supply phosphorus to surface runoff. Position paper. Minimizing Phosphorus Losses from Agriculture / Information Exchange Group (SERA-17/IEG) committee. Available at: www.sera17.ext.vt.edu.

Vetsch, J. A., and G. W. Randall. 2002. Corn production as affected by tillage system and starter fertilizer. *Agron. J*. 94: 532-540.

Webb, J. R., A. P. Mallarino, and A. M. Blackmer. 1992. Effects of residual and annually applied phosphorus on soil test values and yields of corn and soybean. *J. Prod. Agric*. 5: 148-152.

Wittry, D. J., and A. P. Mallarino. 2004. Comparison of uniform- and variable-rate phosphorus fertilization for corn-soybean rotations. *Agron. J*. 96: 26-33.

Pp. 105-116 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop*. St. Joseph, Michigan: ASABE.

8 Using Manure as a Fertilizer for Crop Production

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Utilizing manure as a fertilizer for crop production can be a key component of the economic success of an animal feeding operation. Successful application of any fertilizer requires correctly estimating the nutrient concentration and availability, properly calibrating the application equipment, and obtaining an optimal spreading pattern on the field. Manure has some characteristics that make it more difficult to meet these basic requirements. Failure to appropriately account for the unique attributes of manure as a fertilizer can lead to overestimating its value to the farmer. The objective of this chapter is to help the reader to better understand the characteristics of manure and the value of manure as a fertilizer source.

The Role of Nutrient Management Planning in Protecting Water Quality

Fertilizing agricultural land with the nutrients nitrogen and phosphorus often improves productivity, resulting in greater yields. Unfortunately, these same nutrients can impair water quality if they move off agricultural land into sensitive water resources (Sharpley et al. 1994).

Agriculture frequently is a significant contributor of nutrients to water resources. The *National Water Quality Inventory* (USEPA, 2002), a state-by-state biennial inventory of water quality impairment, typically lists agriculture among the top sources of nutrients in impaired streams and lakes in most states.

Manure is often linked to water quality problems. While nutrients from manure are not inherently more likely to cause water quality problems than nutrients from commercial fertilizers, some characteristics of manure make it more likely that nutrients can be over-applied to some fields.

Movement of nutrients from agricultural land to water resources is a complex process that is controlled by many factors. Nutrients can leach through the soil profile into ground water or re-emerge as seeps, springs, or from tile drains to enter surface waters. Runoff can carry nutrients as dissolved ions and in sediment. The amount of nutrient loss from a field or farm is affected by a diverse range of farm management practices, including animal feeding strategies, manure storage and handling technology, cropping systems, and timing and rate of nutrient application.

The primary water quality concern from phosphorus is its impact on surface freshwater resources such as streams and lakes (Sharpley et al. 1994). Frequently, addition of phosphorus to surface freshwater resources increases algal growth, increases the cost of water treatment, and reduces aesthetic and recreational value. Excess nitrate nitrogen in drinking water can pose health risks to human infants and young livestock. Excess nitrogen in rivers can contribute to the degradation of marine coastal areas, such as the Gulf of Mexico.

Nutrient management planning is the primary mechanism used in the U.S. to reduce the movement of nutrients from agricultural land to surface and ground water. There have been extensive efforts to encourage nutrient management planning by farmers, particularly operations with confined livestock. Two national initiatives to improve nutrient management planning in this decade are:

- The NRCS 590 nutrient management policy, released in 2001 (USDA, 2001).
- The revised concentrated animal feeding operation (CAFO) rules, released by USEPA in 2003 (USEPA, 2003).

The nutrient management planning process is an opportunity to work with a farmer to consider options for improving the efficiency of nutrient use on the farm. The nutrient management planning process can educate farmers about practices that will improve water quality and, in many cases, increase the profitability of their operation.

Comparison of Manure Characteristics with Other Fertilizer Sources

The value of manure as a fertilizer source has been recognized for thousands of years. However, in modern agricultural systems, manure sources often are underutilized as fertilizer resources for crop production. This is directly due to physical and chemical characteristics of manure that reduce its value as a fertilizer compared to other fertilizer sources commonly used by crop producers. Most manure sources have the following liabilities as a fertilizer:

- Nutrient concentration: Total fertilizer nutrient concentration rarely exceeds 10% in most manure sources and frequently is a fraction of that. For example, nitrogen, phosphate, and potash are approximately 8.5% of the weight of poultry litter, 1.5% of the weight of hog slurry, and 0.2% of the weight of hog lagoon effluent. Most commercial-grade fertilizers exceed 30% nutrient concentration by weight. Low nutrient concentration increases the time and cost of transportation and land application.
- Nutrient ratio: Modern fertilizer production practices allow blending of fertilizer constituents providing custom fertilizers that meet the specific nutrient requirements of a crop and field. Manure nutrient ratios reflect animal nutritional considerations and manure storage methods and frequently do not match the crop requirements. For example, applying poultry litter to meet the nitrogen needs of a corn crop also applies over five times more phosphate than the crop removes in the grain. It has been clearly documented that long-term use of unbalanced manure fertilizers leads to high soil test phosphorus and potassium levels. The use of alum or other manure treatments that bind phosphorus into insoluble forms can mitigate the impact of over-application of manure phosphorus on soluble phosphorus losses, but the fertilizer value of the phosphorus is lost.
- Nutrient availability: Most commercial fertilizers are designed to be rapidly available to crops when applied to the soil. The organic nitrogen fraction of manure re-

duces the availability and predictability of the manure as a nitrogen source because the availability of organic nutrients is dependent on microbial activity in the soil. The chemistry of manure makes inorganic nitrogen in manure (ammonium) prone to volatilization losses when it is surface applied. Successful use of manure fertilizer requires adjusting application rates to account for reduced nutrient availability. Sometimes manure management strategies can take advantage of the slow release characteristics of organic nitrogen and phosphorus in manure to help reduce nutrient losses from fertilizer applications.

- Uniformity: Most states have legal requirements for guaranteed analysis of products sold as commercial fertilizers. Nutrient concentrations in manure typically vary spatially and over time within the manure storage, making it difficult to meet fertilizer law requirements. Farmers also are challenged when calculating application rates of highly variable sources of manure. Should they apply a rate that on average supplies the target fertilizer rate, or select a rate that guarantees the whole field gets at least the target fertilizer rate? The first strategy ensures that portions of the field will have nutrient deficits, an economic liability to the farmer; the second strategy maximizes yield but also ensures that part of the field will have nutrient excess, a water quality liability.
- Timing: Manure may have to be applied at times that are not ideal for maximizing availability of nutrients. Manure application decisions are frequently driven by the need to empty a manure storage structure to reduce the risk of overflow or to meet animal management concerns, not to meet crop fertilization requirements.

Potential

Use of manure as a fertilizer does not necessitate that nutrient losses from agricultural systems increase compared to commercial fertilizer systems. Extensive research shows that when equivalent rates of nutrients are applied as manure or commercial fertilizers, the nutrient losses from manure applications are similar to or less than those associated with chemical fertilizers. For example, Arkansas research showed a 55% reduction in phosphorus concentration in runoff seven days after surface application of poultry litter to fescue, compared to a similar rate of inorganic phosphorus (Edwards and Daniel, 1994). In another example, a Wisconsin study demonstrated that surface-applied dairy slurry reduced total phosphorus loss from a tilled field because the manure reduced losses of particulate phosphorus (Bundy et al., 2001).

Manure management is associated with greater potential losses of nutrients because the fertilizer characteristics of manure promote over-application of nutrients. Failure to account for nutrient imbalances in manure, applying high rates to ensure sufficient available nutrients, or failure to properly account for the fertilizer value of manure (e.g., waste applications) lead to over-application of nutrients. Extensive research demonstrates that mismanagement of manure leads to over-application of nutrients and to accumulation of nutrients in excess of crop needs, which in turn leads directly to greater nutrient losses from agricultural systems.

Excessive nutrient application rates typically lead to proportional increases in potential nutrient losses. For example, Minnesota research showed linear increases in residual nitrate in the soil profile associated with over-application of manure nitrogen to corn following alfalfa (Lory et al., 1995). Numerous studies have shown that soil test values increased with increasing over-application of manure phosphorus and potassium. Increasing soil test phosphorus typically results in linear increases in phosphorus concentrations in runoff. Similarly, phosphorus concentration in runoff in the days after manure application often is linearly related to the soluble phosphorus concentration in the applied manure (e.g., Sharpley et al., 1994).

Efforts to improve manure management through nutrient management planning will reduce nutrient losses by reducing excess nutrient applications and by identifying other changes in crop management practices that will reduce the potential for transport of nutrients from agricultural fields to water resources. Other chapters in this publication will address nutrient transport and management practices to limit nutrient loss from agricultural fields.

Important Factors

Manure Nutrient Content

An estimate of manure nutrient concentrations is the starting point for any effort to use manure as a fertilizer, yet obtaining a good estimate of nutrient content in manure can be surprisingly difficult.

Tabular values are often used for planning purposes (see examples in table 8-1). Publications such as *Manure Characteristics* (MWPS-18, 2004) provide book values for many animal types and specialized manure handling systems. Book values should be judged as rough estimates despite the implied specificity of references such as table 8-1. Location-specific characteristics such as rainfall, water use, feed composition, and animal performance limit the utility of book manure nutrient values relative to manure test results from a properly sampled manure storage facility.

Sampling manure storage facilities is an essential part of using manure as a fertilizer. Unfortunately, it can be challenging to obtain a representative sample from a manure storage facility at the time of manure application. Slurry tanks are best sampled

Table 8-1. Estimated nutrient concentration in manure for selected animal types and manure storage and handling systems. All concentrations reported on an "as-is" basis. Data are adapted from Manure Characteristics (MWPS-18, 2004).

Livestock System	Units	Total N	Ammonium N	Phosphate (P_2O_5)	Potash (K_2O)
Pig, nursery, pit slurry	lbs/1000 gal	25	14	19	12
Pig, grow-finish, deep-pit slurry	lbs/1000 gal	50	33	42	30
Pig, farrow-finish, pit slurry	lbs/1000 gal	28	16	24	23
Dairy cow, pit slurry	lbs/1000 gal	31	6	15	19
Layer hen, pit slurry	lbs/1000 gal	57	37	52	33
Finishing cattle slurry	lbs/1000 gal	29	8	18	26
Pig, grow-finish, lagoon water	lbs/acre-in.	113	113	56	85
Pig, farrow-finish, lagoon water	lbs/acre-in.	127	113	81	102
Dairy cow, lagoon water	lbs/acre-in.	114	102	47	82
Broiler, dry litter	lbs/ton	46	12	53	36
Turkey, dry litter	lbs/ton	40	8	50	30
Finishing cattle feedlot	lbs/ton	11	4	7	11

after they are fully agitated, which limits the optimum time for sampling to the time of application. Sampling dry litter in poultry houses is more difficult with birds and feeders in place, so they are frequently sampled at the time of building cleanout.

Under these conditions, manure testing strategies ideally rely on using manure test records to estimate the nutrient content at the time of application. For example, in slurry operations, rates should be calculated based on the average of previous tests or the most recent past test. A new composite sample taken during application is then added to the test records and used to confirm the accuracy of the current manure application rate and to help calculate the next manure application rate. In contrast, anaerobic lagoons can be easily sampled a week or so before pumping to provide results representative of lagoon water if no agitation is planned. In this situation, the results of the current sample should be used to calculate manure application rates.

Current methods to sample lagoon sludge in the bottom of the lagoon are inadequate, and the resulting estimates of lagoon sludge or agitated lagoon nutrient concentrations are unreliable. These systems are also notoriously variable in nutrient concentration during application. Multiple composite samples should be taken during application and used to back-calculate the quantity of nutrients applied.

When test results do not exist for a manure storage facility, results from a similarly managed storage facility will typically be superior to book values. In poultry operations, litter test results from other buildings affiliated with the same processing plant often will have similar nutrient concentrations; these buildings typically have similar design, management, bird type, and feed. Missouri research showed little variation in average nutrient concentration in buildings on the same farm, and phosphorus and total nitrogen concentration in any building were within 10% of the mean of all the sampled buildings associated with a processing plant.

An emerging approach is to estimate manure nutrient content based on animal feed and engineering design criteria of the storage facility (ASAE, 2005). This approach has the most potential for covered slurry storage facilities where water inputs are predictable, nitrogen volatilization is limited, and all excreted phosphorus and potassium is applied annually. These independent estimates of manure nutrient content can be particularly valuable to validate that an operation's manure test results are accurate.

More research is needed on feed- and animal performance-based approaches for estimating manure nutrient concentrations, predicting seasonal and site-to-site variations in nutrient content in manure, developing more efficient sampling strategies for similarly managed buildings, and sampling lagoon sludge. Current regulations and standards suggest sampling every manure storage facility at least annually. There is potential to develop sampling strategies that require less extensive sampling and provide more reliable estimates of manure nutrient concentration.

Use state and regional extension publications for guidance on how to sample specific types of manure storage facilities and how to handle and ship manure samples.

Manure Nutrient Availability to Crops

Manure differs from most commercial fertilizers in that it typically includes a diverse mix of organic nitrogen compounds that require conversion to inorganic nitrogen by microorganisms (a process called mineralization) to make them available to plants. One of the challenges of manure management is to estimate the rate of nitrogen release from manure organic material and the fraction of organic nitrogen that ultimately is available to crops.

Because mineralization is a biological process, it only occurs when soil conditions are suitable for biological activity. The same conditions that promote crop growth also promote mineralization of manure organic nitrogen. Conversely, cold, dry, or waterlogged soil conditions limit nutrient release from manure.

Inorganic nitrogen in manure is dominantly in the ammonium form (NH_4-N) because there is little oxygen in most manure storage facilities, preventing formation of nitrate (NO_3-N). Manure also typically has a pH of at least 7. This combination of ammonium nitrogen and a neutral to slightly alkaline pH makes inorganic nitrogen prone to volatilization. Significant amounts of nitrogen are lost from manure storage facilities as ammonia, and these losses generally continue at greater rates than those associated with commercial fertilizers when manure is surface applied to fields.

To accurately estimate nitrogen availability of manure to a crop requires accurately estimating the fraction of organic nitrogen that is mineralized during the growing season and the fraction of inorganic manure nitrogen that is retained by the soil and available for plant uptake. A further complication is that some of the organic nitrogen can be released by the manure one and two years after application.

Most states have developed equations to estimate nitrogen availability in manure. These vary in approach from state to state. For example:

- Missouri calculations require estimates of both organic nitrogen and ammonium nitrogen in manure. Available organic nitrogen is calculated based on organic nitrogen in the manure sample multiplied by an availability factor. The mineralization factor varies based on animal type and storage type. Available ammonium nitrogen is calculated by multiplying manure ammonium nitrogen by a retention factor that varies based on manure placement method.
- Minnesota calculations require only an estimate of total nitrogen in manure. The fraction of total nitrogen available is based on multiplying total nitrogen by an availability factor that varies based on animal type and manure placement method.

There are large differences among states in estimated available nutrients, particularly in estimates of nitrogen availability (table 8-2). Some differences may be expected due to differences in climate; cool or dry environments may limit the rate of nitrogen mineralization. State-to-state variation also reflects differences in philosophy and approach to calculating nutrient availability in manure.

State-to-state differences have dramatic impacts on the amount of manure that can be applied to a field. An Illinois farmer seeking to apply 150 lbs/acre of available nitrogen can apply 5,000 gal/acre of slurry, a Minnesota farmer can apply 8,350 gal/acre, and a Michigan farmer can apply 16,650 gal/acre. Most states are in agreement that manure phosphorus and potassium is at least as available as commercial fertilizer sources. Farmers in states with a lower estimate of manure phosphorus availability may have a lesser restriction on phosphorus-based manure application rates.

Table 8-2. First-year plant-available nutrients in 1000 gallons of unincorporated surface-applied grow-finish pig slurry for selected north-central states. Based on a manure analysis of 50 lbs total nitrogen, 33 lbs ammonium nitrogen, 42 lbs phosphate and 30 lbs potash per 1000 gallons. State-specific nutrient availability calculated using Purdue University's *Manure Nutrient Availability Calculator* (Joern and Hess, 2007).

	Available Nutrients (lbs/1000 gal)		
State	Nitrogen	Phosphate	Potash
Illinois	30	42	30
Indiana	29	42	30
Iowa	27	42	30
Kansas	7	42	30
Michigan	9	42	30
Minnesota	18	34	27
Missouri	26	42	30
Nebraska	9	42	30
Ohio	27	42	30
Wisconsin	25	25	24

Predicting nitrogen availability in manure is difficult because it is highly dependent on local climate and soil conditions. However, a more integrated, equitable, and accurate system of determining nitrogen availability that accounts for regional differences in temperature and moisture is within the capabilities of the state of the science.

Manure Value

The value of manure nutrients is a topic fraught with misconceptions and oversimplifications. Many casual observers wonder why so many farmers apparently act against their own self-interest and ignore what seems to be a gold mine of nutrients in their manure storage. A more careful analysis demonstrates that the value of manure to the operation's bottom line varies greatly among farms.

An earlier section of this chapter outlined potential liabilities of manure compared to other fertilizer sources: nutrient concentration, nutrient ratio, nutrient availability, uniformity, and timing. All these factors can have a significant impact on manure value.

Nutrient concentration affects manure value through its impact on the time and volume of material that needs to be managed. Consider a farmer wanting to apply 150 lbs/acre of nitrogen. One option could be injected anhydrous ammonia with a guaranteed analysis of 82% nitrogen, requiring injection of 185 lbs of product per acre to meet crop need. If the farmer uses poultry litter, it would require over four tons of manure to provide the same amount of available nitrogen, and if the farmer used lagoon effluent, it would require 110 tons of manure (27,000 gal). Low nutrient concentration increases the time needed for nutrient application and limits the distance manure can be economically transported.

Fixed nutrient ratio also can affect manure value. We have already discussed how repeated applications of some types of manure to meet the nitrogen needs of a crop will lead to over-application of phosphorus and high soil test phosphorus levels. The value of additional manure phosphorus to high phosphorus testing fields is zero, limiting manure value to nitrogen and perhaps potassium content. Valuing all of the nutrients in manure often overestimates the economic value of manure. A farmer buying nutrients values the nutrients he needs, not necessarily what the manure happens to contain.

An analysis of nitrogen-based manure management on 39 hog operations (20 lagoon operations and 19 slurry operations) in five states demonstrated that factors such as manure management system, size of operation, and ownership structure affected manure application costs and net value (Lory et al., 2004a). Extracting manure value was a more important element of profitability for slurry operations. Manure value represented 2% of net income for lagoon operations, compared to 16% for slurry operations. Manure value exceeded application costs for nearly 60% of slurry operations compared to 15% of lagoon operations. Why were lagoons favored by some farmers over slurry tanks if they depressed manure value? Farmers with lagoons needed less land for manure application and were less dependent on land not owned by the operation. Slurry operations also required more time per animal unit to apply manure. Most importantly, investment in slurry storage and handling systems did not increase return on assets for these operations; it was more profitable to invest money in raising more hogs than in extracting more value from manure.

Limitations

Land Needs for Phosphorus-Based Application Rates

Concerns about water quality are forcing some farmers to limit manure applications to the phosphorus removal capacity of the crops harvested from a field. Many animal feeding operations will not need to immediately convert to a phosphorus-based application rate because of the revised rules. An estimate of the phosphorus land base requirement provides farmers an idea of the long-term sustainable land base for manure management. Equation 1 can be used to estimate the change in land needs when converting from nitrogen-based to phosphorus-based land base:

$$\text{Land increase (\%)} = \left[\left(\frac{\text{crop need N} : P_2O_5 \text{ ratio}}{\text{manure available N} : P_2O_5 \text{ ratio}}\right) - 1\right] \times 100 \tag{1}$$

This equation emphasizes that both the nutrient ratio of the crop receiving manure and the nutrient ratio of the manure affect the conversion from a nitrogen-based to a phosphorus-based application strategy. There will be less impact on fields with crops that have low nitrogen-to-phosphate removal ratios, such as wheat (1.9) or corn (2.2), than with crops that have higher ratios, such as alfalfa (4.2) or cool-season pasture (15). Conversion will also have less impact on fields receiving manure with a high nitrogen-to-phosphate ratio, such as injected lagoon effluent (3.4), compared to manures with low ratios, such as surface-applied hog slurry (0.7) or poultry litter (0.6).

A farmer applying hog slurry to a corn field in Missouri will have a 210% [(2.2/0.7) - 1] × 100) increase in land needs. If the operation used 100 acres for nitrogen-based application rates, then it will need 210 additional acres to apply based on phosphorus. Another operation that is injecting lagoon effluent on corn would need no additional land to adopt phosphorus-based application rates [(2.2/3.4) - 1] × 100 < 0). Phosphorus rules also will have a greater impact on farms with less productive land because increased land needs are proportional to current land needs. A farm with lower productivity needs more land for a given amount of manure and will require more acres to adopt phosphorus-based management. Actual changes in land needs also may be greater if no manure can be applied on some of the phosphorus-limited land.

Feasibility of P-Based Application Rates

There are two strategies that farmers can use to implement phosphorus-based application rates on phosphorus limited land:

- "Phosphorus rotation" is the practice of applying manure to meet the nitrogen need of this year's crop (a nitrogen-based application rate) and then refraining from additional manure applications until subsequent crops have removed the excess applied phosphorus.
- "Annual phosphorus" is the practice of limiting manure application rate to the annual crop need for phosphorus.

Both approaches require similar increases in land needs to meet phosphorus-based land application requirements. The difference is that the annual approach requires applying a reduced rate of manure on all acres every year, whereas phosphorus rotation allows application to a fraction of the land base but rotates which land receives manure each year.

An analysis of 39 U.S. swine operations (19 slurry and 20 lagoon operations) indicated that the annual phosphorus limit approach posed significant feasibility issues for farmers spreading slurry manure (Lory et al., 2004b). Annual limits required slurry operations to reduce manure application rates an average of 77% for the 19 slurry operations. To attain such reductions with their current manure application equipment would require some combination of increased travel speed, increased swath width, and reduced discharge rate. The study found that:

- None of the 19 operations could attain the reduction in application rate by only increasing travel speed.
- Reduced discharge rate was necessary to meet annual phosphorus application rates on 14 of the operations. Reducing discharge rate increases application time.
- On two of 19 operations, annual phosphorus rates were infeasible with the current manure applicator.

Rotational phosphorus rates avoid issues of equipment feasibility because manure is applied at the nitrogen-based rate in the year of manure application. It has the further benefit that manure is a complete nitrogen and phosphorus fertilizer in the year of manure application.

Annual phosphorus limits were possible on operations applying un-agitated lagoon effluent. These operations typically make multiple passes to attain nitrogen-based rates, and annual phosphorus limits were obtainable by reducing the number of passes over the field.

The results of this study imply that operations applying poultry litter will have feasibility issues similar to those of slurry operations because of the low nitrogen-to-phosphorus ratio in both types of manure.

One challenge associated with rotational phosphorus limits is to determine the maximum number of years allowed for a manure rotation. In some pasture-based systems, a nitrogen-based rate of poultry litter can apply over 15 years of phosphorus. On permitted animal feeding operations, records must be kept for five years, suggesting that no more than five years of phosphorus can ever be applied in a single phosphorus-based application rate.

Another question is whether the phosphorus rotation application strategy is a greater risk to water quality compared to annual limits. Phosphorus losses in the year of application certainly are greater on land receiving manure using a nitrogen-based limit. This is offset by the balance of the land in rotation receiving no manure, so the net loss of phosphorus from the land base may be similar in both approaches. The phosphorus rotation has the further benefit that it requires less time for manure application (no reduction in discharge rate) and does not require applying manure to every acre every year. The flexibility gained with reduced time for application and the opportunity to not apply on marginal land in wet years has the potential to reduce phosphorus losses from rotational strategies.

Summary

Nutrient management planning is an opportunity to help farmers identify ways to increase the value of manure for their farm and protect water quality. One of the challenges of manure management is that decisions are driven by more than the fertilizer value of the nutrients in the manure. These include:

- Manure storage concerns, such as ensuring that the level of the storage is sufficiently low to prevent overflow.
- Feasibility concerns, such as how much land is needed and how much time it will take to apply the manure.
- Economic concerns, such as the investment in upgraded manure equipment compared to adding to other aspects of animal production.
- Manure value concerns, such as the cost of hauling manure to a particular field and determining whether the manure will provide the needed nutrients for crop production.

Manure has a positive impact on the profitability of many agricultural operations. To fully understand what motivates manure management decisions requires a full understanding of the challenges associated with using manure as a fertilizer.

Interpretive Summary

Using manure as a fertilizer for crop production is the primary accepted mechanism for disposal of manure from animal feeding operations. Successfully using manure as a fertilizer requires assessing the available nutrients in the manure, calculating the appropriate rate to provide the needed nutrients to the crop, and applying the manure uniformly across the field at the target rate.

Efficient use of manure as a fertilizer is complicated by the imbalance of nutrients in manure, the variability in many sources of manure, the difficulties in estimating nutrient availability, and the relatively low nutrient concentration, limiting the distances that manure can profitably be transported for use as a fertilizer. Manure management is most likely to be profitable on farms that have a manure source with a relatively high nutrient concentration (like slurry manure) and that apply manure to fields near the operation and to a crop or crop rotation that can fully utilize all the applied nutrients.

Research has shown that applications of manure that fail to use the fertilizer value of manure for crop production greatly increased the potential for nutrient loss from land receiving manure. Manure application rates that exceeded a crop's nitrogen utilization capacity resulted in higher losses than application rates that were at or below optimum nitrogen rates. Soil test phosphorus above agronomically optimal levels increased the phosphorus concentration in runoff and sediment without improved crop productivity.

There is ample crop capacity to utilize excess manure nutrients to replace fertilizer. In 1997, 24.7 billion tons of nitrogen and 4.06 billion tons of phosphorus were purchased as fertilizer. A Natural Resource Conservation Service (NRCS) study estimated that confined livestock in the U.S. generated 2.58 billion pounds of nitrogen and 1.44 billion pounds of phosphorus available for land application in the same year (<15% of purchased nitrogen and phosphorus) (Kellogg et al., 2000).

The NRCS study concluded that over 60% of recoverable nitrogen and phosphorus were in excess of crop nutrient capacity of the farms where they were produced, and poultry operations were the farm type with the most excess nutrients. The same report indicated that at least 165 U.S. counties were likely to have difficulties utilizing all manure nitrogen generated in the county, and 364 counties would have trouble utilizing all the manure phosphorus generated in the county. This implies that improved utilization of manure as a fertilizer will require transporting manure over substantial distances, and in many instances applying it to fields not owned by the farmer that generated the manure. This perfectly describes the conditions that make it most difficult for a manure producer to cover manure application costs with the fertilizer value of the manure.

There is some opportunity for existing operations to pay for improved manure utilization by better capturing its fertilizer value in local crop production. The bigger challenge is in regions where manure nutrient production exceeds utilization. In these areas, it will be much more difficult to use the fertilizer value of the manure to significantly offset transportation costs to fields with fertilizer need.

There are excellent opportunities to make money with manure as fertilizer by closely linking new facilities with nearby row crop production land. In a Missouri assessment, a 4800 grow-finish hog operation using modern diets could use manure from the facility to meet the fertilizer needs of two sections of land in a corn-bean rotation while increasing net income by at least $25,000 with a greater than 15% return on assets.

References

ASAE. 2005. *Manure Production and Characteristics*. Publication D384.3, March 2005. St. Joseph, Mich.: ASAE.

Bundy, L. G., T. W. Andraski and J. M. Powell. 2001. Management practices effects on phosphorus losses in runoff in corn production systems. *J. Environ. Qual*. 30: 1822-1828.

Edwards, D. R., and T. C. Daniel. 1994. Quality of runoff from fescue plots treated with poultry litter and inorganic fertilizer. *J. Environ. Qual.* 23: 579-584.

Joern, B. C., and P. Hess. 2007. Manure nutrient availability calculator. West Lafayette, Ind.: Purdue University, Department of Agronomy. Available at: www.agry.purdue.edu/mmp/.

Kellogg, R. L., C. H. Lander, D. C. Moffit, and M. Gollehon. 2000. *Manure Nutrients Relative to the Capacity of Cropland and Pastureland to Assimilate Nutrients*. Publication No. NPS00-0589. Washington, D.C.: USDA.

Lory, J. A., G. W. Randall, and M. P. Russelle. 1995. Crop sequence effects on response of corn and soil inorganic N to fertilizer and manure nitrogen. *Agron. J*. 87: 876-883.

Lory, J. A., R. E. Massey, J. M. Zulovich, J. A. Hohne, A. M. Schmidt, M. S. Carlson, and C. D. Fulhage. 2004a. An assessment of nitrogen-based manure application rates on 39 U.S. swine operations. *J. Environ. Qual.* 33: 1106-1113.

Lory, J. A., R. E. Massey, J. M. Zulovich, J. A. Hohne, A. M. Schmidt, M. S. Carlson, and C. D. Fulhage. 2004b. Feasibility of phosphorus-based application rates on 39 swine operations. *J. Environ. Qual.* 33: 1114-1123.

MWPS-18. 2004. *Manure Characteristics*. Section 1, 2nd ed. Ames Iowa: Midwest Plan Service.

Sharpley, A. N., S. C. Chapra, R. Wedepohl, J. T. Sims, T. C. Daniel, and K. R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. *J. Environ. Qual*. 23: 437-451.

USDA. 2001. Nutrient management code 590. NRCS Standards and Practices. Washington, D.C.: USDA Natural Resource Conservation Service. Available at http://efotg.nrcs.usda.gov.

USEPA. 2002. *National Water Quality Inventory: 2000 Report*. EPA-841-R-02-001. August 2002. Washington D.C.: U.S. Environmental Protection Agency.

USEPA. 2003. National pollutant discharge elimination system permit regulation and effluent limitation guidelines and standards for concentrated animal feeding operations (CAFOs). Federal Register Rules and Regulations 68(29:7176-7274). 12 Feb. 2003.

Pp. 117-127 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop*. St. Joseph, Michigan: ASABE.

Effects of Erosion Control Practices on Nutrient Loss

9

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Soil erosion by water is caused by the detachment of soil particles by the direct action of raindrops and runoff water, and the transport of these particles by splash and very shallow overland flow to small channels or rills. Detachment of soil particles also occurs in rills due to the force exerted by the flowing water. When rills join together and form larger channels, they may become gullies. These gullies can be either temporary (ephemeral) or permanent (classical). Non-erodible channels might include grassed waterways, or designed channels that limit flow conditions so that channel erosion does not occur.

Erosion includes sheet, rill, gully, and channel erosion and is the first step in the process of sediment delivery. Because eroded sediment is typically deposited in or near the field of origin, only a fraction of the total eroded soil from an area contributes to sediment yield from a watershed. Sediment delivery is affected by a number of factors, including soil properties, topography, proximity to the stream, man-made structures (including sediment basins, fences, and culverts), channel density, basin characteristics, land use/land cover, and rainfall-runoff factors. Coarse-textured sediment and sediment from sheet and rill erosion are less likely to reach a stream than fine-grained sediment or sediment from channel erosion. In general, the larger the area is, the lower the ratio of sediment yield at the watershed outlet or point of interest to gross erosion in the entire watershed, defined as the sediment delivery ratio (SDR). The SDR for many watersheds ranges between about 15% and 40% (Novotny and Olem, 1994).

Practices to Control Soil Erosion and Sediment Delivery

Practices to control sheet and rill erosion modify one or more of the factors affecting erosion processes: slope length, slope steepness, cropping and management practices, and support practices that slow runoff water or cause deposition. In contrast, rainfall erosivity and soil erodibility, dominant factors affecting soil erosion, cannot be easily modified. In this discussion, erosion control practices are grouped as conservation tillage, which reduces sheet and rill erosion, and other practices that reduce slope length and runoff (contouring, contour strip cropping, and terraces). Other practices to control channel and gully erosion (grassed waterways, grade-control structures, terraces, and water and sediment control basins) reduce the velocity of flowing water (which reduces both erosion and sediment transport in channels) or divert flow into stable channels or pipes.

Conservation tillage is defined as a tillage system that leaves 30% or more of the land surface covered by crop residue after planting. Currently, conservation tillage is used on about 40% of all U.S. cropland. In the Midwest, no-till and strip-till soybeans continue to be more common than no-till corn. Tables 9-1 and 9-2 show current tillage practices for soybean and corn from the seven Corn Belt states (CTIC, 2004). In 2004, Illinois, Indiana, Iowa, Ohio, Minnesota, Missouri, and Wisconsin planted 17.6 million acres of no-till soybeans, while only 6.3 millions acres of corn were planted using no-till practices in those seven states.

Contouring is the practice of performing field operations across the slope. Usually, ridges develop when the land is tilled or planted, and these ridges trap excess rainfall. When there is a mild slope to the row, water may travel along the row to an outlet. Contouring is particularly effective when rainfall amounts and intensities are low, when ridges are high, and when slopes and slope lengths are not excessive. As slopes and slope lengths increase, and as rainfall amounts and intensities increase, contouring loses much of its effectiveness, and may have minimal impact on reducing soil erosion.

Strip cropping is the practice of growing alternate strips of different crops along the contour. Alternating strips are planted to crops that have different growing and harvest times. These might be a strip of row crop, with the next strip being either a small grain

Table 9-1. Tillage practices in seven Corn Belt states for soybean production (CTIC, 2004).

State	Soybean Acres	No-Till	Mulch-Till (30% residue)	Reduced-Till (15% to 30% residue)	Conventional-Till (0% to 15% residue)
Illinois	10,316,344	46.2%	20.9%	19.2%	13.7%
Indiana	5,487,069	61.5%	15.2%	10.0%	13.2%
Iowa	10,179,278	33.1%	47.3%	14.6%	4.3%
Minnesota	7,176,774	7.1%	46.1%	24.6%	21.4%
Missouri	5,143,354	40.1%	9.5%	19.9%	30.1%
Ohio	4,630,915	63.7%	9.0%	8.3%	19.0%
Wisconsin	1,540,605	36.6%	21.4%	15.8%	26.2%
Total	44,474,339	39.6%	27.8%	16.7%	15.6%

Table 9-2. Tillage practices in seven Corn Belt states for corn production (CTIC, 2004).

State	Corn Acres	No-Till	Mulch-Till (30% residue)	Reduced-Till (15% to 30% residue)	Conventional-Till (0% to 15% residue)
Illinois	11,165,908	14.0%	12.1%	22.2%	51.8%
Indiana	5,350,414	18.8%	8.6%	17.3%	55.1%
Iowa	12,348,317	14.4%	26.6%	36.5%	22.2%
Minnesota	7,388,154	1.5%	15.7%	34.1%	48.1%
Missouri	2,887,237	20.2%	7.4%	23.2%	48.9%
Ohio	3,527,939	23.5%	9.9%	13.1%	53.4%
Wisconsin	3,520,402	14.5%	18.1%	20.7%	46.5%
Total	46,188,371	13.7%	16.1%	26.6%	43.2%

or permanent grass. These strips reduce water erosion by being on the contour, and runoff passes from highly erodible row crops into small grains or grass where considerable deposition may occur.

Grassed waterways and grade control structures are designed to keep erosive forces in channels that carry surface runoff below critical values where erosion might occur. Water and sediment control basins are constructed basins that temporarily store runoff water and release it at controlled rates through underground drain lines. The temporary impoundment of runoff water reduces downstream runoff rates, preventing gullying and greatly reducing downstream sediment delivery.

Terraces are broad channels across the slope. Runoff water above the terrace follows these broad channels to an outlet. Terraces reduce slope length and deliver surface runoff through channels that are designed to be non-erodible and to prevent sediment deposition. A well designed terrace system will use grassed waterways or underground outlets to prevent channel erosion as surface runoff exits the area. Some terraces do not follow the contour, and water is stored in small impoundments until discharged through underground outlets.

Potential Benefits of Erosion Control Practices

Although significant gains in erosion control have been made over the last 20 years, soil erosion continues to be an important environmental concern. It is estimated that over 423 million tons of topsoil eroded from the seven Corn Belt states in 1997, while in 1982 the estimated loss was approximately 707 million tons. Individual states vary considerably in the rate of soil loss. In 1997, average annual sheet and rill erosion rates on cropland for Illinois, Indiana, Iowa, Minnesota, Missouri, Ohio, and Wisconsin were 4.1, 3.0, 4.9, 2.1, 5.6, 2.6, and 3.7 tons ac^{-1} $year^{-1}$, respectively (USDA, 2000).

Contouring and contour strip cropping can be very effective in reducing soil erosion. Where it is most effective, contouring can reduce soil erosion about 50%, and contour strip cropping will reduce erosion further in most cases (Wischmeier and Smith, 1978). However, both practices have limits of application.

Contouring is most effective on slopes of 2% to 10%. As slopes get steeper than 10%, the effectiveness of contouring is reduced, and this practice is not well suited to rolling topography having a high degree of slope irregularity (USDA, 2001).

Terraces are an effective means for controlling slope length and reducing soil erosion on erodible areas. Terraces may discharge water through surface channels, by infiltration in a pondage area, or through underground drain lines. They have a negligible effect on crop yields, but a major effect on sediment delivery. Terraces that drain by surface channels are designed to have no erosion in the terrace channels. Controlling slope length will reduce soil erosion and channel erosion between terraces, but to greatly impact sediment delivery, practices that further reduce soil erosion, such as conservation tillage, should be used between terraces. Cropping is generally done on the contour for surface-drained terraces. Depending on design, deposition may occur in surface-drained terraces.

Terraces that drain through underground outlets are very effective at reducing sediment delivery of eroded material. Laflen et al (1972) estimated that about 95% of material eroded between terraces was deposited in pondage areas around underground outlets, and that material discharged was almost all less than 0.016 mm in diameter. One advantage of this type of drainage is that terraces can more easily be constructed parallel to each other or to field boundaries. This type of terrace lends itself to modern farming techniques because rows are parallel to field boundaries, avoiding point rows and small areas that are difficult to farm. Since farming for this type of terrace is generally not done on the contour, other practices, such as conservation tillage, are needed to reduce erosion between terraces.

Terraces that drain via surface channels work well on gently sloping lands with long slopes. They require some routine maintenance to ensure that they drain adequately. They also work nicely when small grains are grown because it is easier to farm over the terraces.

Practice Effectiveness

Table 9-3 summarizes the results of simulations of the effects of various erosion control practices on soil and nutrient losses compared to a tillage system typically used in the Corn Belt. The water erosion prediction project (WEPP) model (Laflen et al., 1997; Flanagan et al., 2001) was used to calculate runoff and soil loss for all tillage systems and to calculate enrichment ratios for sediment. The typical tillage system for a corn-soybean rotation leaves 20% residue cover after corn planting and 40% residue after soybean planting. For all practices except water and sediment control basins,

Table 9-3. Estimated annual soil and nutrient losses for selected erosion control practices (central Iowa climate, average over ten Iowa soils and a 72.6 foot long slope of 9% and a 300 foot long slope of 5%).

Practice	Runoff (in.)	Soil Erosion/ Sediment Yield ($t\ ac^{-1}\ year^{-1}$)	Nutrient Enrichment Ratio[a]		Losses in Surface Runoff Water ($lb\ ac^{-1}$)		Losses in Eroded Soil ($lb\ ac^{-1}$)		Total Water and Soil Losses ($lb\ ac^{-1}$)	
			Sed.	Water	NH_4-N + NO_3-N	PO_4-P	Total N	Total P	N	P
Moldboard plow	5.2	15.0	0.6	0.4	2.2	0.1	53.4	20.9	55.6	21.0
Typical tillage	4.8	7.8	1.0	1.0	3.0	0.4	32.8	12.7	35.8	13.1
No-till	4.2	1.0	1.5	1.7	3.6	0.7	6.1	2.4	9.7	3.1
Contour farming	4.4	3.9	0.8	1.3	3.5	0.5	12.5	4.8	15.9	5.3
Strip cropping	4.4	2.9	0.8	1.3	3.5	0.5	9.5	3.7	12.9	4.2
Terraces (surface drained)	4.4	2.3	0.8	1.3	3.5	0.5	7.4	2.9	11.0	3.4
Water and sediment control basins	3.9	0.4	1.5	1.7	4.0	0.6	2.5	1.0	6.5	1.6

[a] Nutrient enrichment ratios, relative to the typical tillage practice, were calculated based on concentrations obtained by Baker and Laflen (1983), and on soil erosion and sediment yields.

simulated losses are to the end of the slope; for water and sediment control basins, the values represent losses at the end of the outlet for the basin. The values also are not adjusted for sediment deposition or ponding of runoff water prior to reaching a stream. For specific fields, the SDR may range between 0% and 95% depending primarily on distance to a stream.

For reference, the base soil loss of 7.8 tons ac^{-1} $year^{-1}$ is about twice the 1997 average annual soil loss in the Corn Belt (2004 estimates for Illinois indicate that less than 10% of fields have erosion rates >7.5 tons ac^{-1} $year^{-1}$). In many watersheds in the region, total phosphorus yields from intensively cropped watersheds are about 1 lb ac^{-1} $year^{-1}$ (Goolsby et al., 1999).

Total nitrogen yields vary greatly, but are typically less than 10 lb ac^{-1} $year^{-1}$ in non-tiled drained watersheds and greater than 20 lb ac^{-1} $year^{-1}$ in tile-drained watersheds, respectively. The majority of the N lost in eroded soil is organic nitrogen. Due to sediment deposition in the field and in reservoirs, and because organic N is refractory, this form of N is not likely to be a major contributor to eutrophication in the Gulf of Mexico. In tile-drained watersheds and in large rivers, most of the N (>70%) is in the form of nitrate (McIsaac and Hu, 2004; Goolsby et al., 1999).

In our simulations, all erosion control practices considered increased losses of dissolved nutrients compared to the moldboard plow system. The effect of erosion control practices in increasing runoff losses of nitrate is small because the dominant path for nitrate loss is leaching and nitrate concentrations in runoff are usually low compared to subsurface drainage waters. The impacts of increased losses of dissolved phosphorus and decreased losses of particulate phosphorus due to the widespread adoption of conservation tillage systems are less certain. In some settings, dissolved inorganic phosphorus is likely to be more biologically available than sediment-bound phosphorus. In other settings, dissolved phosphorus may become sediment bound and relatively unavailable. On the other hand, sediment-bound phosphorus can become desorbed in anaerobic environments, and thus become more biologically available for phytoplankton.

Factors Affecting Nutrient Loss

Soil erosion and associated nutrient transport is driven by surface runoff, which is generated disproportionately from soils that have low infiltration capacity as a consequence of such factors as high clay content, surface crusting, high water table, or shallow bedrock. Phosphorus transport in runoff tends to increase with increasing phosphorus concentration at the soil surface and increasing runoff (Sharpley et al., 2003). Thus, practices that reduce phosphorus concentrations in the soil surface and/or reduce surface runoff are most effective in controlling P transport. When tillage is reduced or eliminated, particulate phosphorus loss in surface runoff usually declines, but dissolved P losses may increase if phosphorus becomes more concentrated near the soil surface unless P fertilizers or manure are injected or incorporated into the soil. Thus, timing and methods of application of P fertilizer become more important to controlling phosphorus transport in runoff from reduced tillage systems.

Conservation tillage practices that leave crop residue on the soil surface protect fine-textured soils from forming surface crusts, and therefore have the potential to reduce runoff in soils where crust formation is a major limitation to infiltration. There are some reports of dramatic reductions in runoff from continuous no-till on well-drained soils, where after three or four years, accumulations of organic matter and/or earthworms develop and maintain high porosity at the soil surface (Shipitalo et al., 2000). However, in other settings, no-till has not had a significant influence on runoff (Ghidey and Alberts, 1996).

Residue cover also reduces evaporation from the soil surface, thereby increasing soil moisture content, which may increase runoff. Additionally, infiltration can be limited by factors other than the soil surface condition and residue. Residue cover may have little influence on runoff or dissolved phosphorus transport where infiltration is limited by a claypan, shallow bedrock, high water table, or seasonal precipitation patterns that saturate most soils. Conservation tillage is probably most effective in reducing runoff, soil loss, and nutrient transport in well-drained, fine-textured soils, and where phosphorus fertilizer and manure are injected or incorporated into the soil.

The interaction of tillage systems and nutrients in tile drainage is unclear. Tile drainage reduces surface runoff and therefore soil loss and particulate P transport. Phosphorus concentrations in tile drainage water can be high, however, if P concentrations in the soil are high or if soil macropores result in preferential flow (Sharpley et al., 2003). Although phosphorus and ammonia tend to be adsorbed in the top 15 to 30 cm of soil, they can also move through soil and can be found in tile drainage waters, particularly during high flow events when significant quantities of water move rapidly to the tile through macropores such as large cracks or holes in the soil. This results in minimal contact between the water and soil, so less adsorption takes place. Dissolved phosphorus concentrations in excess of 50 ppb have been observed in tile drainage waters when soil phosphorus concentrations are high (Xue et al., 1998).

In contrast to P, nitrate is highly soluble and generally does not adsorb to soils. Rather, when water infiltrates into soil, nitrate tends to move with water into the soil profile. Consequently, there are usually low nitrate concentrations at the soil surface during runoff events and in runoff. In sandy soils and in tile-drained fields, nitrate can be rapidly leached out of the root zone to groundwater, to tile drains, and ultimately to streams and rivers. As a result, tillage practices seem to have little influence on the quantity of nitrate leached (Zucker and Brown, 1998). An exception may occur when fall tillage is followed by warm and wet conditions in the winter and early spring, which may promote mineralization, tile flow, and high nitrate flux (Randall and Goss, 2001).

Most of the soil and nutrient losses in surface runoff tend to occur in a few rare events that involve large quantities of runoff. Most conservation measures are most effective at reducing runoff and erosion from smaller and more frequent events, and are less effective as the amount of precipitation and runoff increases. Soils can be especially vulnerable to runoff and erosion when a moderately large quantity of rain occurs in late winter when frost prevents percolation of water into the soil. If P fertilizers and manure had been surface applied when the soils were frozen, the resulting runoff may be very high in P. Soils are also vulnerable to erosion in the spring planting season, before the crop has developed. Soils tend to have high water content at this time of year, and a moderate rainfall event can produce significant quantities of runoff and erosion. As the season progresses, the crop canopy and the extraction of water from the soil tend to reduce runoff and erosion. The pattern of runoff and erosion that occurs in a given year depends on the timing of precipitation and canopy development, which is highly variable from one year to the next. Thus, the effectiveness of soil conservation practices in reducing runoff and erosion is highly variable and difficult to accurately determine from short-term experiments. A commitment to intensive long-term monitoring is needed to quantify the impacts of conservation practices on water quality.

In many streams and rivers, sediment from the erosion of past decades is stored in stream channels. This sediment becomes mobilized during high flow events, and will probably be a source of turbidity for decades (Trimble, 1999). Agricultural practices that reduce peak runoff rates may also reduce the problems related to the remobilization of this stored sediment.

Additionally, it should be recognized that reducing sediment concentrations in streams may allow for greater light penetration into the water column, which may allow for more algae growth where phosphorus concentrations are sufficiently high. This possibility should not discourage conservation efforts, but should be considered when comparing expectations and strategies for conservation programs.

Limitations of Erosion Control Practices

Conservation tillage systems that maintain crop residues on the soil surface for erosion control can be successfully used for almost any land, and any crop or crop rotation. Recent work by Buman et al. (2004) demonstrated that profits from conservation tillage systems for a corn-soybean rotation in the Corn Belt were greater than for conventional tillage systems. While yields were slightly lower for no-till systems for corn production as compared to other tillage systems (including a strip tillage system), the reduced production costs for no-till more than offset the yield advantage of conventional tillage systems.

Conservation tillage has a significant effect on soil erosion and water quality. Changes in soil structure, water infiltration, and distribution of nutrients and pesticides in the soil profile are all influenced by the type and extent of tillage. Although balancing water quality goals and adjusting tillage practices to address specific water concerns are important considerations, modifying other management practices may have more immediate impacts. Nutrient application rates, timing, placement, cropping systems, and the extent and management of subsurface drainage could have a greater influence on water quality than tillage practices.

Conventional tillage with a moldboard plow that buried nearly all crop residue has virtually disappeared from Midwestern agriculture. The moldboard plow has been replaced with the chisel plow or other full-width tillage tools that leave considerable residue on the soil surface. When combined with secondary tillage, the chisel plow may not leave 30% of the surface covered with plant residue after planting, the minimum level to be considered conservation tillage. These tillage tools have become the "conventional" tillage tools of modern agriculture and have few limitations. There are a wide variety of these systems that can have a major impact on reducing soil erosion. Even small amounts of residue may considerably reduce soil erosion on many lands in the Corn Belt.

Many conservation tillage tools have virtually no constraints as far as costs, production risks, or machinery shortcomings. The best system for conserving soil, the no-till system, may have major constraints in some situations. In cool climates and the wet, poorly drained soils common in the northern Corn Belt, delayed planting, emergence, and plant growth may reduce yields in some years. While long-term results using no-till might be satisfactory, yields are more variable than for other conservation tillage systems, restricting acceptance by farmers in some areas.

Contouring is an effective practice capable of reducing soil erosion on land that does not suffer from severe soil erosion. However, since farm equipment has increased in size, it is less frequently used because it is difficult to follow the contour with large equipment, and it is difficult to farm the small portions of fields that result when fields are rectangular and rows curve to follow the contour of the land. True contouring is seldom practiced, while cross-slope farming is more common, with machines traveling parallel to field boundaries.

Contouring is effective with small and medium-sized storms, and has limited effectiveness during large storms. It diminishes in effectiveness as annual rainfall increases, and as slopes increase. At its maximum effect, contouring will reduce erosion by about 50%. However, on long slopes, or very steep slopes, this practice is not very effective. Contouring has little impact on crop yields, unless ridges are high and it is used in areas where yields are limited by soil moisture availability. In these cases, yields may be increased because of water conservation.

Terraces that drain via underground drain lines trap sediment so that pondage volume will be reduced over time, rendering the terraces ineffective because of overtopping. Use of conservation tillage systems that reduce soil erosion between terraces may extend the life of terraces. An additional benefit of terraces is that runoff is stored in the impoundments, and released at very low rates, reducing downstream channel erosion and off-site damages due to flooding. However, terraces are usually designed to store a limited amount of runoff, and storms that are larger than the usual ten-year design period may lead to overtopping, causing damage not only to the terrace but also to channels and structures downstream. Terraces are expensive to construct, some designs remove land from production, and they interfere with farming operations. Unfortunately, terraces have a relatively short span of effectiveness because they are designed to hold a limited amount of runoff water. Few terraces in the Corn Belt constructed prior to 1970 are still functional.

Water and sediment control basins perform similarly to terraces with underground outlets, but do not reduce slope length or erosion losses in the field. It is very important to have soil erosion control on the watershed above the sediment control basin to ensure a long, effective basin life.

Table 9-4. Estimated annual costs for reductions in soil and nutrient losses for selected erosion control practices compared to typical tillage. Practice effectiveness from table 9-3 used for estimates of cost-effectiveness. For each constituent, annual costs are calculated based on total practice cost.

Practice	Incentive Payment/ Construction Cost ($ ac^{-1})	Practice Lifespan (years)	Annual Cost: Erosion Reduction ($ t^{-1} $year^{-1}$)	Annual Cost: Nitrogen Reduction ($ lb^{-1} $year^{-1}$)	Annual Cost: Phosphorus Reduction ($ lb^{-1} $year^{-1}$)
No-till	$20	2	$1.46	$0.38	$1.00
Contouring	$10	5	$0.51	$0.10	$0.26
Strip cropping	$25	5	$1.03	$0.22	$0.56
Terrace with vegetative outlet	$550	20	$5.00	$1.11	$2.84
Water and sediment control basin	$600	10	$8.10	$2.05	$5.22

Cost-Effectiveness of Erosion Control Practices

Even though some structural practices may be more effective than cropping system practices in reducing sediment and nutrient losses, the cost per unit of soil or nutrient saved is typically much greater (table 9-4) .The cost estimates shown in table 9-4 should be considered order-of-magnitude estimates of the cost-effectiveness of various erosion control practices. If a producer adopts a cropping system practice as a result of an incentive payment or for cost savings (e.g., no-till soybeans), the per-ton or per-pound cost of the practice will rapidly approach zero.

The incentive payments for changes in management practices, such as no-till or contouring, are usually offered at a specific rate per acre. Therefore, the cost per ton of soil loss and associated nutrient reduction is dependent on the change of the erosion rate on the field after implementing the practice. The costs of structural practices vary widely based on site conditions and the assumed life of the practice. Forster and Rausch (2002) reported costs in two Ohio watersheds of about $2.50 ton^{-1} of soil saved for no-till and more than $40 ton^{-1} of soil saved for sediment or water control structures. At erosion rates equal to the 1997 NRI estimates for average soil loss rates in the Corn Belt states, the per-ton or per-pound costs double.

The effective cost of erosion control practices in reducing losses of sediment and nutrients to a stream also varies greatly depending on the delivery of runoff water and sediment to the stream. A field immediately adjacent to a stream may deliver almost all of the sediment and nutrients to that stream, while a field several miles away may contribute only a small portion. Consequently, the cost of reduction per ton of soil or per pound of nutrient may be significantly different, depending on location.

Summary

The maximum annual amount of soil that can be removed before long-term natural soil productivity is adversely affected is referred to as *T*, or the tolerable soil loss level. However, reducing soil erosion losses to *T*, typically 3 to 5 tons per acre per year for Corn Belt soils, may not adequately protect water quality. Erosion control practices can substantially reduce particulate phosphorus and nitrogen loss from fields, but may increase dissolved phosphorus losses if fertilizer or manure is not effectively incorporated into the soil. Erosion control practices have relatively little impact on inorganic nitrogen losses. The fraction of the nutrient and sediment losses delivered to surface water is affected by practices in the field as well as the distance and path traveled between the field and stream. For example, a field with high concentrations of phosphorus in the soil surface adjacent to a stream and eroding at half the *T* value may have greater impacts on water quality than a field with low phosphorus levels eroding at >3*T* but four miles from the stream.

One significant benefit of erosion control practices is the maintenance of the soil productivity. Grass waterways and conservation tillage also provide food and habitat for birds and small mammals. In addition, continuous no-till systems may sequester more carbon than conventional tillage systems (Hooker et al., 2005).

In order to accurately assess the costs and benefits of erosion control practices, they should be considered as part of an overall system. Conservation systems need to consider individual landscapes, watershed conditions, and production resources. In addition, the cost of water quality improvements may not be uniform across production systems. For example, the cost-effectiveness for reducing sediment, nitrogen, or phosphorus will produce a greater return when practices are targeted to vulnerable areas. In contrast, the incremental cost of water quality improvements may become limiting if current conditions are already favorable.

The most immediate research needs regarding the effectiveness of erosion control practices in reducing nutrient losses are: (1) accounting for the ultimate fate of the various forms of phosphorus leaving the edge of field, and (2) quantifying the environmental significance of those forms within surface water. While the greatest losses of phosphorus from many fields are attached to sediment, some erosion control practices, such as conservation tillage systems, may increase losses of dissolved phosphorus. The bioavailability of particulate and dissolved phosphorus within different water body types must be better understood to ensure that efforts to reduce total phosphorus losses do not increase losses in a form that may have more negative impacts on water quality.

References

Baker, J. L., and J. M. Laflen. 1983. Water quality consequences of conservation tillage. *J. Soil Water Conserv*. 38(3): 186-193.

Buman, R. A., B. A. Alesii, J. L. Hatfield, and D. L. Karlen. 2004. Profit, yield, and soil quality effects of tillage systems in corn-soybean rotations. *J. Soil and Water Conserv*. 59(6): 260-270.

CTIC. 2004. National crop residue management survey. West Lafayette, Ind.: Purdue University, Conservation Technology Information Center. Available at: www.ctic.purdue.edu.

Flanagan, D. C., J. C. Ascough II, M. A. Nearing and J. M. Laflen. 2001. Chapter 7: The water erosion prediction project (WEPP) model. In *Landscape Erosion and Evolution Modeling*, 145-199. R. S. Harmon and W. W. Doe III, eds. New York, N.Y.: Kluwer Academic.

Forster, D. L., and J. N. Rausch. 2002. Evaluating agricultural nonpoint-source pollution programs in two Lake Erie Watersheds. *J. Environ. Qual*. 31: 24-31.

Ghidey, F., and E. E. Alberts. 1996. Comparison of measured and WEPP-predicted runoff and soil loss for midwest claypan soil. *Trans. ASAE* 39(4): 1395-1402.

Goolsby, D. A., W. A. Battaglin, G. B. Lawrence, R. S. Artz, B. T. Aulenbach, R. P. Hooper, D. R. Keeney, and G. J. Stensland. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River basin. Topic 3. Report of the Integrated Assessment on Hypoxia in the Gulf of Mexico. Decision Analysis Series No. 17. Silver Spring, Md.: NOAA Coastal Ocean Program.

Hooker, B. A., T. F. Morris, R. Peters, and Z. G. Cardon. 2005. Long-term effects of tillage and corn stalk return on soil carbon dynamics. *SSSA J*. 69: 188-196.

Laflen, J. M., H. P. Johnson, and R. C. Reeve. 1972. Soil loss from tile-outlet terraces. *J. Soil Water Conserv*. 27(7): 74-77.

Laflen, J. M., W. J. Elliot, D. C. Flanagan, C. R. Meyer and M. A. Nearing. 1997. WEPP-Predicting water erosion using a process-based model. *J. Soil and Water Conserv.* 52(2): 96-102.

McIsaac. G. F., and X. Hu. 2004. Net N input and riverine N export from Illinois agricultural watersheds with and without extensive tile drainage. *Biogeochemistry* 70: 251-271.

Novotny, V., and H. Olem. 1994. *Water Quality: Prevention, Identification, and Management of Diffuse Pollution*. New York, N.Y.: Van Nostrand Reinhold.

Randall, G. W., and M. J. Goss. 2001. Nitrate losses to surface water through subsurface tile drainage. In *Nitrogen in the Environment: Sources, Problems, and Management*. R. F. Follet and J. L. Hatfield, eds. Amsterdam, The Netherlands: Elsevier Science.

Sharpley, A. N., T. Daniel, T. Sims, J. Lemunyon, R. Stevens, and R. Parry. 2003. *Agricultural Phosphorus and Eutrophication*. 2nd ed. ARS-149. Washington, D.C.: USDA Agricultural Research Service.

Shipitalo, M. J., W. A. Dick, and W. M. Edwards. 2000. Conservation tillage and macropore factors that affect water movement and the fate of chemicals. *Soil and Tillage Research* 53: 167-183.

Trimble, S. W. 1999. Decreased rates of alluvial sediment storage in the Coon Creek basin, Wisconsin, 1975-93. *Science* 285: 1244-46.

USDA. 2000. Summary report: 1997 national resources inventory (revised December 2000), Washington, D.C.: USDA Natural Resources Conservation Service, and Ames, Iowa: Iowa State University, Statistical Laboratory.

USDA. 2001. Field office technical guide. Conservation practice standard: Contour farming (330). Washington, D.C.: USDA Natural Resources Conservation Service.

Wischmeier, W. H., and D. D. Smith. 1978. Predicting rainfall erosion losses: A guide to conservation planning. Agriculture Handbook No. 537. Washington, D.C.: USDA.

Xue, Y., M. B. David, L. E. Gentry, and D. A. Kovacic. 1998. Kinetics and modeling of dissolved phosphorus export from a tile-drained agricultural watershed. *J. Environ. Qual*. 27: 917-922.

Zucker, L. A., and L. C. Brown, eds. 1998. Agricultural drainage: Water quality impacts and subsurface drainage studies in the Midwest. Extension Bulletin 871. Columbus, Ohio: Ohio State University.

Pp. 127-148 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008.
Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop. St. Joseph, Michigan: ASABE.

10

Potential and Limitations of Cover Crops, Living Mulches, and Perennials to Reduce Nutrient Losses to Water Sources from Agricultural Fields in the Upper Mississippi River Basin

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Losses of various forms of the major nutrients, nitrogen (N) and phosphorus (P), to water resources in the five states (Minnesota, Wisconsin, Illinois, Iowa, and Missouri) that comprise most of the Upper Mississippi River basin threaten aquatic ecosystems and impair water sources. Numerous studies at the field and watershed scale have shown that a significant proportion of the nitrate and phosphorus in surface waters in the Upper Mississippi River basin comes from agricultural land (Goolsby et al., 1999). Summer annual grain crops are the predominant agricultural cropping system in the Upper Mississippi River basin, and attempts to reduce nitrate and P losses to surface waters in this region have focused on fertilizer and manure management. Unfortunately, fine-tuning fertilizer and manure management will probably not reduce nutrient losses to acceptable levels. For example, nitrate losses can still be substantial even when nitrogen fertilizers are applied at less than the economic optimum rate for corn production (Baker et al., 1975; Gast et al., 1978; Jaynes et al., 2001; Dinnes et al., 2002). Thus, additional management practices are needed.

Soil nutrients in summer annual grain cropping systems are susceptible to losses, in part because there are extended periods during each year when living plants are not removing nutrients from the soil. The conversion of the prairies or other native vegetation ecosystems to summer annual grain crops resulted in a shortening of the growing season. Summer annual grain crops, like corn and soybean, accumulate water and nutrients and provide living cover for only about four months of the year (mid-May to mid-September), whereas in natural systems some living plants are actively accumulating nutrients and water whenever the ground is not frozen (at least seven months in the Upper Mississippi River basin; April to October). Cover crops, living mulches, and perennial crops can extend the period of active nutrient and water uptake in agricultural systems. Lengthening the period of active uptake increases annual plant uptake of nutrients, reduces soil concentrations of nutrients, and provides living plant cover for the soil surface, thus reducing the potential for nutrient and sediment losses into surface waters.

This chapter was originally prepared for the "Gulf Hypoxia and Local Water Quality Concerns Workshop: A Workshop Assessing Tools to Reduce Agricultural Nutrient Losses to Water Resources in the Corn Belt" held 26-28 September 2005, in Ames, Iowa. The papers prepared for the conference were intended to supplement and summarize an oral presentation, were prepared for a lay audience, were to use limited references and rely on the expert opinions and experience of the authors, and were to follow the general outline and questions posed by the workshop organizers. As a result, this chapter is presented in outline form and attempts to address the questions posed at the workshop.

What is the specific practice that would be recommended?

Cover crops are literally "crops that cover the soil" and may be used to reduce soil erosion, reduce nitrogen leaching, provide weed and pest suppression, and increase soil organic matter (fig. 10-1; Wilson et al., 1993; Sullivan and Diver, 2001; Sustainable Agriculture Network, 2007; UC SAREP, 2002; Singer et al., 2005; Snapp et al., 2005). Winter cover crops are planted shortly before or soon after harvest of the main grain crop and are killed before or soon after planting of the next grain crop. Small grains, such as oat, winter wheat, barley, triticale, and winter rye, are excellent winter cover crops because they grow rapidly in cool weather, withstand moderate frost, and their seed is relatively inexpensive. Many varieties of winter rye, triticale, and winter wheat can overwinter in the Upper Mississippi River basin and continue growing in the spring. These winter-hardy cover crops must be killed with herbicides, tillage, or mechanical rolling prior to planting corn or soybean. Oat, barley, spring wheat, and some varieties of winter wheat and triticale are not winter-hardy in this region. Because the non-winter-hardy small grain varieties do not survive the winter, they do not need to be killed prior to planting the main crop, but they also do not produce as much shoot or root growth as winter-hardy small grain varieties planted after full-season grain crops (Johnson et al., 1998). When the non-winter-hardy small grains are seeded in August after short-season crops or by overseeding, they can produce substantial biomass. Winter-hardy legumes, such as alfalfa, hairy vetch, red clover, white clover, and sweet clover are also excellent cover crops, and they fix nitrogen as an added benefit. However, if nitrogen is available in the soil, then legumes will generally take up N rather than fix it. Legumes usually do not grow as well as the small grains during the fall and winter months, they accumulate less soil N than the small grains, their seed is relatively expensive, and most must be killed with tillage or herbicides in the spring. Grasses (such as annual ryegrass) and brassicas (such as oilseed radish, oriental mustard, and forage radish) are also potential cover crops. Cool-season grasses and brassicas grow well in cool weather, but winter hardiness is species and location dependent. The brassicas have been shown to suppress nematodes, some diseases, and winter annual weeds. Seed costs are higher, and seed is usually more difficult to obtain than small grain seed.

Figure 10-1. Rye winter cover crop in April planted following corn silage.

Figure 10-2. Mustard living mulch growing in a corn crop.

Living mulches are defined by Hartwig and Ammon (2002) as cover crops planted either before or with a main crop and maintained as a living ground cover throughout the growing season (fig. 10-2). Living mulches are often perennial species and are maintained from year to year. Annual or biennial plant species can also be used as living mulches, but they need to be replanted or allowed to reseed to maintain their stand. Ideally, the growth of the living mulch is suppressed when the main crop is growing and increases as the main crop matures or when it is no longer present. Perennial legumes (such as alfalfa, red clover, kura clover, birdsfoot trefoil, crownvetch, and white clover) and perennial grasses (such as orchardgrass, reed canarygrass, and turfgrasses) can be used as living mulches. Currently, living mulch systems are being used in vineyards and orchards, but their use in annual grain crop systems is mostly experimental at this time. A major problem with living mulches in annual grain crop systems is that the living mulch competes with the main crop for water and nutrients, which can reduce main crop growth and yield or result in mortality of the living mulch. Additionally, living mulch plants often have difficulty surviving in the reduced light conditions present under the full canopy of an annual grain crop. Thus, improved living mulch management strategies need to be developed before they will be widely adopted in annual grain crop systems.

Perennials crops grow for multiple years from a single planting and would replace annual grain crops as the cash crop (fig. 10-3). Obviously, for a perennial crop to replace an annual grain crop, a market for the perennial crop must be available or the crop must be used on-farm. Currently, the most common perennials found in agricultural systems in the Upper Mississippi River basin are forages (grasses and legumes)

Figure 10-3. Perennial grass pasture system.

planted for hay, grazing, or pasture. Perennials also include trees and woody species grown for nut, fruit, or wood production (apples, grapes, hazelnuts, poplars, and walnuts). In the future, perennial biomass crops (switchgrass and poplar), and perennial grains and oil seed crops (Illinois bundleflower, wheat, sunflower, and flax) may become more important.

What is the logic or process behind the practice?

Soluble nutrients in the soil, like nitrate and dissolved phosphorus, are susceptible to leaching losses. When plants are present and actively growing, they remove water and soluble nutrients from the soil, and this decreases the downward movement of water and nutrients in the soil. Because cover crops, living mulches, and perennial crops grow earlier in the spring and later in the fall than summer annual grain crops, they extend the season of active water and nutrient uptake beyond that of annual grain crops, increase annual plant uptake of nutrients, and decrease soil nutrient concentrations during late fall, winter, and early spring. For all three of these cropping system practices, the amount of nutrient uptake is proportional to the amount of growth of these plants. Ideally, a cover crop or living mulch accumulates most of its nutrients when the main crop is not actively growing (or nutrient demand by the main crop is low) and when the nutrients accumulated would have been susceptible to leaching losses. Additionally, for cropping systems with cover crops and living mulches, it is assumed that annual fertilizer applications are lower or the same as what would be applied to the main crops without cover crops. In the case of perennial cropping systems, the perennials are the main crop. In order for them to reduce nutrient losses relative to an annual grain crop, they must maintain nutrient uptake for a larger portion of the year, and they must be managed in such a way as to minimize nutrient losses from applied fertilizers or manures. Less frequent tillage of perennials relative to annual crops also reduces mineralization of soil organic matter and nutrient loss, while their more established and more extensive root systems scavenge nutrients from a larger soil volume.

Because cover crops, living mulches, and perennial crops increase surface cover, anchor residues (e.g., main crop residues for cover crops and living mulches), increase infiltration, and reduce both rill and interrill erosion, they reduce phosphorus, nitrogen, and pesticide losses and movement associated with soil sediment. Additionally, because these practices increase infiltration, we would assume that they also reduce runoff volume and thus losses of dissolved phosphorus, nitrogen, and pesticides in surface runoff. The potential for dissolved P losses from shoots and residues of cover crops, living mulches, and perennial crops during winter or after freeze-thaw cycles has not been evaluated extensively and may contribute to P losses.

Cover crops, living mulches, and perennial crops normally begin growth and water use earlier in the spring and then continue water uptake later into the fall than most summer annual grain crops. This extended period of water use normally reduces the total annual volume of drainage water and runoff because of reduced soil water contents. Thus, losses of nitrate and dissolved phosphorus in drainage water and runoff are reduced.

Potential

What are the estimated range and mid-range values for percent loss reductions, given a defined application of the practice?

Winter cover crops and rye cover crops specifically can reduce water flows, nitrate concentrations, and total nitrate load in tile drainage. Effectiveness of the rye cover crop varies with growth of the cover crop, weather, and management of the main crops. More growth of the cover crop will result in greater reductions in nitrate leaching, but growth of cover crops can be limited by cold temperatures, water stress, nutrient availability, and delays in establishment. Similarly, lack of precipitation and soil freezing may eliminate or greatly reduce nitrate leaching losses and thus reduce the impact of the cover crop. Lastly, reducing N fertilizer rates and applying N fertilizer closer to the time of crop uptake will also reduce nitrate leaching and the impact of the cover crop. Reductions in nitrate load observed with a rye cover crop range from 13% in Minnesota to 94% in Kentucky (table 10-1). We hypothesize that the smaller reduction in the nitrate load in the Minnesota study compared with the Kentucky study would be partly caused by less cover crop growth and frozen soils in Minnesota. Additionally, combining a winter cover crop with other nitrogen best management practices can also effectively reduce nitrate losses. A study in Indiana reduced nitrate loads by 61% with a reduction in fertilizer N rates and a winter wheat cover crop following the corn crop.

No direct information on nitrate losses is available for living mulches, but it is assumed that the reduction of N losses would be similar to or greater than that of cover crops because the living mulches would be present all year. Even if the living mulches were legumes, we would assume that they would take up available soil N and reduce N leaching losses. If the living mulch is suppressed during the annual grain crop growing season, we would assume that N released from the living mulch or its residues would be taken up by the grain crop.

Perennial crops, like alfalfa, can result in extremely low nitrate concentrations and losses in drainage water, due to both nitrate uptake and water use. Compared with a continuous corn system, Randall et al. (1997) observed a 97% reduction in annual nitrate load with unfertilized (no N) alfalfa. Management of forage or pasture, however, using high rates of fertilizer or manure or intensive grazing may result in substantial nutrient losses. Additionally, killing, plowing down, or stresses, like drought, can cause substantial losses of N from legume perennial forages or pastures unless another crop or cover crop is present for N uptake. Thus, perennials can reduce nitrate losses substantially compared with annual grain crops, especially if no nitrogen fertilizer is applied, but the relative reduction depends to some extent on management.

Another way to consider the potential impact of cover crops, living mulches, and perennials may be to consider the cumulative impact that these practices could have in a more diversified and integrated agricultural system. A recent Minnesota study (Rolf et al., 2003; Rolf, 2005) examined how diversifying cropland with more living cover, perennials, and longer-season plants can impact water quality. Two side-by-side 160 acre fields were studied over three years. One field was planted to a corn-soybean rotation on 96% of the area. The other field had corn and soybeans on 46% of the area, and integrated various small grains (30% of total area), alfalfa (14%), and native grasses (10%) into the cropping system. Each field received some nitrogen each year, with the more diverse system receiving on average slightly more total nitrogen, all from organic sources. Over the three-year evaluation of the tile discharge, the more diverse system reduced average total annual nitrogen loss by 73% and stayed below the drinking water standard of 10 ppm.

All three of these practices should reduce losses of nutrients associated with soil erosion and surface runoff. In Iowa, rye and oat cover crops reduced rill erosion following soybean in a no-till system by 79% and 49%, respectively (Kaspar et al., 2001). We estimate that total P losses to surface waters, which are linked to sediment losses, might be reduced by a similar amount by cover crops. Sharpley and Smith (1991) summarized research on the effect of cover crops on total P losses and found that the reductions in total P losses ranged from 54% to 94% (table 10-2). They also pointed out, however, that the effects of cover crops on soluble P in runoff were more variable and did not always result in reductions. There is evidence that soluble P can be lost in runoff flowing over plant residues. However, on an annual basis, plant water use and infiltration would be expected to increase with cover crops, living mulches, and perennial crops, which should reduce the volume of runoff. Kaspar et al. (2001) found that a rye cover crop significantly increased infiltration and decreased runoff in one of three years, even under steady-state simulated rainfall conditions, which would negate the additional benefit of cover crop water use.

Table 10-1. Literature summary of percent reduction in nitrate-N leaching losses due to rye or ryegrass winter cover crops (adapted in part from Meisinger et al., 1991).

Reference	Location	Cover crop	Reduction in N Leaching
Morgan et al., 1942	Connecticut, U.S.	Rye	66%
Karraker et al., 1950	Kentucky, U.S.	Rye	74%
Nielsen and Jensen, 1985	Denmark	Ryegrass	62%
Martinez and Guirard, 1990	France	Ryegrass	63%
Staver and Brinsfield, 1990	Maryland, U.S.	Rye	77%
McCracken et al., 1994	Kentucky, U.S.	Rye	94%
Wyland et al., 1996	California, U.S.	Rye	65% to 70%
Brandi-Dohrn et al., 1997	Oregon, U.S.	Rye	32% to 42%
Ritter et al., 1998	Delaware, U.S.	Rye	30%
Kaspar et al., 2007	Iowa, U.S.	Rye	61%
Strock et al., 2004	Minnesota, U.S.	Rye	13%
Kladivko et al., 2004	Indiana, U.S.	Winter wheat + less fertilizer	61%

Table 10-2. Literature summary of percent reduction in total P losses in runoff due to barley, winter wheat, or legume winter cover crops (adapted from Sharpley and Smith, 1991).

Reference	Location	Cover Crop	Reduction in Total P Losses in Runoff
Angle et al., 1984	Maryland, U.S.	Barley	92%
Langdale et al., 1985	Georgia, U.S.	Rye	66%
Pesant et al., 1987	Quebec, Canada	Alfalfa/timothy	94%
Yoo et al., 1988	Alabama, U.S.	Wheat	54%

What is the predicted timing and "seasonality" of the reductions?

Most nitrate losses in agricultural drainage water occur during the periods of the year when the ground is not frozen, the soil is near saturation, and the annual grain crop is not actively growing. In the northern part of the region, this would be mostly from the spring thaw through mid-June. In much of the southern part of the region (Illinois and Missouri), nitrate losses would occur in late fall, winter, and early spring, when most of the drainage occurs.

For a cover crop, living mulch, or perennial crop to reduce losses of nitrate in drainage water compared to those of an annual grain crop system, these plants must be actively growing and taking up nutrients during the periods of the year when the ground is not frozen and an annual grain crop is not normally growing. For many winter-hardy small grains and cool-season perennial species, the most active period of growth occurs in late spring, although there is also some growth in the fall. For cool-season species that do not overwinter, such as oats, oilseed radish, and oriental mustard, substantial fall growth can occur if they are planted early enough. Even a relatively small amount of growth in the fall seems to reduce nitrate concentrations in the late fall and winter in areas where flow occurs during this time.

Reductions of P losses associated with sediment would most likely occur during the late winter and early spring, when runoff and erosion potential are greatest. Reductions in dissolved P losses associated with runoff would also likely occur in late winter and spring.

What is the degree of confidence of the estimations?

Confidence is high that cover crops will reduce nitrate losses when reasonable establishment and growth occurs. Magnitude of reductions is dependent on the growth of the cover crops, management of annual grain crops, and weather in a given year. In some years, there may be less reduction of nitrate losses because the cover crops do not grow very much or because there is not very much nitrate leaching even without cover crops. More information is needed on: range and feasibility of cover crops to the north and west; cover crops effects on P losses; reduction of nutrient losses by cover crops on a field, landscape, or watershed scale; cultivar or species selection; time of planting and termination; seeding rate; and effect of multispecies mixtures.

Confidence is reasonably high that living mulches will reduce nitrate losses if reasonable growth of the living mulch occurs and growth of the main crop is not reduced too much by the living mulch. The magnitude of reductions depends on the amount of growth of the living mulch when the annual grain crop is not taking up any or limited amounts of nutrients. In some years, growth of the living mulch may be limited or the living mulch could reduce the annual grain crop growth, both of which may reduce annual nutrient uptake and greatly lessen reductions in nitrate losses.

Confidence is extremely high that perennial crops would significantly reduce nitrate losses compared with an annual grain crop, if the perennial crop is not fertilized with N fertilizers or manures and is managed properly. If a perennial crop receives N fertilizer or manures at the same rates as an annual grain crop and is managed properly, the annual losses would still be expected to be less than that from the annual grain crop system because the perennial crop would take up water and nutrients over more of the year

(Crosson et al., 2007). The advantage of the extended uptake period of a perennial crop relative to an annual grain crop, however, may be negated or minimized by sandy soils that are particularly vulnerable to leaching or by large precipitation events.

Confidence is high that all three of these practices would substantially reduce P losses associated with sediment and runoff. The amount of reduction is dependent on the amount of growth, surface cover, annual water use, and anchorage of residues.

Important Factors

How do site conditions (soils, topography, hydrology, and climate) affect effectiveness in reducing nutrient losses?

While the effect of site conditions is still unknown based on research, we would speculate that reductions in nitrate losses may be greater for soils that are high in organic matter, at lower positions in the landscape, and relatively wet or poorly drained (provided cover crop, living mulch, or perennial growth is not limited by excessive water or that most N is not lost by denitrification). Because cover crops, living mulches, and perennials are very effective at reducing erosion, we would speculate that these practices would reduce losses or movement of P from upper and steeply sloped landscape positions or from areas of the field where overland water flow occurs. Climate differences across the Upper Mississippi River basin will affect the seasonality and the effectiveness of cover crops, living mulches, and perennials. All three practices will likely be more effective in reducing nitrate loads where the soils are not frozen the entire winter and where much of the annual drainage occurs throughout the fall, winter, and spring seasons, rather than in a short drainage period of March through June. Additionally, these practices will be more effective where the climate favors growth of these plants between late summer and early spring. For example, more cover crop growth would be expected in the southeastern part of the Upper Mississippi River basin, which is warmer and wetter from September to May, than in the northwestern part of this region, which is colder and drier from September to May. In Iowa, we developed an empirical relationship between oat cover crop fall growth and temperature and precipitation. We then used that equation and 40 years of climate data to predict the average fall growth across Iowa. Figure 10-4 shows that predicted oat cover crop fall growth varies from 900 to 400 kg ha^{-1} from southeastern Iowa to north central Iowa. Certainly, other factors will also limit growth, but in general we expect that this general climate trend for fall growth would hold and that trends for spring growth of rye would be similar. The effectiveness of perennial crops will depend on whether their periods of active growth and nutrient uptake coincide with the annual periods of water movement through the profile. For example, we speculate that a perennial forage crop consisting of warm-season grasses may not be very effective at reducing nutrient losses in the Upper Mississippi River basin because these grasses are not actively growing during the normal periods of annual drainage.

Length of the cover crop growing season will also determine the crop's effectiveness in reducing nutrient losses. In general, the longer the time between planting and termination of the cover crop, the greater the growth and nutrient uptake. Normally, cover crops are planted after harvest of a grain crop and terminated before planting of

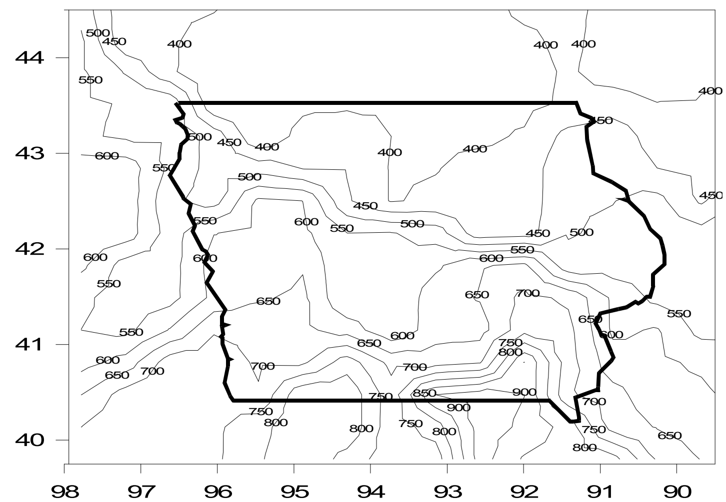

Figure 10-4. Contour map for predicted fall production of oat cover crop shoot dry matter (kg ha^{-1}) versus latitude and longitude in Iowa.

the next crop. This can result in relatively short cover crop growing seasons in the northern parts of the Upper Mississippi River basin. Feyereisen et al. (2003) used modeling to predict that in most years a rye cover crop planted in southern Minnesota on October 15 would reduce nitrate losses in drainage water by at least 25 kg ha^{-1} if terminated on May 1 and by at least 36 kg ha^{-1} if terminated on June 1. Thus, extending the cover crop growing season by developing new, less expensive management strategies to establish cover crops before main crop harvest or to terminate cover crops after main crop planting would improve the effectiveness of cover crops in northern parts of the Upper Mississippi River basin.

How does a range of weather conditions affect effectiveness in reducing nutrient losses?

Weather affects the effectiveness of cover crops, living mulches, and perennial crops in reducing nitrate losses in two ways. First, if weather conditions are such that N mineralization and nitrate leaching are not favored, then these practices will not show much of an advantage over conventional annual cropping practices. For example, in Iowa in some years, little if any nitrate leaching occurs, because the soil profile was not recharged with water between harvest and planting of the annual grain crops, because the soil surface layers froze before recharge occurred, or because the soil remained so cold that N mineralization was greatly reduced. It is assumed that these scenarios would be more likely to occur in the colder and more northern portions of the region and in the drier and more western portions. Second, if weather conditions limit the establishment or growth of cover crops, living mulches, or perennials, then their effectiveness will be reduced. Dry conditions, very cold conditions, or early freezing of the soil surface in the fall or late thawing in spring will limit cover crop growth, but will also limit N mineralization and nitrate leaching.

Limitations

In terms of physical constraints, what percent of crop acres could benefit from this practice?

We estimate that cover crops would show some reduction in nitrate losses on 70% to 80% of all corn and soybean acres. Establishment on some acres would be limited because of lack of rainfall in some years, late planting because of harvest delays, and poor soil conditions at time of planting. Reductions in nitrate loss and cover crop growth would be diminished in the northern part of the region because of cold temperatures and frozen soil between main crops and because of less growth of the cover crops. Benefits and cover crop growth would also be limited in the western part of the region (unless irrigated) because of water limitations for cover crop growth and nitrate leaching. Crop acres with more diverse rotations than a typical corn-soybean rotation may have even better opportunities for cover crops. Short-season crops like vegetables, sweet corn, corn silage, seed corn, and winter wheat would be particularly suited to cover crops.

Living mulch systems are not ready for widespread adoption at this time in the Upper Mississippi River basin in annual grain crop rotations, but they have potential to be an important management option in the future especially if stover is harvested. Living mulch systems can and should be used in orchards, vineyards, and tree plantations.

In general, perennial crops can be physically grown almost anywhere in the region, but realistically their adoption is limited by availability of on-farm utilization, markets, product demand, processing facilities, and infrastructure. Markets exist for some perennial crops or their products, such as pasture-raised beef, dairy products, and timber. Forage crops can be widely marketed, but even those markets could be quickly saturated if a considerable number of acres were converted from corn and soybean production to forage production. Increased demand for grass-fed meat and dairy products and for bioenergy produced by direct combustion or through cellulosic ethanol production could rapidly open up new markets. We speculate that up to 20% of corn and soybean acres could be converted to pasture, forages, bioenergy crops, and tree crops, if development of processing facilities, infrastructure, and markets were encouraged and supported.

In terms of cost constraints, what are the annualized costs?

Cost estimates were prepared using the 2005 Iowa Farm Custom Rate Survey (Edwards and Smith, 2005) and central Iowa and/or internet prices for seed and glyphosate. Cost estimates for living mulch-alfalfa establishment year (approximately every third year); no-till:

- Alfalfa seed at \$3.00 lb^{-1}, 0.66 of surface area planted at 10 lb ac^{-1}.
- Custom rate for planting alfalfa with grain drill = \$9.65 ac^{-1}.
- Generic glyphosate at \$20.00 gal^{-1}, two applications per year applied on 0.33 of surface area at rate of 0.33 qt ac^{-1} = \$3.32 ac^{-1} $year^{-1}$.
- Custom rate for spraying herbicide two applications per year at \$4.75 ac^{-1} $appl^{-1}$ = \$9.50 ac^{-1} $year^{-1}$.
- Chopping/mowing living mulch two operations per year at \$7.15 ac^{-1} op^{-1} = \$14.30 ac^{-1} $year^{-1}$.

- Total = \$121.05 ac^{-1} in three years = \$40.35 ac^{-1} $year^{-1}$.

Estimates are based on custom rates for field operations, which include fuel, labor, and machinery costs. Tillage system is assumed to be no-till. Additional assumptions are that alfalfa would need to be reestablished every third year, which may be too conservative, and that other living mulch species would have similar costs. Estimates do not include any potential yield decreases of annual grain crops caused by living mulches.

Cost estimates for establishing a perennial alfalfa crop (approximately every third year; may last longer), assuming no-till for establishment and not including harvest or other management costs in succeeding years:

- Alfalfa seed at \$3.00 lb^{-1}, planted at 15 lb ac^{-1}.
- Custom rate for planting alfalfa with no-till grain drill = \$9.65 ac^{-1}.
- Generic glyphosate at \$20.00 gal^{-1}, applied preplant at 1 qt ac^{-1} = \$5.00 ac^{-1}.
- Custom rate for spraying herbicide = \$4.75 ac^{-1}.
- Total = \$64.60 in three years = \$21.53 $year^{-1}$.

Although costs for establishment of a perennial alfalfa crop are presented here, because alfalfa is replacing the annual grain crop, the difference between the annual return for selling the alfalfa and the return for the "normal" annual grain crop could be considered the cost of practice. In some cases, including alfalfa in a rotation can be more profitable than a corn-soybean rotation (Singer et al., 2003). In addition, the costs presented here are for no-till, and alfalfa is commonly established following tillage.

Cost estimates for a rye cover crop in no-till corn-soybean rotation:

- Rye seed (bagged) at \$6.00 bu^{-1}, planted at 1 bu ac^{-1}.
- Custom rate for planting rye with no-till grain drill = \$9.65 ac^{-1}.
- Generic glyphosate to kill rye at \$20.00 gal^{-1}, applied at 1 qt ac^{-1} = \$5.00 ac^{-1}.
- Custom rate for spraying herbicide = \$4.75 ac^{-1}.
- Total = \$25.40 $year^{-1}$.

Estimates do not include any potential yield decreases of corn crop following a rye cover crop. Costs can be reduced by using operator-owned equipment, and by assuming that glyphosate application is a preplant application for the no-till system.

What are the costs per pound reduction of nitrate losses?

Based on Iowa and Minnesota data for rye cover crops, if we assume a range of 20 to 50 kg ha^{-1} reduction in nitrate-N loss (equivalent to 17.8 to 44.6 lbs ac^{-1}), then the costs per pound of reduction would range from \$1.42 to \$0.57 per pound for cover crops, from \$1.21 to \$0.48 per pound for perennial alfalfa, and from \$2.27 to \$0.90 per pound for an alfalfa living mulch. Costs per pound will vary and may be very high in years when little or no nitrogen is lost or when the cover crop, living mulch, or perennial crop does not establish or fails. In addition, actual farmer costs are expected to be lower than custom rates on which our calculations are based, and other environmental benefits are not credited to the practices.

Current or potential perennial-based systems may be economically viable through on-farm utilization or the sale of the perennial crop or its by-products. In that case, there would be no cost for reduction of nitrate losses. Alternatively, cost of a perennial-based system could be calculated as the return of the local annual grain crop system minus the return of the perennial-based system. In some situations, however, the perennial-based system may have greater or equal returns. Singer et al. (2003) found equal or greater returns for a five-year rotation, which included three years of alfalfa, than for a corn-soybean rotation. Additionally, a University of Wisconsin study (Kriegl and McNair, 2005) documented a 90% higher net income per cow from grazing-based dairy operations than from confinement dairy farms. Crossan et al. (2007) used modeling to show that conversion of a beef-producing farm from a corn-based production system to a perennial grassland system would provide both economic and environmental benefits. Additionally, increasing consumer interest in 100% grass-fed meat may improve the profitability of forage or pasture perennial crops. The economics of other potential perennial systems, such as biomass energy or perennial grain crops, has yet to be determined.

Costs per pound of reduction of nitrate losses do not consider the other benefits of cover crops, living mulches, and perennial crops. These systems would result in additional agroecosystem benefits, such as increased soil organic matter, reduced surface runoff of precipitation, increased wildlife diversity, reduced erosion, and increased diversity of soil organisms.

In terms of production risks, will crop yields be reduced or become more variable?

Corn yields may be reduced following winter-hardy small grain cover crops that are killed immediately before corn planting (Johnson et al., 1998). We believe that this yield reduction is similar to the yield reduction that occurs when corn follows corn. Although not completely understood, corn yield reductions following a small grain cover crop are probably caused by a combination of factors, including nutrient immobilization, disease organisms, insect pests, low soil water content, and plant growth inhibitors released from decomposing plant residues. Yield reduction can be minimized by killing small grain cover crops 10 to 14 days before corn planting and using starter fertilizer. Corn yields following an oat cover crop, which dies when the ground freezes in the fall, or a legume cover crop are not reduced. Soybean yields are not reduced following any small grain cover crop unless low soil water content limits soybean germination and emergence (Strock et al., 2004). There is also a risk that the cover crop will not be completely killed the first time it is sprayed or tilled in spring. This would then require additional operations and may increase the risk of a yield reduction in the cash crop.

Living mulches can reduce corn and soybean yields by competing for water and nutrients during the growing season if they are not sufficiently suppressed by management or if the growing season is abnormally hot and dry.

Perennial crops such as alfalfa or forage or hay crops, which are replacements for the annual grain main crops, have somewhat different production risks than corn and soybean. Too much summer rainfall during drying of the hay is one such risk. Tree crops or woody perennials have different pests and in general spread weather-related risks over many years. Fruit or nut crops can be highly susceptible to late spring frosts.

Are there other limitations, such as negative attitudes, lack of knowledge, or additional equipment needed?

Many producers are not familiar with cover crops, living mulches, or perennials. Living mulch systems are not widely used with annual grain crops, due to lack of knowledge about how to manage these systems to reduce living mulch competition with the grain crop and lack of shade-tolerant living mulch species or cultivars that will survive under the main crop canopy.

A grain drill is needed for planting many cover crop, living mulch, and perennial species. Perennial forage or hay crops require equipment (mowers, rakes, and balers) not required for corn and soybean production.

Timeliness of cover crop and living mulch field operations in spring and fall will be limited by weather/soil conditions and competition for machinery and labor because of field operations associated with planting and harvesting of annual grain crops.

These systems are more complicated to manage and implement than some other practices to reduce nitrate losses such as reducing N fertilizer rates and applying N fertilizer in the spring rather than the fall. The additional management and risk are considered “a hassle” by many farmers.

Most producers in the Upper Mississippi River basin would not see an immediate monetary benefit or reduction in costs from including cover crops and living mulches in their farming systems and would have increased cost and labor to implement these practices. In other words, farmers would not receive any monetary benefit or compensation for using these practices to reduce nutrient losses to water sources. In addition:

- There are limited markets for perennials such as harvested forages, including alfalfa, orchardgrass, red clover, and smooth bromegrass.
- There is a great need for development of new perennial crops or new uses for well-known perennials (e.g., bioenergy).
- There are limited seed sources and adapted cultivars or genotypes available for cover crops, living mulches, and some perennials for use in the Upper Mississippi River basin. To our knowledge, few available cultivars have been bred specifically for use as cover crops or living mulches in colder climates.
- Cover crops, living mulches, and perennial crops do not qualify for the government risk protections that are provided to program crops under federal farm policy.

Finally, there is a great need to quantify the nutrient loss reductions of these systems under a range of locations and growing conditions. The nutrient losses of managed pastures need to be compared not only with the nutrient losses of corn-soybean annual grain crop systems, but also with nutrient losses from a farming system with confined beef or dairy cattle.

Other Issues

Are there any common misconceptions about this practice that need to be corrected?

There may be a misconception that pasture- or forage-based systems always have nutrient loss rates much less than annual grain crops because they are perennial. However, management of these systems for high productivity, mismanagement, or application of high rates of fertilizer or manure may result in substantial nutrient losses.

There may be a misconception that pasture- or forage-based systems are always less profitable than annual grain crop systems.

There may be a misconception that cover crops, living mulches, and perennial crops will always result in substantial reductions in nutrient losses. In some years, nutrients losses may be very low without these practices because of weather, management, soils, and main crop growth. Conversely, in some situations or combinations of timing, soils, weather, management, or storm events, these practices will be overwhelmed by an influx of water, resulting in substantial percolation and runoff at times when substantial concentrations of nutrients are susceptible to losses. Additionally, in some years, the cover crop, living mulches, and perennial crops may fail to establish or grow poorly.

There may be a misconception that fertilizer management alone will reduce nutrient losses from agricultural systems to environmentally acceptable levels and that cover crops, living mulches, and perennials are not needed. Because substantial amounts of nutrients originate from soil mineralization and decomposition of plant residues, and because even optimum economic rates result in substantial nutrient losses, fertilizer management alone will not eliminate nutrient contamination of surface waters.

There may be a misconception that one management practice can be used to address nutrient losses to surface waters, when in reality a combination of practices will be needed to effectively address this problem.

Are there any potential positive or negative effects on other resources (e.g., soil, air, wildlife)?

Cover crops, living mulches, and perennial crops have the potential to increase soil organic matter and soil quality. These systems:

- May also reduce some pest and disease pressures (nematodes, disease, insects, and weeds) but may also increase others (rodents, insects, and weeds).
- Increase plant diversity and provide food and cover for wildlife.
- Reduce wind and water erosion, in-field relocation of topsoil, and sediment load to surface waters.
- May improve water infiltration and help to remove excess water from the soil.
- May reduce surface runoff, concentrated or channel water flow, residue transport, and accumulation of water in low areas of fields.
- Have the potential to increase carbon sequestration in soils.

The ecological benefits provided by these management systems could be used as the basis for federal agricultural payments to producers that would not conflict with World Trade Organization guidelines.

What new information or research is needed to enhance the practice and accurately assess its benefits?

Research is needed on adaptation of these systems to more northerly climates including: better adapted cultivars or species, strategies for quick establishment in fall, and consistent control in spring. In addition:

- Information is needed on the potential geographic range of these practices.
- Adaptation of water quality models to include cover crops, living mulches, and perennials is needed to estimate environmental benefits of these practices.
- Information is needed on when to kill cover crops to optimize N uptake and N release for the cash crop, given that weather is variable and unpredictable.
- Information is needed on long-term cycling and balance of N, P, C, and other nutrients in these systems, and whether N, P, and K fertilizer rates can be reduced in future due to improvements in soil organic matter and nutrient cycling.
- Research is needed on management strategies to use cover crops and living mulches to trap N from manure application and recycle the N at an appropriate time for the next crop.
- Screening, selection, and breeding programs are needed for new cultivars, genotypes, and species for use as cover crops, living mulches, or perennials.
- Discovery and development of new oil, fiber, starch, or chemical products derived from perennial plants are needed.
- New production, harvesting, transporting, and processing technologies are needed for perennial crops.
- Markets for products of perennial crops must be developed, and existing markets must be strengthened and supported.
- The economic viability of these systems needs to be re-evaluated in response to dramatic increases in fuel and energy costs.
- Guidelines are needed for site/soil/landscape specific application of these practices to target areas in fields susceptible to nutrient and sediment loss.
- Research is needed on minimizing the impacts on growth and yield of annual grain crops by cover crops and living mulches.
- Research is needed on the nutrient losses from intensively managed pastures and hay fields.
- Additional research is needed on determining the effect of cover crops, living mulches, and perennial crops on P losses.
- New management practices are needed to reduce the costs of implementing cover crops and living mulches.
- Research is needed to evaluate cover crops and living mulches for biosuppression of weeds, nematodes, insects, and diseases.
- Research is needed to evaluate cover crops, living mulches, and perennials for low-input systems.
- New strategies are needed for dissemination of information concerning these systems to overcome cultural and societal reluctance in both rural and urban populations to implement and accept these systems.
- Quantification of the direct and indirect ecological benefits of these systems in diverse locations over a number of years is needed.
- Watershed-scale implementation projects are needed to access the potential for larger-scale outcomes from these practices on water quality.

- Research is needed to understand placement or site-specific application of these practices within landscapes or watersheds to provide the maximum environmental benefits.
- Research and selection of living mulch species and varieties are needed to find living mulch plants that are tolerant of shading under main crop canopies.
- Research and development are needed for seed coatings or treatments to allow cover crop seeds to be planted earlier during the main crop growing season and then germinate shortly before main crop maturity.

Interpretative Summary

Cover Crops

Site conditions: Cover crops may be less effective in the northern parts of the Upper Mississippi River basin because cold temperatures in late fall, winter, and spring will limit cover crop establishment and growth. Sites or field areas without significant fall, winter, and spring losses of sediment, nitrate, or phosphorus because of frozen ground, drier conditions, soils, or topography may not show water quality benefits from cover crops.

Water quality improvement: Reductions in nitrate load observed with a cover crop range from 13% in Minnesota to 94% in Kentucky. Reductions in total P losses with cover crops ranged from 54% to 94%.

Cost: Based on custom rates and bagged seed, establishment of a rye cover crop may cost up to \$25 ac^{-1}, and costs per pound of N loss prevented may be \$1.42 to \$0.57 lb^{-1}.

Extent of area: We estimate that cover crops would show some reduction in nitrate losses on 70% to 80% of all corn and soybean acres in the Upper Mississippi River basin.

Limitations for adoption: Cover crops require time and money to establish and do not provide short-term economic returns. Many producers are not familiar with cover crops and their management.

Impact on other resources: Corn yields may be reduced following winter-hardy small grain cover crops that are not killed until immediately before corn planting. This yield reduction can be managed by killing the cover crop 10 to 14 days before corn planting. In general, soybean yields are not affected by cover crops, if managed properly.

Living Mulches

Site conditions: Living mulches may be less effective, more difficult to maintain, and will compete more with the main crop in the western parts of the region because of increasing frequency of drought and reduced precipitation during the growing season.

Water quality improvement: No direct information on reduction of nitrate or phosphorus losses is available for living mulches, but it is assumed that if the living mulch is successfully established and maintained, then the reduction of N and P losses would be similar to or greater than that of cover crops because the living mulches would be present all year, rather than part of the year like cover crops.

Cost: Establishment of an alfalfa living mulch may cost up to \$40.35 ac^{-1} year^{-1}, and costs per pound of N loss prevented may be \$2.27 to \$0.90 lb^{-1}. We expect these costs to decrease as management improves.

Extent of area: Currently, living mulches are not ready for widespread adoption in grain cropping systems, but they may be in the future. Living mulches can and should be used in orchards, vineyards, and tree plantations.

Limitations for adoption: Information is needed on how to manage living mulch systems to reduce competition with the grain crop. Cultivars or genotypes of perennial forage species suitable for use as living mulches are not available.

Impact on other resources: Living mulches can reduce corn and soybean yields by competing for water and nutrients during the growing season if they are not sufficiently suppressed by management or if the growing season is abnormally hot and dry.

Perennial Crops

Site conditions: Perennial crops should be effective at reducing nutrient losses throughout the region.

Water quality improvement: Nitrate and phosphorus losses with perennials is assumed to be less than or equal to that of cropping systems with grain crops and cover crops. Nutrient loss reductions relative to grain cropping systems can be above 90%, but they are partly dependent on fertility management of the perennial.

Cost: Costs for establishment of perennials are relatively high, but unlike cover crops and living mulches, perennials produce significant economic returns. Because they replace the annual grain crop, it is more appropriate to compare their returns with those of grain cropping systems that are appropriate for a particular location.

Extent of area: Perennial crops could be adopted throughout the region, but are limited by demand, processing facilities, infrastructure, and markets. We speculate that 20% to 30% of corn and soybean acres could be converted to perennial crops, if infrastructure, processing facilities, and markets were encouraged and supported.

Limitations for adoption: Perennial crops are limited by demand, processing facilities, infrastructure, and markets. There is a great need for development of new perennial crops or new uses for well-known perennials.

Impact on other resources: By removing land from corn and soybean production, adoption of perennials may positively influence grain prices.

References

Angle, J. S., G. McClung, M. C. McIntosh, P. M. Thomas, and D. C. Wolf. 1984. Nutrient losses in runoff from conventional and no-till watersheds. *J. Environ. Qual.* 13: 431-435.

Baker, J. L., K. L. Campbell, H. P. Johnson, and J. J. Hanway. 1975. Nitrate, phosphorus, and sulfate in subsurface drainage water. *J. Environ. Qual.* 4: 406-412.

Brandi-Dohrn, F. M., R. P. Dick, M. Hess, S. M. Kauffman, D. D. J. Hemphill, and J. S. Selker. 1997. Nitrate leaching under a cereal rye cover crop. *J. Environ. Qual.* 26: 181-188.

Crosson, P., C. A. Rotz, and M. A. Sanderson. 2007. Conversion from corn to grassland provides economic and environmental benefits to a Maryland beef farm. *Forage and Grazinglands* doi:10.1094/FG-2007-0119-01-RS. Available at: www.plantmanagementnetwork.org/sub/fg/research/2007/beef/. Accessed 26 Jan. 2007.

Dinnes, D. L., D. L. Karlen, D. B. Jaynes, T. C. Kaspar, J. L. Hatfield, T. S. Colvin, and C. A. Cambardella. 2002. Nitrogen management strategies to reduce nitrate leaching in tile-drained

Midwestern soils. *Agron. J.* 94: 153-171.

Edwards, W., and D. Smith. 2005. 2005 Iowa Farm Custom Rate Survey (FM 1698). Ames, Iowa: Iowa State University Extension. Available at: www.extension.iastate.edu/. Accessed 16 Aug. 2005.

Feyereisen, G. W., A. Mendez, G. R. Sands, B. N. Wilson, P. M. Porter, and J. S. Strock. 2003. Probabilistic assessment of the potential for winter cereal rye to reduce field nitrate-nitrogen loss in southwestern Minnesota. ASAE Paper No. 032323. St. Joseph, Mich.: ASAE.

Gast, R. G., W. W. Nelson, and G. W. Randall. 1978. Nitrate accumulation in soils and loss in tile drainage following nitrogen application to continuous corn. *J. Environ. Qual.* 7: 258-262.

Goolsby, E. A., W. A. Battaglin, G. B. Lawrence, R. S. Artz, B. T. Aulenbach, R. P. Hooper, D. R. Keeney, and F. J. Stensland. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya river basin. Topic 3. Report for the Integrated Assessment of Hypoxia in the Gulf of Mexico. Decision Analysis Series No. 17. Silver Spring, Md.: NOAA Coastal Ocean Program.

Hartwig, N. L., and H. U. Ammon. 2002. Cover crops and living mulches. *Weed Sci.* 50: 688-699.

Jaynes, D. B., T. S. Colvin, D. L. Karlen, C. A. Cambardella, and D. W. Meek. 2001. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. *J. Environ. Qual.* 30: 1305–1314.

Johnson, T. J., T. C. Kaspar, K. A. Kohler, S. J. Corak, and S. D. Logsdon. 1998. Oat and rye overseeded into soybean as fall cover crops in the upper Midwest. *J. Soil Water Conserv.* 3: 276-279.

Karraker, P. E., C. E. Bortner, and E. N. Fergus. 1950. Nitrogen balance in lysimeters as affected by growing Kentucky bluegrass and certain legumes separately and together. Bulletin 557. Lexington, Ky.: Kentucky Agricultural Experiment Station.

Kaspar, T. C., D. B. Jaynes, T. B. Parkin, and T. B. Moorman. 2007. Rye cover crop and gramagrass strip effects on NO_3 concentration and load in tile drainage. *J. Environ. Qual.* 36: 1503-1511.

Kaspar, T. C., J. K. Radke, and J. M. Laflen. 2001. Small grain cover crops and wheel traffic effects on infiltration, runoff, and erosion. *J. Soil Water Conserv.* 56: 160-164.

Kladivko, E. J., J. R. Frankenberger, D. B. Jaynes, D. W. Meek, B. J. Jenkinson, and N. R. Fausey. 2004. Nitrate leaching to subsurface drains as affected by drain spacing and changes in crop production system. *J. Environ. Qual.* 33: 1803-1813.

Kriegl, T., and R. McNair. 2005. Pastures of plenty: Financial performance of Wisconsin grazing dairy farms. Madison, Wisc.: University of Wisconsin. Available at: www.cias.wisc.edu/pdf/grzgfin.pdf. Accessed 13 Dec. 2005.

Langdale, G. W., R. A. Leonard, and A. W. Thomas. 1985. Conservation practice effects on phosphorus losses from southern Piedmont watersheds. *J. Soil Water Conserv.* 40: 157-160.

Martinez, J., and G. Guirard. 1990. A lysimeter study of the effects of a ryegrass catch crop during a winter wheat/maize rotation on nitrate leaching and on the following crop. *J. Soil Sci.* 41: 5-16.

McCracken, D. V., M. S. Smith, J. H. Grove, C. T. MacKown, and R. L. Blevins. 1994. Nitrate leaching as influenced by cover cropping and nitrogen source. *SSSA J.* 58: 1476-1483.

Meisinger, J. J., W. L. Hargrove, R. L. Mikkelsen, J. R. Williams, and V. W. Benson. 1991. Effects of cover crops on groundwater quality. In *Cover Crops for Clean Water: Proc. Intl. Conference*, 57-68. W. L. Hargrove, ed. Ankeny, Iowa: Soil and Water Conservation Society of America.

Morgan, M. F., H. G. M. Jacobson, and S. B. LeCompte Jr. 1942. Drainage water losses from a sandy soil as affected by cropping and cover crops. Bulletin 466. New Haven, Conn.: Connecticut Agricultural Experiment Station.

Nielsen, N. E., and H. E. Jensen. 1985. Soil mineral nitrogen as affected by undersown catch crops. In *Assessment of Nitrogen Fertilizer Requirement: Proc. NW-European Study Ground for the Assessment of Nitrogen Fertilizer Requirement*. Haren, The Netherlands: Netherlands Fertilizer Institute.

Pesant, A. R., J. L. Dionne, and J. Genest. 1987. Soil and nutrient losses in surface runoff from conventional and no-till corn systems. *Canadian J. Soil Sci.* 67: 835-843.

Randall, G. W., D. R. Huggins, M. P. Russelle, D. J. Fuchs, W. W. Nelson, and J. L. Anderson. 1997. Nitrate losses through subsurface tile drainage in Conservation Reserve Program, alfalfa, and row crop systems. *J. Environ. Qual.* 26: 1240-1247.

Ritter, W. F., R. W. Scarborough, and A. E. M. Chirnside. 1998. Winter cover crops as a best management practice for reducing nitrogen leaching. *J. Contam. Hydrol.* 34: 1-15.

Rolf, K. 2005. Evaluation of alternative versus conventional farming systems impacts on subsurface drainage flow and water quality. MS thesis. St. Paul, Minn.: University of Minnesota.

Rolf, K. A., J. S. Strock, D. J. Mulla, and P. H. Gowda. 2003. Implications of conventional and organic farming practices on nonpoint-source pollution and TMDLs. In *Total Maximum Daily Load (TMDL) Environmental Regulations II: Proc. 2nd Conf. on Watershed Management to Meet Emerging TMDL Environmental Regulations*, 473-476. St. Joseph, Mich.: ASAE.

Sharpley, A. N., and S. J. Smith. 1991. Effects of cover crops on surface water quality. In *Cover Crops for Clean Water: Proc. Intl. Conference*, 41-49. W. L. Hargrove, ed. Ankeny, Iowa: Soil and Water Conservation Society of America.

Singer, J. W., C. A. Chase, and D. L. Karlen. 2003. Profitability of various corn, soybean, wheat, and alfalfa cropping systems. *Crop Management* doi:10.1094/CM-2003-0130-01-RS.

Singer, J., T. Kaspar, and P. Pedersen. 2005. Small grain cover crops for corn and soybean. Ames, Iowa: Iowa State University Extension. Available at: http://extension.agron.iastate.edu/soybean/documents/PM1999._covercrops.pdf. Accessed 13 Dec. 02005.

Snapp, S. S., S. M. Swinton, R. Labarta, D. Mutch, J. R. Black, R. Leep, J. Nyiraneza, and K. O'Neil. 2005. Evaluating cover crops for benefits and costs and performance within cropping system niches. *Agron. J.* 97: 322-332.

Staver, K. W., and R. B. Brinsfield. 1990. Patterns of soil nitrate availability in corn production systems: Implications for reducing groundwater contamination. *J. Soil Water Conserv.* 45: 318-323.

Stout, W. L., S. L. Fales, L. D. Muller, R. R. Schnabel, and S. R. Weaver. 2000. Water quality implications of nitrate leaching from intensively grazed pasture swards in the northeast U.S. *Agric. Ecosyst. Environ.* 77: 203-210.

Strock, J. S., P. M. Porter, and M. P. Russelle. 2004. Cover cropping to reduce nitrate loss through subsurface drainage in the northern U.S. Corn Belt. *J. Environ. Qual.* 33: 1010-1016.

Sullivan, P., and S. Diver. 2001. Overview of cover crops and green manures. Fayetteville, Ark.: University of Arkansas, Appropriate Technology Transfer for Rural Areas (ATTRA). Available at: www.attra.org/attra-pub/covercrop.html. Accessed 16 Aug. 2005.

Sustainable Agriculture Network. 2007. Managing cover crops profitably. Sustainable Agriculture Network. Beltsville, Md.: USDA-ARS, National Agricultural Library. Available at: www.sare.org/publications/covercrops/covercrops.pdf. Accessed 9 Oct. 2007.

UC SAREP. 2002. Cover crop resource page. Davis Cal.: University of California, Sustainable Agriculture Research and Education Program. Available at: www.sarep.ucdavis.edu/ccrop. Accessed 16 Aug. 2005.

Wilson, R., J. Smith, and R. Moomaw. 1993. Cover crop use in crop production systems NebGuide. G93-1146-A. Lincoln, Neb.: University of Nebraska Cooperative Extension, Institute of Agriculture and Natural Resources. Available at: www.ianr.unl.edu/pubs/fieldcrops/g1146.htm. Accessed 16 Aug. 2005.

Wyland, L. J., L. E. Jackson, W. E. Chaney, K. Klonsky, S. T. Kolke, and B. Kimple. 1996. Winter cover crops in a vegetable cropping system: Impacts on nitrate leaching, soil water, crop yield, pests and management costs. *Agric. Ecosys. Environ.* 59: 1-17.

Yoo, K. H., J. T. Touchton, and R. H. Walker. 1988. Runoff, sediment, and nutrient losses from various tillage systems of cotton. *Soil Tillage Res.* 12: 13-24.

Pp. 149-158 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008.
Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop. St. Joseph, Michigan: ASABE.

Sustaining Soil Resources While Managing Nutrients

11

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The focus of most nutrient management studies has logically been on economic viability and water quality. In this chapter, we examine the wider issue of sustaining soil resources when developing practices designed to improve water quality. Sustainability when applied to crop production is often an emotionally charged word that has been used in many contexts. It has been used interchangeably with terms such as low-input sustainable agriculture, alternative agriculture, organic farming, regenerative farming, best management practices, and maximum economic yield (Keeney, 1990). Here we wish to use it in a more formal sense as defined by the 1987 Iowa Groundwater Protection Act. The Act defined sustainable agriculture as "the appropriate use of crop and livestock systems and agricultural inputs supporting those activities, which maintain economic and social viability whereas preserving the high productivity and quality of Iowa's land." Similarly, the National Food, Agriculture, Conservation, and Trade Act of 1990, Section 1603, defines sustainable agriculture as "an integrated system of plant and animal production practices that will, over the long term, enhance environmental quality and the natural resource base upon which the agricultural economy depends." Thus, to be sustainable in the Corn and Soybean Belt, a farming system not only needs to be economically viable and protect water quality but also must preserve or enhance the soil resource that makes the highly productive agriculture possible.

How do various management practices affect yield, water quality, and the soil resource? When discussing soil productivity, we are primarily concerned with maintaining or building soil organic matter (SOM) within the topsoil, because SOM provides much of the nutrient reservoir (fertility); determines physical characteristics that control infiltration, aeration, and aggregation associated with good soil tilth; and provides the energy or substrate for biological processes. SOM can be lost from a soil through two primary mechanisms: soil erosion and *in situ* decomposition. Soil erosion is a natural process, but enhanced erosion has been a consequence of agriculture from its inception. Sediment derived from soil erosion is the primary pollutant of surface waters today and a major cost to society. As the topsoil erodes, the sediment is enriched in SOM and the nutrients required for crop production. Current soil conservation programs are targeted towards reducing soil erosion to the tolerable, or *T*, level as determined by the Revised Universal Soil Loss Equation, and all management practices targeted for nutrient loss reduction must also keep soil loss below *T*. As topsoil protection from erosion is covered elsewhere in this book (see chapter 9), we will not examine it further here except to point out that achieving *T* alone is not sufficient for sustaining soil resources. Instead, we will concentrate on the second loss mechanism for SOM: decomposition.

SOM is composed of many different organic compounds, ranging from fresh crop residues through their various decomposition products to stable humus that is only very slowly decomposed to CO_2 and soluble compounds that can be leached from the soil. On average, soil humus contains about 5.6% N and 56% C, for a C:N ratio of 10 (Waksman, 1938). Thus, we can speak interchangeably about either the soil organic carbon (SOC) pool or the total soil organic nitrogen (TN) pool when discussing SOM. To maintain SOC levels, the long-term input of C or N into the humus pool must equal the long-term loss. Therefore, long-term gains and losses of either C or N from the organic soil fractions can be used to monitor changes in SOC.

Soil Nitrogen Mass Balance

Nitrogen mass balance calculations have been made at the field and watershed scales. At the field scale, studies by Karlen et al. (1998), Drinkwater et al. (1998), Jaynes et al. (2001), and Webb et al. (2004) have all shown agricultural systems with lower N inputs compared to N outputs. There have also been numerous watershed-scale studies (David et al., 1997; Burkart and James, 1999; Goolsby et al., 1999; Libra et al., 2004), with studies using the more conservative net N input approach also finding a total N outputs exceeding total N inputs (McIsaac and Hu, 2004). All of these studies use the conservation of mass to compute the N balance, i.e.:

$$\text{inputs} - \text{outputs} - \Delta\ \text{soil residual mineral N} = \text{residual} \qquad \text{(eq. 1)}$$

A residual > 0 indicates that inputs of N exceed losses from the field and that N is available for other processes, such as increasing SOM. A residual < 0 indicates that inputs do not balance outputs and that additional N must be coming from sources not included in the inputs, such as decomposition of SOM, to account for the observed losses. A residual = 0 indicates that N inputs and outputs from the field are in balance, and therefore the production system is sustainable from a SOM perspective.

For a typical corn-soybean rotation, inputs of N to a field include the application of fertilizer and manure, N contained in rain and dry deposition, and N fixed by soybean. Outputs from the field include N removed with the grain harvest and NO_3 in deep drainage and runoff. Approaches for estimating each input and output are summarized below.

Fertilizer and Manure Inputs

Fertilizer inputs are usually known and include N applied through sources such as anhydrous ammonia, urea-ammonium nitrate (UAN), ammonium nitrate, or ammonium sulfate and N associated with P fertilizers (e.g., mono-ammonium phosphate and diammonium phosphate). Manure inputs are based on total N at time of application minus a volatilization loss (Killorn and Lorimor, 2003) that depends on mode of application (i.e., injected, broadcast, etc).

Wet and Dry Deposition

Nitrogen supplied by precipitation can be estimated from measurements made by the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/). Across the Corn and Soybean Belt, average annual combined wet deposition of NO_3 and NH_4 ranges from 3.7 to 7.0 kg N ha^{-1} $year^{-1}$. For dry deposition, the approximation used by Goolsby et al. (1999) can be used where dry deposition equals 0.7× wet deposition.

Fixation

Nitrogen fixation by soybean ranges considerably (McIsaac et al., 2002) not only because of the plant, soil, and climatic factors involved, but also because fixation depends on the availability of soil N to the plant (Russelle and Birr, 2004). Estimates of fixation also vary because of differences associated with methods (i.e., fertilizer replacement or isotopic (^{15}N) uptake) used to estimate its contribution. Barry et al. (1993) found a linear relationship between soybean grain yield (Mg ha^{-1}) and N fixed (kg ha^{-1}) as:

$$N_{fixed} = 81.1 \times \text{yield} - 98.5$$

This compares to a more conservative estimate when soybean yields are >2.1 Mg ha^{-1}, as used by McIsaac et al. (2002):

$$N_{fixed} = 33.4 \times \text{yield}$$

Grain Removal

The amount of N exported in grain depends on both the yield and protein content. Protein content for corn typically ranges from 60 to 90 g kg^{-1}, whereas in soybean it typically ranges from 336 to 375 g kg^{-1} (Russelle and Birr, 2004). Assuming a typical protein to N ratio of 6.25:1 (David et al., 1997), estimating the total N mass removed with corn or soybean is easily calculated. However, for other crops (e.g., wheat) the typical protein to N ratio would be 5.75 to 1.

Drainage

Measuring N losses in percolation below the root zone is often very difficult. However, in fields where subsurface tile has been installed to improve drainage, the volume of flow and N losses in tile water can be directly measured and accounts for most of the percolation of N below the root zone.

Runoff

Runoff losses of N can be measured at the edge of a field using flumes or other techniques to determine the volume of flow and from which samples can be collected to determine N concentrations. However, it is often safe to assume that very little N is lost in runoff, especially if injected or applied as a liquid (Gascho et al., 1998). Nitrate is very soluble in water and leaches below the soil surface at the start of each rainfall event. Therefore, it is generally not available for loss in surface runoff.

Weathering of the soil mineral fraction is not accounted for in this partial N balance, but this is generally considered to be trivial. Denitrification is not explicitly accounted for either, but this loss can be substantial in some locations and years. Unfortunately, it is extremely variable and difficult to estimate or measure accurately at the field scale (Parkin and Meisinger, 1989). Volatilization is also not accounted for, although such losses are generally minimal when N fertilizer is applied properly. Volatilization from manures is accounted for in the computation of N applied with manure inputs. N can also be lost directly from senescing plants (Francis et al., 1997), but the magnitude of this loss is variable and not well quantified for corn, soybean, or other crops. As presented, the partial N balance (eq. 1) does not include these unaccounted for loss pathways, but it is important to note that they all occur in the field to some extent, resulting in somewhat lower residual mass balances than computed in the next section.

Finally, the conversion of mineral N to organic N (immobilization) and the mineralization of N from SOM to mineral N are also not considered explicitly in the N mass balance. Instead, these processes are captured by the residual term. A positive residual indicates surplus N that would be available to build additional SOM. A negative residual indicates that N is being supplied from an unaccounted for source, most likely mineralization or loss of SOM.

Examples

To illustrate how N management affects the sustainability of soil resources and crop production, we will examine data from three N management studies conducted in Iowa.

Deficit Fertilization

In a study described in Jaynes et al. (2001), three N fertilizer rates were replicated three times in a producer's field planted to a corn-soybean rotation. Nitrogen was applied in the spring after corn emergence at three multiples (1×, 2×, and 3×) of a base or target rate of 67 kg N ha^{-1}. Corn yields ranged from 6.63 to 10.73 Mg ha^{-1} over the four-year study, with the economic optimum N rate being equal to about the 2× rate. Soybean yields were not affected by N application rate in the corn year and averaged 3.66 Mg ha^{-1}. By monitoring the tile drainage from each treatment plot, Jaynes et al. (2001) found that the annual flow-weighted NO_3 concentrations ranged from 11.4 mg L^{-1} for the 1× treatment to 18.8 mg L^{-1} for the 3× treatment. Using the maximum contaminant level (MCL) for NO_3 in drinking water determined by the USEPA, none of these N treatments could be characterized as sustainable from a water quality perspective, although lowering the N rate substantially lowered NO3 concentrations in the drainage water.

To compute the N mass balance, inputs from the N and P-K fertilizations were measured. Inputs from wet deposition were estimated using measured precipitation and NADP average NO_3 and NH_4 concentrations in precipitation for central Iowa. Dry deposition was estimated to be 0.7× wet deposition (Goolsby et al., 1999). Nitrogen fixed by soybean was estimated from the measured soybean yield and the relationship between yield and N fixed, as described by Barry et al. (1993). Grain removal of N was determined using measured grain yield and protein content. Drainage losses of NO_3 were computed using measured drainage volume and NO_3 concentration in the water from tile drains installed 1.2 m below the surface. Runoff losses of N were not measured but considered negligible since the field was nearly level (<1% slope) and runoff was observed only twice over four years. Changes in soil mineral N were measured every fall within the top 1.2 m by collecting cores, extracting and analyzing for NO_3 and NH_4.

The four-year average partial N balance for each N rate in this study is shown in table 11-1. Wet and dry deposition as well as N fixed by soybean was nearly identical for each treatment. N removed in grain harvest varied by about 15% due to treatment differences in corn yield. As shown for the NO_3 concentrations, the mass of N loss through drainage water was also a function of N rate, where the loss increased by approximately 64% as fertilization rates increased from 1× to 3×. Changes in runoff and residual soil mineral N were nominal. Summing the inputs and outputs for these treatments shows residual values of -55, -26, and 47 kg ha^{-1} for the three treatments. Residuals of <0 for the 1× and 2× treatments indicate that more N was being lost from

Table 11-1. Partial N mass balance for four-year rate study by Jaynes et al. (2001).

	N Inputs (kg ha^{-1})			N Outputs (kg ha^{-1})				
Fertilizer Rate	Total Fertilizer Applied	Total Wet and Dry Deposition	Total Fixed	Total Grain Removal	Total Drainage Loss	Total Runoff[a]	Change in Residual Mineral N	N Balance Residual
1×	144[b]	43	395	522	119	0	6	-55
2×	289	43	397	590	142	0	13	-26
3×	414	43	394	606	195	0	-7	47

[a] Not measured, but little runoff observed during the four-year period.
[b] Not exact multiples of 67 because of systematic over-application in the third year of the study.

those systems than was being applied. This missing N had to come from some source unaccounted for in table 11-1, with the most likely source being the soil organic N pool. The lower two N rates were thus effectively mining N from the SOM, which would result in a measurable decrease in SOM and a degradation of the soil resource over the long term. Only for the 3× rate do we see a residual N balance >0, indicating that more N was being applied than was being removed. Thus, only for the 3× treatment was SOM not being consumed, but rather sufficient N was being applied to potentially increase SOM. The existence of a positive N balance was also presumed responsible for a SOM increase, even with moldboard plowing, after 15 years of continuous corn fertilized annually with approximately 200 kg N ha^{-1} on an Iowa Till Plain site (Karlen et al., 1998). SOM increases in that study accounted for approximately 42% of the N budget for the period. However, it is important to remember that table 11-1 shows only a partial N balance, and we are not considering additional N loss pathways such as denitrification or volatilization. Including these loss pathways would result in a smaller N balance than is shown.

In summary, whereas the economic optimum N fertilizer rate was approximately 134 kg N ha^{-1} for the four-year study, nitrate concentrations in the tile drainage water for all treatments exceeded the 10 mg L^{-1} MCL for drinking water, and the lower N treatments (67 or 134 kg N ha^{-1}) were mining N from the SOM fraction. Thus, simply applying lower N fertilizer rates fails the definition of sustainability by not maintaining the long-term productivity of the soil, whereas the high N fertilizer rate does not meet the definition because of high NO_3 concentrations leaving the field in tile drainage. Based on this assessment, the practice of deficit fertilization, although suggested as a viable alternative for solving NO_3 contamination of surface waters and the northern gulf (Mitsch et al., 1999), is not a sustainable management practice with regard to long-term soil productivity.

Cover Crops and Bioreactors

A second example evaluating the sustainability of alternative N management strategies comes from the unpublished data collected for a study reported by Jaynes et al. (2004) using cover crops and an in-field bioreactor (see chapter 2). In their study, a corn-soybean rotation with conventional management and subsurface drainage was compared to the same rotation with an annual rye cover crop planted in the fall following each crop. In addition, a bioreactor consisting of wood chips buried in trenches on both sides of the subsurface drainage pipe was also investigated. The wood chips in the bioreactor served as a carbon source for denitrifying bacteria that reduced the nitrate in the shallow groundwater to N_2 before the nitrate could enter the subsurface drain and be carried from the field. Nitrogen fertilization for all treatments consisted of 224 kg ha^{-1} of N applied as UAN after corn emergence, which is on the upper end of the optimum N rate, but was used to stress the system with excess NO_3. Yields and grain protein were measured each year, with corn averaging 11.8 Mg ha^{-1} and soybean averaging 2.77 Mg ha^{-1}. Tile drainage volume and nitrate concentration were monitored continuously. For the years 2000-2004, the average flow-weighted NO_3 concentration for the conventional treatment was 22.4 mg N L^{-1}, well above the MCL for drinking water. The flow-weighted average NO_3 concentration for the cover crop treatment was 14.4 mg N L^{-1}, although the average was below 10 mg L^{-1} in the three years where a cover crop was well established. The flow-weighted average NO_3 concentration for the bioreactor treatment was 8.5 mg N L^{-1}. Thus, the conventional treatment was not sustainable from a water quality perspective, nor was the cover crop treatment in every year, although it greatly reduced NO_3 losses in years where good cover crops could be established. The bioreactor treatment was sustainable from a water quality perspective, as the NO_3 concentration in drainage was less than the MCL, but the longevity and profitability of this treatment remains to be determined.

A partial N balance for the conventional, cover crop, and bioreactor treatments is shown in table 11-2. Again, runoff was minimal from these 0.4 ha plots and assumed to contribute little to N losses. As in the previous example, most of the N inputs were from inorganic fertilizer, although estimated fixation was a significant N source for soybean. Outputs were dominated by grain removal, with tile drainage loss representing about a quarter of the N inputs. The overall N balance for the conventional system was slightly >0, indicating that the inputs and outputs were roughly in balance and SOM was not being mined from the soil. For the cover crop system, the N balance was substantially >0, indicating a potential build-up of soil N in the form of SOM, most likely due to uptake of N and the increased biomass input from the rye cover crop. For the bioreactor, the partial N balance was >>0, indicating a large N surplus. However, this surplus most likely did not represent a net gain of N within the soil but rather represented the increased denitrification that the bioreactor was designed to foster. Efforts to confirm the projected changes in SOM through direct measurements of SOM or more sensitive soil carbon fractionation are not planned for these plots for a few years because of the expected difficulty in detecting small changes in the large SOM pool.

The conventional production system in this case was sustainable from a soil productivity perspective, as the N mass balance indicates that the SOM content of the soil

Table 11-2. Partial N mass balance from five-year rate study by Jaynes et al. (2004).

Treatment	N Inputs (kg ha^{-1})			N Outputs (kg ha^{-1})				N Balance Residual
	Total N Fertilizer Applied	Total Wet and Dry Deposition	Total Fixed	Total Grain Removal	Total Drainage Loss	Total Runoff[a]	Change in Residual Mineral N	
Conventional	673	51	281	697	253	0	46	9
Cover crop	673	51	265	673	153	0	105	59
Bioreactor	673	51	258	676	100	0	88	118

[a] Not measured, but little runoff observed during the five-year period.

would be stable over the long term. However, this management system cannot be viewed as sustainable because of the high nitrate concentrations that leave the field in tile drainage. By adding a cover crop to the system, nitrate losses in drainage decreased substantially, but average NO_3 concentrations were still greater than the MCL. The cover crop also added biomass to the system that may combine with the nitrate that is not leached to form additional SOM, thus maintaining or enhancing long-term soil productivity. Based on these results, a corn-soybean rotation with a cover crop would be considered a sustainable system from both a water quality and soil quality perspective. Adding a bioreactor to the system dramatically reduces nitrate leaching and makes the system sustainable from a water quality perspective, but probably does little to enhance the soil resource.

Liquid Manure

On diversified farms, animal manure was traditionally applied to provide essential plant nutrients and to build SOM by returning crop residues that had been used for bedding. However, as small and medium-sized farms were replaced by concentrated animal feeding operations (CAFOs) and separate crop production enterprises, animal manure came to be considered more of a waste than a resource. Developing systems that reverse this perception and show that manure can be utilized in environmentally sound and economically profitable ways has been a research focus since the early 1990s (Hatfield and Stewart, 1998). This transition was not without many challenges associated with all aspects of the animal, crop, manure, and soil management systems (Karlen et al., 2004). Changes in manure management resulting in less solid (bedding) material, variability in nutrient composition as storage facilities were emptied, limited time for application, and the difficulty of regulating application rates are just a few examples.

With regard to sustainability of the soil resources and the potential impact on water quality, a six-year study conducted on Iowa Till Plain soils near Nashua, Iowa, using liquid swine manure as the N source provided the following insights. Tile drainage volume was highly variable among the 0.4 ha (1 acre) plots, presumably because of subtle differences in slope and inherent soil characteristics. This variation in drainage volume in addition to variation in seasonal precipitation, current year and prior manure application rates (caused by variation in nutrient composition and application challenges), and the crop (corn or soybean) being grown resulted in NO_3 losses that varied from 4 to 48 kg N ha^{-1} $year^{-1}$ during the six-year study (Karlen et al., 2004). When averaged for continuous corn, drainage loss accounted for 16% of the applied N,

Table 11-3. Partial N mass balance for a six-year swine manure study by Karlen et al. (2004).[a]

	N Inputs (kg ha^{-1})			N Outputs (kg ha^{-1})				
Treatment	Total N[b] Applied	Total Wet and Dry Deposition	Total Fixed	Total Grain Removal	Total Drainage Loss	Total Runoff[c]	Change in Residual Mineral N	N Balance Residual
Continuous corn	958	63	0	510	156	0	82	273
Corn phase[d]	794	63	0	600	84	0	116	57
Soybean phase[d]	0	63	1058	1092	150	0	2	-123

[a] Estimated six-year values based on 1996, 1997, and 1998 measurements (Bakhsh et al., 2001).
[b] A 2% loss from volatilization was assumed for liquid injection.
[c] Not measured, but very little runoff was observed during the six-year study.
[d] Both phases of a corn-soybean rotation were present each year.

whereas for the corn-soybean rotation it accounted for only 10%. Grain yield was also variable, averaging 6.4 Mg ha^{-1} for continuous corn (range 2.8 to 8.4 Mg ha^{-1}), and 7.9 Mg ha^{-1} for corn (range 5.5 to 9.8 Mg ha^{-1}) and 3.4 Mg ha^{-1} for soybean (range 2.6 to 3.9 Mg ha^{-1}) in the two-year corn-soybean rotation. The measured amount of N removed with the grain crops averaged 85, 100, and 182 kg ha^{-1} for the continuous corn and the corn and soybean phases of the rotation, respectively. Summing inputs and outputs for this manure study shows a substantial residual N balance for continuous corn (table 11-3), but the combined corn-soybean rotation residual was negative. Thus, continuous corn may be building SOM in this field, whereas the corn-soybean rotation was probably mining N from the SOM. However, measurements of SOM in the surface 20 cm (Karlen et al., 2004) did not reveal any significant changes over the six years of the study. Perhaps measuring one of the more active C/N pools, such as particulate organic matter (Cambardella and Elliot, 1993), would be more sensitive to changes in SOM content.

Limitations

As illustrated by our inability to detect SOM differences in soil samples from the three studies described above, assessing the impact of nutrient practices on the SOM content and long-term productivity of soil is difficult because we are trying to measure small changes in a large quantity. For example, Russell et al. (2005) measured lower particulate organic carbon content in a long-term corn-soybean rotation compared to continuous corn and a corn-corn-oat-alfalfa rotation, but could only detect an increase in SOM with increasing fertilizer rates for a continuous corn rotation. In contrast, Omay et al. (1997) were able to measure a 32% increase in potentially mineralizable C due to application of N fertilization, but could not consistently measure differences due to crop rotation. Much of the inability to quantify differences is due to the large variability in measured organic fractions in soil.

Therefore, we have substituted an N mass balance for a C mass balance because N inputs and losses are more easily measured. Nevertheless, there are large uncertainties in N mass balance computations, even at the field scale. On the input side, the contribution of soybean to the available N pool through fixation is highly uncertain and variable (Russelle and Birr, 2004). Although more N is removed in grain harvest of soybean than is fixed (Heichel, 1987), soybean still fixes considerable amounts of N. Better quantification of this fixed N would greatly reduce the uncertainty of the input calculations.

On the output side, volatilization of N from fertilizer, manure, and soil are important but also difficult to quantify as they are weather and practice specific. Estimates of the N emissions from senescing plants vary over an order of magnitude but have been rarely measured. New methods need to be developed for measuring N losses at the field-scale from senescing plants. Denitrification is highly variable over time and space (Parkin and Meisinger, 1989), making annual field-wide estimates suspect.

Finally, direct measurement of changes in SOM that could detect changes in a few years would greatly ease the assessment of the sustainability of potential practices. New techniques need to be developed to allow measurement of changes in the large SOM pool in most soils from the Corn and Soybean Belt.

Summary

Effects of nutrient management practices need to be evaluated against not only economics and water quality, but also against long-term soil productivity to ensure a profitable and environmentally sustainable agricultural production system within the Corn and Soybean Belt. Soil organic matter is an important indicator of soil productivity and soil tilth, as many of the biological, chemical, and physical properties of a soil that are important for crop production in the Corn and Soybean Belt are strongly influenced by SOM levels. For the studies reviewed here, the negative N mass balances for the 1× treatment in the first example and the corn-soybean rotation under liquid manure application in the third example are equivalent to ~0.2% year^{-1} loss in SOM from the top 20 cm of the profile. Conversely, the positive N balances for the cover crop in table 11-2 and the continuous corn with liquid manure application in table 11-3 represent increases in SOM of ~0.2% and 0.9% year^{-1}, respectively. While small in terms of our current ability to directly measure, these changes represent about a 5% loss in SOM over 30 years for the first two, and a gain of 5% and 26% in 30 years for the latter two. Compared to the 10% to 20% minimum differences in SOM required to detect significant differences typically found in soil carbon studies (Russell et al., 2005), it is little wonder that losses in SOM due to organic matter decomposition have been difficult to quantify with direct SOM measurements

Nutrient management practices need to be assessed for their ability to enhance or maintain SOM content in addition to their impact on yield and profit. Just as nutrient management studies are incomplete if they consider only yield and ignore water quality, water quality studies evaluating nutrient practices that neglect the long-term effects on the soil resource are also incomplete. Future nutrient management studies must be designed to measure impacts on soil and water resources as well the economics of various practices.

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References

Bakhsh, A., R. S. Kanwar, D. L. Karlen, C. A. Cambardella, T. B. Bailey, T. B. Moorman, and T. S. Colvin. 2001. N-management and crop rotation effects on yield and residual soil nitrate levels. *Soil Sci*. 166(8): 530-538.

Barry, D. A. J., D. Goorahoo, and M. J. Goss. 1993. Estimation of nitrate concentrations in groundwater using a whole-farm nitrogen budget. *J. Environ. Qual*. 22: 767-775.

Burkart, M. R., and D. E. James. 1999. Agricultural-nitrogen contributions to hypoxia in the Gulf of Mexico. *J. Environ. Qual*. 28: 850-859.

Cambardella, C. A., and E. T. Elliott. 1993. Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. *SSSA J*. 57: 1071-1076.

David, M. B., L. E. Gentry, D. A. Kovacic, and K. M. Smith. 1997. Nitrogen balance in and export from an agricultural watershed. *J. Environ. Qual*. 26: 1038-1048.

Drinkwater, L. E., P. Wagoner, and M. Sarrantonio. 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. *Nature* 396(19): 262-265.

Francis, D. D., J. S. Schepers, and A. L. Sims. 1997. Ammonia exchange from corn foliage during reproductive growth. *Agron. J.* 89: 941-946.

Gascho, G. J., R. D. Wauchope, J. G. Davis C. C. Truman, C. C. Dowler, J. E. Hook, H. R. Sumner, and A. W. Johnson. 1998. Nitrate-nitrogen, soluble, and bioavailable phosphorus runoff from simulated rainfall after fertilizer application. *SSSA J.* 62: 1711-1718.

Goolsby, D. A., W. A. Battaglin, G. B. Lawrence, R. S. Artz, B. T. Aulenbach, R. P. 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. Decision Analysis Series No. 17. Silver Springs, Md.: NOAA Coastal Ocean Program.

Hatfield, J. L., and B. A. Stewart, eds. 1998. *Animal Waste Utilization: Effective Use of Manure as a Soil Resource*. Chelsea, Mich.: Sleeping Bear Press.

Heichel, G. H. 1987. Legume nitrogen: Symbiotic fixation and recovery by subsequent crops. In *Energy in Plant Nutrition and Pest Control*, 63-80. Z. R. Helsel, ed. Amsterdam: Elsevier.

Jaynes, D. B., T. S. Colvin, D. L. Karlen, C. A. Cambardella, and D. W. Meek. 2001. Nitrate losses in subsurface drainage as affected by nitrogen fertilizer rate. *J. Environ. Qual*. 30: 1305-1314.

Jaynes, D. B., T. C. Kaspar, T. B. Moorman, and T. B. Parkin. 2004. Potential methods for reducing nitrate losses in artificially drained fields. In *Proc. International Drainage Symp*., 59-67. St. Joseph, Mich.: ASAE.

Karlen, D. L., A. Kumar, R. S. Kanwar, C. A. Cambardella, and T. S. Colvin. 1998. Tillage system effects on 15-year carbon-based and simulated N budgets in a tile-drained Iowa field. *Soil and Till. Res*. 48: 155-165.

Karlen, D. L., C. A. Cambardella, and R. S. Kanwar. 2004. Challenges of managing liquid swine manure. *Applied Eng. in Agric*. 20(5): 693-699.

Keeney, D. 1990. Sustainable agriculture: Definition and concepts. *J. Prod. Agric*. 3: 281-285.

Killorn, R., and J. Lorimor. 2003. Managing manure nutrients for crop production. PM 1811. Ames, Iowa: Iowa State University Extension.

Libra, R. D., C. F. Wolter, and R. J. Langel. 2004. Nitrogen and phosphorus budgets for Iowa and Iowa watersheds. Tech. Info. Ser. No. 47. Des Moines, Iowa: Iowa Department of Natural Resources, Iowa Geological Survey.

McIsaac, G. R., and X Hu. 2004. Net N input and riverine N expert from Illinois agricultural watersheds with and without extensive tile drainage. *Biogeochem*. 70: 251-271.

McIsaac, G. R., M. B. David, G. Z. Gertner, and D. A. Goolsby. 2002. Relating net nitrogen input in the Mississippi river basin to nitrate flux in the lower Mississippi river: A comparison of approaches. *J. Environ. Qual*. 31: 1610-1622.

Mitsch, W. J., J. W. Day Jr., J. W. Gilliam, P. M. Groffman, D. L. Hey, G. W. Randall, and N. Wang. 1999. Reducing nutrient loads, especially nitrate-nitrogen to surface water, groundwater, and the Gulf of Mexico. Topic 5. Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico. Decision Analysis Series No. 19. Silver Springs, Md.: NOAA Coastal Ocean Program.

Omay, A. B., C. W. Rice, L. D. Maddux, and W. B. Gordon. 1997. Changes in soil microbial and chemical properties under long-term crop rotation and fertilization. *SSSA J*. 61: 1672-1678.

Parkin, T. B., and J. Meisinger. 1989. Denitrification below the crop rooting zone as influenced by surface tillage. *J. Environ. Qual*. 18: 12-16.

Russell, A. E., D. A. Laird, T. B. Parkin, and A. P. Mallarino. 2005. Impact of nitrogen fertilization and cropping system on carbon sequestration in Midwestern mollisols. *SSSA J*. 69: 413-422.

Russelle, M. P., and A. S. Birr. 2004. Large-scale assessment of symbiotic dinitrogen fixation by crops: Soybean and alfalfa in the Mississippi River basin. *Agron. J*. 96: 1754-1760.

Waksman, S.A. 1938. *Humus*. New York, N.Y.: Williams and Wilkins.

Webb, J., S. Ellis, R. Harrison, and R. Thorman. 2004. Measurement of N fluxes and soil N in two arable soils in the U.K. *Plant and Soil* 260: 253-270.

Pp. 171-187 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008.
Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop. St. Joseph, Michigan: ASABE.

Field-Scale Tools for Reducing Nutrient Losses to Water Resources

12

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Abstract. *Phosphorus indices are field-scale assessment tools that have potential for identifying areas most likely to contribute P to water resources and for focusing management practices to control these losses. They function by evaluating factors known to affect the extent of P in runoff and using these results as the basis for fertilizer and manure management planning. Phosphorus indices are more comprehensive than soil test P alone for identifying field P runoff risks because they consider both the amount of P available to runoff (source factors) and the potential for runoff and erosion to occur (transport factors). They are currently developed at the state level, with some states following a multiplicative matrix approach that leads to categorical risk rankings and other states using a semi-quantitative modeling approach that estimates a field's annual runoff unit area P loads. Limited validation data show good relationships between measured site-specific, field-scale P runoff losses and P index values. The semi-quantitative modeling approach may provide some advantages, since it allows quantitative consideration of site-specific factors affecting P runoff losses. Substantial local research data bases on the effects of site and management factors on the risk of P losses in runoff are needed to construct reliable P indices.*

Phosphorus (P) loss in runoff from cropland is a water quality concern because this P often promotes algae and other vegetative growth in freshwater lakes and streams (Carpenter, et al., 1998; Correll, 1998). When this vegetation decomposes, dissolved oxygen levels in the natural waters are depleted. This can cause death or damage to fish and other aquatic organisms as well as odors and a general degradation of the aesthetic and recreational value of the environment. Some evidence also exists that certain blue green algae in eutrophic waters can produce toxins that contribute to taste and odor problems and may pose a health hazard to livestock and humans if these waters are used for drinking purposes (Kotak et al., 1993; Sharpley et al, 1994). In addition, algal blooms can contribute to trihalomethane formation during chlorination of drinking water supplies.

Phosphorus entry into natural waters from point sources such as industrial discharges and municipal sewage treatment facilities is currently regulated under the provisions of water quality protection legislation. Nonpoint or diffuse P entry into natural waters, such as that occurring in runoff from managed and natural landscapes, is more difficult to quantify and manage. Since cropland receives frequent P additions from fertilizers and manures, and sediment-bound P losses can occur through soil erosion, runoff from agricultural fields can contribute substantial amounts of P to surface water.

Initially, P was not identified as a key nutrient influencing the extent of Gulf of Mexico hypoxia (Rabalais et al., 2001). More recently, an EPA (Region 4) paper suggested that controlling both nitrogen (N) and P loading into the Gulf of Mexico could be beneficial in minimizing hypoxia. However, the emphasis in this chapter will be on addressing local surface water quality concerns.

Specifically, this chapter will focus on the use of field-scale tools to manage and reduce P losses from cropland. Since the development of P indices has occurred in essentially every state in the U.S., these products are among the most promising approaches to predicting the risk of P losses from agricultural fields and developing appropriate management practices to control or reduce these losses (Maguire et al., 2005; Sharpley et al., 2003). The P indices developed are intended primarily to assess risk of P loss from fields and, therefore, are intended for use as planning tools for P-based nutrient management and conservation practices. The high level of activity in development of P indices in the U.S. is largely in response to USDA and/or EPA proposals that all animal feeding operations (AFOs) have a nutrient management plan (addressing both N and P) in place by 2008 to address water quality concerns related to nutrient management (Heathwaite et al., 2005).

Field-Scale Tools for Assessing P Losses

National policy and general guidelines on nutrient management issued by the USDA-NRCS (1999) recognized the need for enhanced P-based nutrient management in agriculture to control nonpoint-source losses of P. Three risk assessment tools were proposed in the USDA-NRCS national policy: agronomic soil test P interpretation categories, soil test P threshold values resulting in a critical runoff P concentration, or a comprehensive P loss risk assessment tool (P index). The soil test P category option is appealing because soil test information is widely available for many agricultural fields, and this parameter can be readily obtained at low cost. However, soil test P is not a reliable predictor of P loss risk because it does not consider the transport component required for P losses in runoff and subsurface drainage.

Use of optimal soil test P levels for crop production as an upper limit to minimize risk of P loss from fields would be reasonable only when both animal production economics and the transport component contributing to P loss are ignored. Likewise, the soil test P threshold value option considers primarily the level of P source and not the many variables involved in transporting P from the field. In addition, this method would necessitate a massive data collection effort to determine the soil test P value associated with a critical runoff P concentration. Because soils may differ in runoff P concentrations at a given soil test P value (Pote et al., 1999; Cox and Hendricks, 2000; Andraski and Bundy, 2003) these relationships would need to be determined on many agriculturally important soils in each state. In addition, there is no consensus on what critical runoff P concentration should be used as the threshold value. A concentration of 1 ppm P, which is the typical threshold value used for point sources, has been suggested (Sharpley et al., 1996). It seems likely that the critical P concentrations would need to be determined for individual receiving waters depending on the sensitivity of water quality to P additions in each case. This concept is similar to the USEPA ecoregional approach described by Gibson et al. (2000).

Of the alternative strategies proposed in the USDA-NRCS national policy, the P index risk assessment tool is most likely to provide realistic estimates of P loss risks because it can consider both source and transport components involved in P runoff losses. Most P indices in use or under development consider various source and transport factors affecting the risk of P loss (Mallarino et al., 2002). These factors typically include soil erosion potential, site characteristics affecting runoff, soil test P, fertilizer or manure P application rates and methods, and field proximity to a stream channel or water course.

Structure of P Indices

Initially, Lemunyon and Gilbert (1993) proposed a P index structure that involved assigning a numerical value to each major source or transport factor likely to influence P loss. In addition, a weighting coefficient reflecting the relative importance of each factor in influencing P loss was assigned. A P index value was calculated by multiplying the factor P loss rating by its weighting coefficient and summing these products across the source and transport factors considered. Index values for individual fields were categorized using a general P loss risk ranking (low to very high), and nutrient management recommendations appropriate for the level of P loss risk were made. Lemunyon and Gilbert (1993) recommended that individual states modify and adapt the original P index structure to reflect local conditions and practices.

In P indices based on this initial design, the influence of factors affecting P losses were additive, which often did not accurately reflect the interaction of P source and transport contributions to P losses. Subsequent P indices continued with the matrix structure proposed by Lemunyon and Gilbert (1993), but included additional factors affecting P loss potential, grouped P loss factors into separate P transport and P source categories, and employed a multiplicative approach to calculating the P index value, as proposed by Gburek and Sharpley (1998) and Gburek et al. (2000). Multiplying the P source loss potential value by the corresponding P transport value allowed the P loss risk index value to indicate the strong interdependence of source and transport factors. For example, low P index values were produced when either source or transport factors were low, even when the corresponding source or transport loss potential factor was very high (Gburek et al., 2000).

The P indices currently in use in Delaware (Leytem et al., 2003), Pennsylvania (Weld et al., 2002), and Maryland (Coale et al., 2002) are examples of the multiplicative matrix or row and column P index structure described above. These indices provide a numerical or categorical rating of P loss potential on a field scale, but do not attempt to provide a quantitative estimate of annual P loss in runoff. The P index used in Pennsylvania illustrates the P source (table 12-1) and transport (table 12-2) factors typically included in P indices, along with the weighting factors assigned to various components. A complete list of P index approaches and modifications can be found in Sharpley et al. (2003).

Table 12-1. The Pennsylvania P index: Source factors (Weld et al., 2003).

Contributing Factor	Risk Level				
	Very Low	Low	Medium	High	Very High
Soil test P risk	Risk value = Mehlich-3 soil test P (mg kg^{-1} P) × 0.20				
Loss rating for P application method and timing	Placed with planter or injected >2 in. deep	Incorporated <1 week after application	Incorporated >1 week or not incorporated following application in spring-summer	Incorporated >1 week or not incorporated following application in autumn-winter	Surface applied on frozen or snow-covered soil
	0.2	**0.4**	**0.6**	**0.8**	**1.0**
Fertilizer P risk	Risk value = Fertilizer P application rate (lbs P_2O_5 $acre^{-1}$) × Loss rating for P application				
Manure P availability	Based on organic P source availability coefficients[a]				
Manure P risk	Risk value = Manure P application rate (lbs P_2O_5 $acre^{-1}$) × Loss rating for P application × P availability coefficient				
Source factor = Soil test P risk + Fertilizer P risk + Manure P risk					

[a] The appropriate phosphorus availability coefficient to use in developing a nutrient management plan is determined based on the organic P source: 1.0 = swine slurry; 0.9 = layer, turkey, duck, liquid dairy; 0.8 = broiler, bedded pack dairy, beef, biological nutrient removal biosolids; 0.5 = alum-treated manure; 0.4 = alkaline-stabilized biosolids; 0.3 = conventionally stabilized and composted biosolids; and 0.2 = heat-dried and advanced-alkaline stabilized biosolids.

Table 12-2. The Pennsylvania P index: Transport factors (Weld et al., 2003).

Characteristic	Risk Level				
Soil erosion	Risk value = annual soil loss = _____ tons $acre^{-1}$ $year^{-1}$				
Runoff potential	Very Low **0**	Low **1**	Medium **2**	High **4**	Very High **8**
Subsurface drainage	None **0**		Random **1**		Patterned[a] **2**
Contributing distance	>500 ft **0**	500 to 350 ft **1**	350 to 250 ft **2**	250 to 150 ft **4**	<150 ft **8**
Transport sum = Erosion + Runoff potential + Subsurface drainage + Contributing distance					
Modified connectivity	Riparian buffer (applies to distances <150 ft)		Grassed waterway or none	Direct connection (applies to distances >150 ft)	
	0.7		**1.0**	**1.1**	
Transport factor = Transport sum × Modified connectivity / 22[b]					
P index = 2 × Source sum × Transport sum					

[a] Or a rapidly permeable soil near a stream.
[b] Transport value is divided by 22 (i.e., the highest value obtainable) in order to normalize transport to a value of 1, where full transport potential is realized.

Table 12-3. General structure of P indices in Iowa, Minnesota, and Wisconsin.

State	P Index Formula[a]
Iowa	PI = PP + DP + subsurface P
Minnesota	PI = PP + rainfall DP + snowmelt DP
Wisconsin	PI = (PP + DP + event losses) × TP delivery ratio

[a] PI = P index value, P = particulate P, DP = dissolved P, and TP = total P

Several states in the North Central region of the U.S. have developed P indices using semi-quantitative modeling approaches that attempt to estimate annual P losses on a field-by-field basis. In the eastern U.S., North Carolina has developed a P index using a generally similar modeling approach (N.C. PLAT Committee, 2005). These indices are sensitive to the need to utilize input data that are available or easily obtainable by users, and they are much less data-intensive than more complex process-based research P loss models. The P indices developed in Iowa (Mallarino et al., 2002; available at: www.ia.nrcs.usda.gov/technical/Phosphorus/phosphorusstandard.html), Minnesota (Minnesota Phosphorus Site Risk Index, 2005), Missouri (available at: www.nmplanner.missouri.edu), and Wisconsin (available at: wpindex.soils.wisc.edu/; see also: www.snapplus.net) using a semi-quantitative modeling approach were independently constructed based on available data within each state. Informal interaction and information exchange among the four states allowed comparisons of techniques for estimating P index parameters and probably promoted commonality among the individual indices. While some distinct differences remain among the P indices in the three states where the index is at the most advanced stages of development and implementation, there are many similarities in the approaches used to estimate P loss potential on a field-by-field basis. These similarities are apparent in the general formulae used to calculate P index values in the three states (table 12-3).

In all cases, the P indices seek to estimate the amount of annual P load (lb P/acre/year) lost on a field-by-field basis. The Iowa P index suggests distinct P index calculations for different “conservation management units” within a field. This approach is useful for identifying areas within fields that may be sources of high P loss and for targeting soil conservation and/or crop management practices to these areas to minimize losses. All three indices estimate particulate P (PP) and dissolved P (DP) separately and sum these values. The separate estimates of PP and DP are useful indicators of the mechanism of P losses in a given field and the management options that may be effective in lowering the P loss. For example, if PP is the major contributor to P loss, then modification of cropping systems, tillage, or other conservation practices to control sediment loss would likely reduce overall P loss. Alternatively, a high DP contribution to the P index total suggests losses from surface applications of P sources, high soil test P levels, or winter runoff.

While the general approach for calculating annual P loads in runoff is similar among states, the specific algorithms for calculating individual components needed to estimate P loss are often different. Some of the similarities and differences in the Iowa, Minnesota, and Wisconsin P indices are summarized in table 12-4. All three states use RUSLE2 to estimate sediment delivery. Iowa and Minnesota calculate a field-to-stream sediment delivery ratio using the distance from field to stream. Wisconsin takes into account both sediment-bound and dissolved P transport from field to stream in its total P delivery factor, which is based on distance and slope of the drainage path. The influence of vegetative buffers is accounted for by somewhat different approaches in Iowa, Minnesota, and Wisconsin. In the Wisconsin index, RUSLE2 is used to make a field-specific assessment of particulate P delivery through a buffer. Particulate P loss estimates are adjusted for recent P applications (since the last soil test P measurement) in Minnesota and Wisconsin, but not in Iowa.

Similar approaches are also employed in the three states for estimating the dissolved or soluble P component of P loss, with runoff volume estimates being based on runoff curve numbers and precipitation data. Soil test P values from several recommended tests for crop production are uniformly employed to calculate dissolved P concentrations in runoff, and adjustments for recent P additions are accomplished using soil P buffer capacity information. Dissolved P loss in runoff from recent surface P applications from rainfall and snowmelt events are accounted for through use of time and method of application factors in the Iowa P index. Minnesota and Wisconsin use somewhat different processes to estimate soluble P from winter runoff. However, all states use information on the amount of P applied, expected percentage of applied P lost in runoff, tillage, and application time in their estimates.

In the Iowa P index, a separate internal drainage component considers the impacts of subsurface tile drainage systems, water flow volume to tile lines, surface water recharge from subsurface flow, and the soil P level on the amount of total dissolved P delivered to surface water through flow to tile lines or surface water recharge from subsurface flow. It uses existing databases for soils and landscape forms, an estimate of water flow as a proportion of historic county precipitation data, and a two-class soil P factor based on soil test P and empirical data.

Table 12-4. Comparison of components used in the Iowa, Minnesota, and Wisconsin P indices.

P Index Component	Iowa	Minnesota	Wisconsin
Particulate P			
Sediment delivery	RUSLE2	RUSLE2	RUSLE2
Sediment delivery ratio	Distance to stream	Distance from field to stream	Distance and slope from field to stream on TP
Buffer factor	Buffer width	Sediment trap factor	Under development
Sediment P content	Calculated from soil test P	Calculated from soil test P and organic matter	Calculated from soil test P and organic matter
Adjust. of PP for recent P additions	None	Optional based on soil P buffer capacity	Soil test P adjusted based on buffer capacity
PP enrichment factor	1.1 to 1.3 depending on management practices	None	Under development
Dissolved/soluble P			
Runoff volume	From runoff curve numbers and % of precipitation	From runoff curve numbers and % of precipitation	Avg. precipitation, runoff curve numbers, plus winter runoff
Dissolved P in runoff	From soil test P	From soil test P	Soil soluble P from soil test P times extraction efficiency
Adjust. of DP for recent P additions	From buffer capacity and method and time factor	Optional based on soil P buffer capacity	Soil test P adjusted based on buffer capacity
Dissolved P from surface P applications	Included in adjustment of DP for recent P additions (above)	From amount of P applied, timing, and tillage	Included in soil test P adjustment (above) plus single event P loss
P in snowmelt and from winter applied manure	Included in adjust. of DP for recent P additions (above)	From snowmelt runoff volume, tillage, % of applied P, and residue P	Est. worst-case loss from: manure soluble P, % P loss, slope, and tillage

Validation of P Indices

Validation of P indices as tools for predicting the risk of P runoff from agricultural landscapes requires measurement of actual annual P runoff losses from field-scale areas where P index values for the same fields can be obtained. Currently, little information is available confirming the relationship between P index values and measured annual P runoff losses from individual fields.

Several reports have compiled information on the relative proportion of agricultural fields in a designated region that would be assigned to various interpretive categories for the P index being evaluated (Coale et al., 2002; Leytem et al., 2003). While these studies provide valuable information on the magnitude of management changes needed to bring most fields into an acceptable interpretive category, no information on the relationship between P index values and actual P losses is obtained. Usually the P index interpretive categories used are not directly tied to environmental criteria for P loss, and the need for field validation is recognized by the authors (Coale et al., 2002; Leytem et al., 2003).

Veith et al. (2005) recently compared measured P runoff losses from a south-central Pennsylvania watershed with losses from this watershed predicted by the Soil and Water Assessment Tool (SWAT). The SWAT model is a complex watershed-level research-based simulation model (Arnold et al., 1998). Direct measurements of runoff P were conducted during a 7-month period (April through October) during four years (1997-2000); thus, the runoff P measurements did not include winter runoff contributions. In addition, field-level P loss predictions from SWAT for 22 fields within the monitored watershed were compared with values from the Pennsylvania P index for the same fields. Results showed that watershed P loss measurements for dissolved and total P were of the same magnitude as SWAT P loss predictions. The P index and SWAT categorized P loss risk similarly for 73% of the 22 fields evaluated, and P loss assessments by the two methods were well correlated. The authors concluded that the P index can be reliably used to assess where P losses occur in a watershed and where management practices are needed to control losses and ultimately provide for improved water quality.

In Wisconsin, (Good et al., 2005, unpublished) annual (12-month) measurements of P runoff losses were obtained from 21 crop years at a field or subwatershed scale, and these measurements were compared with the Wisconsin P index values for the same areas. The 21 sites represented 18 fields on seven farms in four major topographic areas of the state. Soil textures included silty clay loam, silt loam, and loam; slopes ranged from 4% to 13%; crops included alfalfa, alfalfa/brome, corn grain, and corn silage; and manure was applied in the monitoring year in 11 (4 incorporated, 7 surface) of the 21 sites. Eight of the runoff monitoring stations utilized passive interception devices with drainage areas of 0.04 to 2.5 acres. The remaining 13 sites were equipped with H-flumes and USGS automated gauging stations with drainage areas of 9 to 40 acres. Runoff volumes and analyses of runoff for sediment, total P, and dissolved P were compiled for each site.

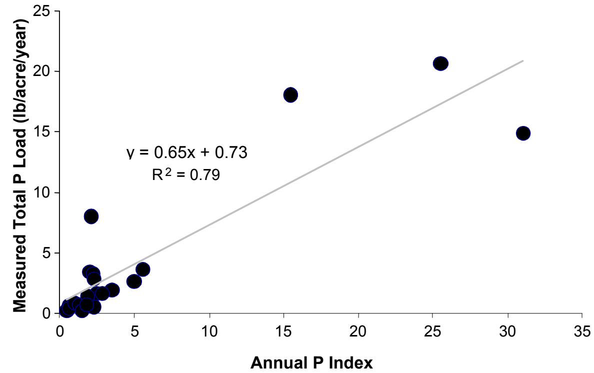

Figure 12-1. Relationship between measured annual runoff P loads and Wisconsin P index values for 21 field locations in Wisconsin.

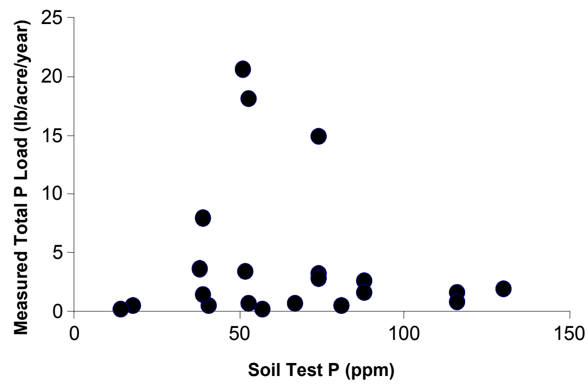

Figure 12-2. Relationship between measured annual runoff P loads and Bray P-1 soil test values for 21 field locations in Wisconsin.

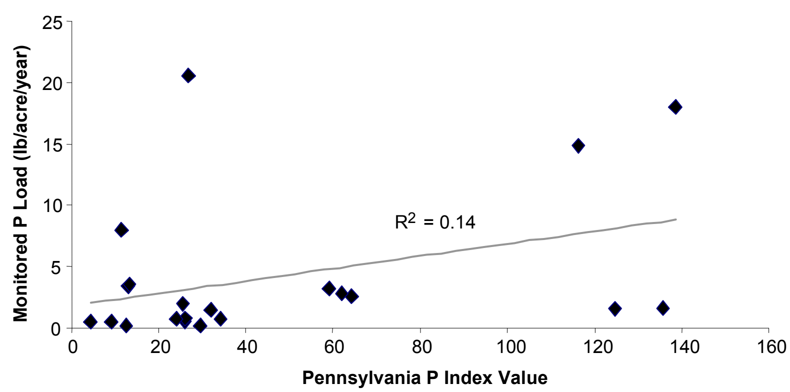

Figure 12-3. Relationship between measured annual runoff P loads and P index values calculated using the Pennsylvania P index for 21 field locations in Wisconsin.

The data in figure 12-1 show that measured annual edge-of-field P loads from the monitored areas were well correlated ($r^2 = 0.79$) with the Wisconsin P index edge-of-field values calculated for the same areas. This finding indicates that the Wisconsin P index is a reliable predictor of actual P runoff losses from cropland. As expected, no relationship was found between annual runoff P loads and field average soil test P values, since soil test P alone indicates only the level of P source and does not reflect the transport component involved in runoff P losses (fig. 12-2).

Little information is available to evaluate the performance of matrix or row and column P indices relative to indices using a semi-quantitative modeling approach. Figure 12-3 shows the relationship between Pennsylvania P index values and measured annual P runoff loads from the same 21 locations as used in figures 12-1 and 12-2. Comparison of Figures 12-1 and 12-3 indicates that the Wisconsin P index values are much more closely related to measured P losses than the P index values calculated with the Pennsylvania P index. Since the P indices used in Wisconsin and Pennsylvania were developed from local information available in each state, part of the difference in performance may be due to state-specific influences that are reflected in the P index calculations. Specifically, the Pennsylvania P index may not reflect measured P losses under Wisconsin conditions because this index was developed using information specific to factors affecting P losses in Pennsylvania. Alternatively, the site-specific, quantitative consideration of factors affecting P runoff losses that can be obtained with the modeling approach used in the Wisconsin P index may have better capability to predict runoff P losses.

Relationships between P index values and measured P losses have been determined in several other states. Harmel et al. (2005) found significant linear relationships between both the Texas and Iowa P index values and measured annual runoff P loads. Using simulated rainfall, Sharpley et al. (2001) found high correlations between Pennsylvania P index values and runoff P concentrations and loads. Arkansas P index values were highly correlated with P losses in natural rainfall runoff from pastures receiving poultry litter (DeLaune et al. 2004), and the Arkansas P index was further refined to reflect conditions in a specific watershed (DeLaune et al. 2006). Collectively, these findings emphasize the need for each state to develop unique P indices that consider local factors likely to influence P runoff losses.

Summary and Conclusions

Field-scale tools for assessing the risk of P losses have potential for identifying areas most likely to contribute P to water resources and for focusing management practices to control these losses. Phosphorus loss assessment tools function by evaluating factors known to affect the extent of P losses and using these results as the basis for nutrient management planning. Ideally, these tools will consider both source and transport components involved in P losses. The majority of the field-scale tools currently used for assessing the risk of P losses to water resources are either based on soil test P or are P indices. The extent of loss identified by these tools is expressed as a categorized risk level (e.g., low to high) or as a semi-quantitative estimate of annual P loads in runoff. Limited validation work indicates a good relationship between measured field-scale P losses and edge-of-field index values from P indices used in several states.

The field-scale assessment tools available are intended for use as planning tools to identify appropriate management practices that will lower P losses. As such, the quantitative reduction in P loss that could be achieved by application of these tools will vary on a field-by-field basis and will depend on the factors influencing these losses and the practices selected to reduce the losses. Field-scale P loss assessment tools are useful for identifying cropland that could benefit from improved management to control losses. Some P indices may also have potential for identifying high P loss areas within fields and for targeting practices to control these losses. Application of these tools should have limited impact on crop yields and may enhance long-term productivity by minimizing soil erosion. Effective application of these tools will require user training.

Evaluation of field-scale tools indicates that field-average soil test P levels have little value in predicting P loss because this parameter considers only P source components and does not consider P transport factors. A good relationship was found between annual field-scale measurements of P loss and P index values derived from a semi-quantitative model P index in Wisconsin. Less favorable relationships were found between these measured P runoff losses and P index values from the matrix-type P index used in Pennsylvania. As noted earlier, the weaker relationships with the Pennsylvania P index may be due to its application to conditions different from those found in Pennsylvania. Additional validation of field-scale tools against measured annual P losses is needed.

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References

Andraski, T. W., and L. G. Bundy. 2003. Relationship between phosphorus levels in soil and in runoff from corn production systems. *J. Environ. Qual*. 32: 310-316.

Arnold, J. G., R. Srinivasan, R. S. Muttiah, and J. R. Williams. 1998. Large-area hydrologic modeling and assessment: Part 1. Model development. *J. American Water Res. Assoc*. 34(1): 73-89.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8: 559-568.

Coale, F. J., J. T. Sims, and A. B. Leytem. 2002. Accelerated deployment of an agricultural nutrient management tool: The Maryland phosphorus site index. *J. Environ. Qual.* 31: 1471-1476.

Correll, D. L. 1998. The role of phosphorus in the eutrophication of receiving waters: A review. *J. Environ. Qual*. 27: 261-266.

Cox, F. R., and S. E. Hendricks. 2000. Soil test phosphorus and clay content effects on runoff water quality. *J. Environ. Qual.* 29: 1582-1586.

DeLaune, P. B., P. A. Moore, D. K. Carman, A. N. Sharpley, B. E. Haggard, and T. C. Daniel. 2004. Evaluation of the phosphorus source component in the phosphorus index for pastures. *J. Environ. Qual*. 33: 2192-2200.

DeLaune, P. B., B. E. Haggard, T. C. Daniel, I. Chaubey, and M. J. Cochran. 2006. The Eucha/Spavinaw phosphorus index: A court-mandated index for litter management. *J. Soil Water Conserv*. 61: 96-105.

Gibson, G. R., R. Carlson, J. Simpson, E. Smeltzer, J. Gerritson, S. Chapra, S. Heiskary, J. Jones, and R. Kennedy. 2000. Nutrient criteria technical guidance manual: Lakes and reservoirs (EPA-822-B00-001). Washington, D.C.: U.S. Environmental Protection Agency.

Gburek, W. J., and A. N. Sharpley. 1998. Hydrologic controls on phosphorus loss from upland agricultural watersheds. *J. Environ. Qual.* 27: 267-277.

Gburek, W. J., A. N. Sharpley, A. L. Heathwaite, and G. J. Folmar. 2000. Phosphorus management at the watershed scale: A modification of the phosphorus index. *J. Environ. Qual.* 29: 130-144.

Harmel, R. D., H. A. Torbent, P. B. DeLaune, B. E. Haggard, and R. L. Haney. 2005. Field evaluation of three phosphorus indices on new application sites in Texas. *J. Soil Water Conserv.* 60: 29-42.

Heathwaite, A. L., A. Sharpley, M. Bechmann, and S. Rekolainen. 2005. Assessing the risk and magnitude of agricultural nonpoint-source phosphorus pollution. In *Phosphorus: Agriculture and the Environment*, 981-1020. J. T. Sims and A. N. Sharpley, eds. Agronomy Monograph No. 46. Madison, Wisc.: ASA-CSSA-SSSA.

Kotak, B. G., S. L. Kenefick, D. L. Fritz, C. G. Rousseaux, E. E. Prepas, and S. E. Hrudey. 1993. Occurrence and toxicological evaluation of cyanobacterial toxins in Alberta lakes and farm dugouts. *Water Res*. 27: 495-506.

Lemunyon, J. L., and R. G. Gilbert. 1993. The concept and need for a phosphorus assessment tool. *J. Prod. Agric*. 6: 483-486.

Leytem, A. B., J. T. Sims, and F. J. Coale. 2003. On-farm evaluation of a phosphorus site index for Delaware. *J. Soil Water Conserv*. 58: 89-97.

Maguire, R. O., Q. M. Ketterings, J. L. Lemunyon, A. B. Leytem, G. Mullins, D. L. Osmond, and J. L. Weld. 2005. Phosphorus indices to predict risk for phosphorus losses. SERA-17 position paper. Available at: www.sera17.ext.vt.edu/SERA_17_Publications.htm.

Mallarino, A. P, B. M. Stewart, J. L. Baker, J. D. Downing, and J. E. Sawyer. 2002. Phosphorus indexing for cropland: Overview and basic concepts of the Iowa phosphorus index. *J. Soil Water Conserv*. 57: 440-447.

Minnesota Phosphorus Site Risk Index. 2005. Minnesota phosphorus site risk index, User's guide (May, 2005). St. Paul, Minn.,: University of Minnesota, Department of Soil, Water, and Climate.

N.C. PLAT Committee. 2005. North Carolina phosphorus loss assessment: I. Model description, and II. Scientific basis and supporting literature. Technical Bulletin 323. Raleigh, N.C.: North Carolina State University, North Carolina Agricultural Research Service.

Pote, D. H., T. C. Daniel, D. J. Nichols, A. N. Sharpley, P. A. Moore, Jr., D. M. Miller, and D. R. Edwards. 1999. Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff. *J. Environ. Qual*. 28: 170-175.

Rabalais, N. N., R. E. Turner, and W. J. Wiseman, Jr. 2001. Hypoxia in the Gulf of Mexico. *J. Environ. Qual.* 30: 320-329.

Sharpley, A. N., S. C. Chapra, R. Wedepohl, J. T. Sims, and T. C. Daniel. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. *J. Environ. Qual*. 23: 437-451.

Sharpley, A., T. C. Daniel, J. T. Sims, and D. H. Pote. 1996. Determining environmentally sound soil phosphorus levels. *J. Soil Water Conserv*. 51: 160-166.

Sharpley, A. N., R. W. McDowell, J. L. Weld, and P. J. A. Kleinman. 2001. Assessing site vulnerability to phosphorus loss in an agricultural watershed. *J. Environ. Qual*. 30: 2026-2036.

Sharpley, A. N., J. L. Weld, D. B. Beegle, P. J. A. Kleinman, W. J. Gburek, P. A. Moore, and G. Mullins. 2003. Development of phosphorus indices for nutrient management planning strategies in the United States. *J. Soil Water Conserv*. 58: 137-152.

USDA-NRCS. 1999. General Manual, 190-GM, Issue 9, Part 402: Nutrient Management. April 1999. Washington, D.C.: USDA Natural Resource Conservation Service.

Veith, T. L., A. N. Sharpley, J. L. Weld, and W. J. Gburek. 2005. Comparison of measured and simulated phosphorus losses with indexed site vulnerability. *Trans. ASAE* 48: 557-565.

Weld, J. L., R. L. Parsons, D. B. Beegle, A. N. Sharpley, W. J. Gburek, and W. R. Clouser. 2002. Evaluation of phosphorus-based nutrient management strategies in Pennsylvania. *J. Soil Water Conserv*. 57: 448-454.

Weld, J. L., D. B. Beegle, W. J. Gburek, P. J. A. Kleinman, and A. N. Sharpley. 2003. The Pennsylvania phosphorus index: Version 1. Publication Cat. US 180 5M3/03PS4591. University Park, Pa.: Pennsylvania State University, College of Agriculture. (http://panutrientmgmt. cas.psu.edu/main_related_programs.htm).

Pp. 171-187 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008.
Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop. St. Joseph, Michigan: ASABE.

Developing Watershed-Scale Tools: A Case Example in the Wisconsin Buffer Initiative

13

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Focusing resource management efforts at the watershed scale is not new. It was a feature of the Organic Administration Act of 1897, an option in the Standard State Soil Conservation District Law of 1936, and the then Soil Conservation Service was brought into watershed management through the Watershed Protection and Flood Prevention Act (PL-566) in 1954. At issue in this chapter are the lessons learned through these historical and contemporary efforts in designing and implementing resource management efforts at the watershed scale.

A common metaphor in this effort has been to employ the expression of using "tools" to achieve the desired resource management objectives. A tool in the broadest sense is a means to accomplish a desired end, but in this context the term refers to the analytical, mechanical, structural and behavioral techniques used in pursuing water quality and conservation objectives. There is an extensive scientific literature that describes, analyzes, and critiques the various tools that can be used in watersheds (see chapter 14). It is important, however, to emphasize that tools are more than just the remedial practices installed or employed in a watershed. The changing nature of how one thinks about the causes of and solutions to degradation in a watershed can also be thought of as an analytical or intellectual tool. This way of thinking about or analytical perspective employed in the study of watersheds can be as important, if not more important, than the most innovative Best Management Practice or Best Available Technology. In short, it needs to be emphasized that a watershed tool is more than a simple technical fix applied to a watershed problem. How one thinks about both watershed degradation and remediation processes is also a tool that can be employed in water quality initiatives. This chapter employs this latter perspective by offering a different type of analytical tool by which to study watershed processes, and then provides an example of a novel application based on insights that emerged in the Wisconsin Buffer Initiative.

Our thesis is that the effectiveness and efficiency of any effort to improve the environmental performance of a watershed is directly related to the spatial congruence between (1) the jurisdictional boundaries specified within program objectives, (2) the spatial dimensions of the degradation processes within the watershed, and (3) the specific spaces addressed by the remedial practices. The effectiveness of any water quality program, and the efficiency by which it is implemented, is related to the extent and accuracy with which the program specifies the spatial and temporal distribution of the degradation processes, and the capacity of the tools employed to address these distributions. Tools are viewed as the process by which one attempts to enhance or increase the spatial congruence between these three components of any watershed initiative. This connotes that an effective and efficient watershed program will specify those specific portions of a watershed where the greatest degradation is occurring, and where remediation is feasible with available and compatible tools. For example, an ineffective and inefficient program would have programs focusing on farms, and remedial practices implemented on fields, while degradation processes are occurring at the sub-field scale. Achieving the greatest effectiveness in the most efficient fashion in the use of watershed tools will occur when there is scalar congruence between policy, remedial tools, and the degradation processes themselves. Scalar congruence, or emphasizing the point that space matters, can be used to organize how one analyzes or thinks about watershed processes.

The Rationale for a Spatial Congruency Thesis

The core scientific principle behind the spatial congruency thesis just proposed is that degradation within any watershed is not random. It is spatially and temporally patterned. The spatial and temporal pattern that emerges is highly dependent on the interaction between the appropriateness of the management behaviors or activities occurring and the relative vulnerability or resiliency of the biophysical attributes in the specific location where this interaction occurs. The non-random pattern emerges from the interaction between the social and the biophysical attributes that characterize the watershed. These patterned interactions change across time as climatic events, agronomic cycles, market forces, and technology change the values associated with the appropriateness of the behavior and vulnerability of the site where this behavior occurs. The concept of watershed tools, therefore, needs to address this dynamic spatial and temporal nature of degradation processes in agricultural watersheds.

The social attributes found in this dynamic interaction are often simplified in models used in watershed activities by focusing on the average, typical, or recommended behavior. That is, rather than allowing the full range of behavior to be reflected in model parameters, it is often assumed that land user behavior follows recommended guidelines. This approach purposively limits variation to be accounted for by the model, and in effect, allows the biophysical measures of vulnerability to not only dominate, but limit the characterization of watershed processes. However, it is possible that more attention needs to be given to the exceptional, rather than the average, when designing and implementing watershed tools that explicitly involve the human dimension. Paying attention to what might be termed statistically exceptional behaviors within a watershed recognizes the potential for disproportionate impacts on system processes.

Disproportionality may occur within a system to the extent that high-impact; extreme events (Albeverio et al., 2006) of low frequency dominate the system's behavior. Infrequent but high-impact events can either directly determine system outputs, or structure the conditions within the system so that the consequences of the event continue to influence the system long after the extreme event has ended; that is, a legacy impact (Bazzaz et al., 1998; Palmer et al., 2004). Acknowledging that disproportionality is a form of an extreme event could have significant long-term implications for USDA and USEPA efforts to induce improvements in the quality and quantity of the nation's water bodies (Nowak et al., 2006). Unfortunately, the agricultural behaviors of land users are not all normal, average, or within recommended parameters. Research has found that distributions of behaviors that are especially salient to resource degradation processes (i.e., fertility practices) are often skewed so as to represent lognormal probability distributions (Shepard, 2000). If the behaviors represented by the "tails" of these distributions occurred in a particularly vulnerable biophysical place or time, then it is highly probable that these few locations would contribute disproportionate impacts on overall watershed performance. It is a situation where the "tail could be wagging the watershed" or where the exceptional behavior needs to become the focus of watershed tools.

As has been noted, disproportionality may emerge from the interaction of how a management practice is implemented (e.g., tillage, manure application) relative to the spatial and temporal biophysical settings (e.g., field unit, time of year) of these decisions (Nowak et al., 2006, p. 156). These biophysical settings exhibit variability in their particular likelihood to generate runoff or resiliency for buffering water quality impacts during runoff events. Given the potential variability in contributions from the biophysical settings, the issue of the appropriateness of the specific management practices used in those settings must be a focal point of any watershed initiative. Scalar congruence is achieved by ensuring that the program focuses watershed tools on those situations where inappropriate behaviors in vulnerable locations are causing disproportionate water quality or quantity impacts relative to overall frequency of occurrence (fig. 13-1).

The challenge is that scalar congruence is difficult to achieve when extreme situations exert a critical impact on water quality, and consequently need to be reflected in watershed research, modeling, and management efforts. Both management practices and the biophysical resiliency of the settings where these actions occur may be described in terms of their probability of occurrence (Nowak and Cabot, 2004). Disproportionality is then a function of the magnitude of the multiplicative effect of these probabilities on overall water quality, as disproportionately large impacts will occur when inappropriate behaviors occur in vulnerable locations or times (fig. 13-2). A critical and as yet unmet research need is an assessment of the optimal spatial scale at which to examine this interaction. That is, one could look for and address disproportionality at spatial scales ranging from sub-meter or even sub-field to the hydrologic basin scales. Finding the optimal scale conducive to effective federal, state, or local programs will be a complex issue, as the factors leading to this decision are dynamic. Both the appropriateness of a behavior or activity and the vulnerability of a location will vary across time due to short-term and long-term climatic variation, changes in agricultural technologies, and our increasing abilities to monitor and measure forms of degradation.

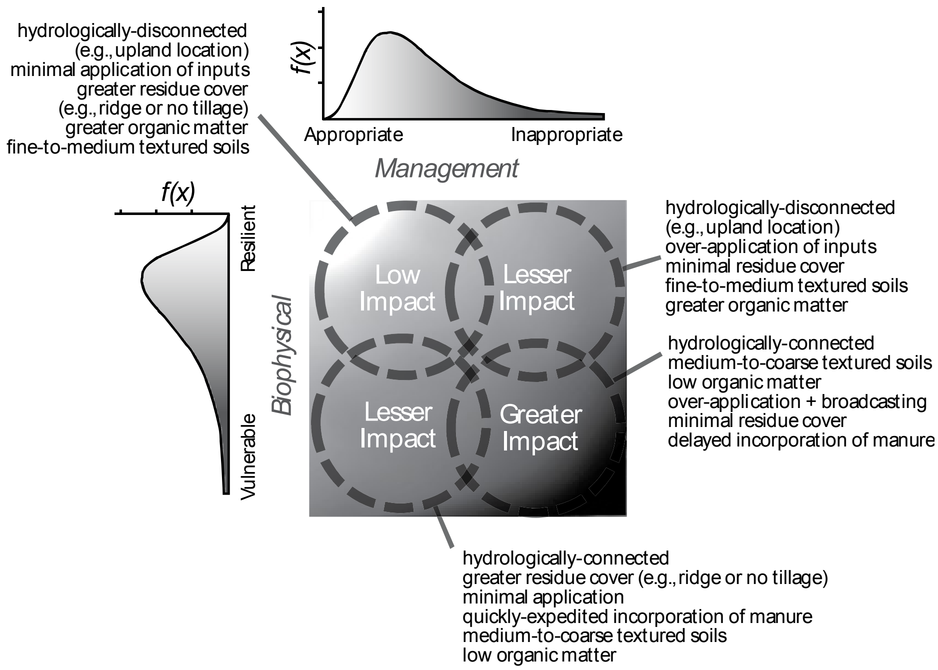

Figure 13-1. Watershed impact of the multiplicative outcome of the social and biophysical.

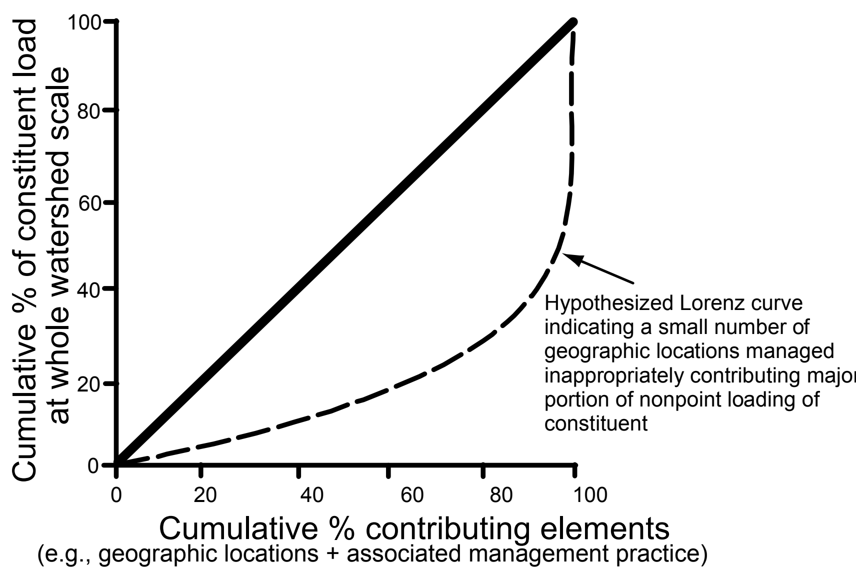

Figure 13-2. Disproportionality hypothesis for nonpoint origins of constituent delivery (e.g., P runoff from agricultural fields).

The argument up to this point is that “space and time matter” in the design and implementation of both programs and remedial tools used in watersheds. This spatial dimension is important because it is highly probable that a small proportion of inappropriate behaviors in a small proportion of biophysically vulnerable areas are driving overall watershed water quality parameters. Recognizing the occurrence and salience of disproportionality raises a challenging question: should we design tools for the average, or should we design for the exceptional? For example, should the policies and remedial practices that attempt to address sediment transport to Lake Pepin in the Mississippi River along the Minnesota-Wisconsin border be designed for the entire Upper Mississippi Basin, or that small proportion of the entire basin that contributes a disproportionate amount of sediments?

Another reason why “space and time matter” is that many watershed remedial practices are incapable of optimizing remediation or prevention across forms of degradation. While there are ample watershed tools available for specifically addressing, for example, sedimentation, nitrogen leaching, or wildlife habitat, there are few practical tools available that are capable of addressing multi-media forms of degradation. A related theme is the trade-off that may occur when addressing one form of degradation, which then results in increasing the degradation from another form. For example, finding a “solution” for phosphorus transport at one place in a watershed may exacerbate nitrate-N transport at this same location. While there is ample discussion of “systems” in the watershed literature, much of this discussion has not been translated into practical watershed tools capable of addressing systemic issues. At minimum, more attention needs to be given to various optimization strategies to avoid disproportionality from occurring in one medium when addressing multi-media forms of degradation. In short, a critical policy question is whether we should accept the trade-off of addressing a disproportionate contribution in one media while accepting a modest increase in another media.

A final reason why “space and time matter” is that the processes or outcomes occurring at any particular location within a watershed change across time. All available watershed tools, implicitly or explicitly, are impacted by uncertainty. The stochastic variation found in climatic processes, behavioral patterns, technological changes, and cross-scale nonlinearities has required that the resulting uncertainty be addressed through simplifying assumptions in our approaches to designing and implementing watershed tools. Yet these simplifying assumptions (e.g., models based on the unrealistic assumption that all land users are adhering to recommended practices and rates in a uniform fashion across the space being modeled) are rarely the focus of research on watershed processes that result in the development of tools. It is possible that some of the more important breakthroughs in the development of innovative watershed tools may be found by examining the underlying assumptions of our current approaches. Specifically, more attention needs to be given to understanding salient watershed processes that explicitly addresses the dynamic behavioral patterns of the land user.

We believe that giving more attention to the potential for disproportionate contributions occurring within specific spatial and temporal frameworks may, in itself, be a valuable watershed tool. An example of how the concept of disproportionality can influence watershed management activities occurred when the state of Wisconsin began to look for new ways to address water quality degradation. Rather than being an abstract discussion, these questions concerning the role of disproportionality, scale of appropriate tools, and the role of science in developing nonpoint policy were explored across a three-year process called the Wisconsin Buffer Initiative (WBI).

The Wisconsin Buffer Initiative

The Wisconsin Buffer Initiative emerged in response to a political controversy and evolved into a process in which some of challenging spatial congruency questions raised earlier in this chapter were addressed directly or indirectly. What resulted from this process was a unique watershed tool. That is, the analytical perspective and recommendations recognized the possibility of disproportionality, attempted to optimize across different potential watershed objectives, and explicitly addressed uncertainty through an adaptive management framework.

Re-designing Wisconsin's nonpoint agricultural pollution abatement policy was the context for the WBI. Controversy emerged over the role of riparian buffers during the legislative deliberations and public hearings on the re-design of the nonpoint pollution program. Some argued for standard-width (i.e., 30 ft) riparian buffers to be mandated for all the perennial rivers and streams in Wisconsin. Others argued that existing federal and state programs that promote riparian buffers were adequate to address the overall objectives of the nonpoint program. Polarization on this issue in the Wisconsin Legislature and among the elected or appointed natural resource decision bodies threatened to bring the re-design process to a halt.

Resolution of this conflict was sought by the Wisconsin Natural Resource Board, which approached the University of Wisconsin (UW) and asked for recommendations on how the application of "best available and complete science" could be used to determine where in Wisconsin's diverse agricultural landscape riparian buffers would have the greatest impacts on water quality. The UW was given a little over three years to meet this charge with a final product to be delivered on or before December 31, 2005. The response to this charge was the formation of a working group that included representatives from all the vested interests that had been involved in the conflict. Approximately twelve major environmental groups, agricultural organizations, conservation professional associations, and other salient non-governmental organizations were invited to participate in this process. University scientists from a variety of disciplines and representatives from state and federal agencies were also invited to participate (see the Appendix for a list of participants). Participation was organized in accord with a civic science approach in which all parties were treated as equals. That is, it was not the typical citizen participation process, where the scientists provide their science-based recommendations with the expectation that local interest groups accept these conclusions. The meeting began with a blank slate, other than the charge from the Natural Resources Board. Much of the time at the initial meetings was spent addressing the stereotypes and perceptions that these various vested interests had of each other in a constructive fashion. Moving beyond the past history of confrontation allowed for an open dialogue on what questions needed to be addressed, what would be a credible methodology to use in addressing these questions, and what type of information was needed to address the charge to the WBI. The scientists involved did not receive a clear charge on needed research on specific questions until after a full year of WBI meetings had been held.

Early meetings of the WBI were also spent discussing program expectations, prior findings in the scientific literature, available tools, and data availability. Three decisions were made in this process. First, it was agreed that vegetative strips by themselves adjacent to streams or rivers were not adequate to address the complex forms of degradation occurring across the Wisconsin landscape. The group rejected the idea that riparian buffers are a "silver bullet" that would solve the state's agricultural nonpoint pollution problems. Instead, the participants in the WBI adopted a conservation systems approach to acknowledge that a complement of practices would have to be applied to the hydrologic contributing area of specific segments along a river or stream. Thus, the typical approach of recommending a uniform width buffer was rejected in favor a spatial and topographic approach for identifying upland areas where a conservation systems approach needed to be applied.

The second decision was that fiscal constraints would prevent implementation of this approach on a wide basis. Consequently, the WBI explicitly addressed the scale question at the state, watershed, field, and sub-field levels in establishing priorities for where buffer technology needed to be implemented. This was based on engaging in an assessment process to determine where implementation of a conservation system including riparian buffers would have the greatest likelihood of inducing improvement in water quality. This second decision was accompanied by a significant amount of discussion, as it implied that severely degraded or exceptional waters would receive a lower ranking than those watersheds with a higher probability of responding to the installation of riparian buffer systems. In other words, contrary to current policy, the WBI approach placed a lower weight on severely degraded waters (e.g., the 303d or TMDL locations), as buffers would probably have little impact on these waters. In the same way, some of the exceptional cold water trout streams in Wisconsin would receive a lower ranking, as buffers would have little impact on stream water quality.

Third, a decision was made that the current recommendations be based on the prevalent science at this time, although this was not deemed sufficient to achieve long-term improvement in the state's waters. An adaptive management framework was recommended because of its ability to learn and adapt based on the consequences of earlier actions. Such a process is designed to address incomplete understanding of cause and effect relationships, and accommodate the "surprises" that may emerge due to changing circumstances.

These early decisions resulted in three general questions that guided both the research and the discussion of the final recommendations. These questions were:

1. How can we identify the hydrologic units most likely to show demonstrable improvements with investment in riparian buffers as part of a larger conservation system?

2. What types of tools can be developed that can be employed at the local level to assist in identifying portions of watersheds where a buffer-based conservation system would be an effective option?

3. How do we develop techniques for determining the optimal placement and configuration of buffer-based conservation systems on designated landscapes?

The remainder of this chapter will describe the responses developed to date regarding these three questions.

Identifying the Appropriate-Size Watershed

Watersheds vary in spatial scale due to their nested nature. Selecting the appropriate spatial scale to be both a focal point of policy and appropriate to the potential tools that can be employed is a critical decision. Successfully implementing a remedial program in a large watershed could produce significant environmental benefits, but it will be very expensive for a number of reasons. Implementing a program in a small watershed could be very cost-effective, but environmental gains will be highly variable and probably minimal relative to the larger basin. Time is also a critical factor in selecting the size of the watershed to be targeted. Small watersheds are easier to organize, and results from implementation activities will be detected earlier, but small watersheds are also more dynamic in response to external influences, such as development or new technology. Large watersheds are much more complex relative to the time needed to gain political support, and for any results to be measured with a monitoring effort. Yet WBI participants agreed that a focus on smaller watershed units would be more beneficial for two reasons. First, the adaptive management process requires some form of monitoring or feedback. Measuring the impacts of installing buffer systems would be more direct in smaller watersheds, as the impacts are less likely to be masked by other activities or legacy processes. Second, it was agreed that in smaller watersheds, it would be more likely to get local land owners to accept ownership of their waters because of familiarity with the geography involved. The overall relation between watershed size and any measure of effectiveness or efficiency is represented in figure 13-3.

The characteristics of the resulting WBI watersheds relative to more familiar watersheds are presented in table 13-1. At the coarse scale, Wisconsin has 42 USGS 8-digit HUCs watersheds with an average size of 3400 km^2. The Wisconsin Priority Water-

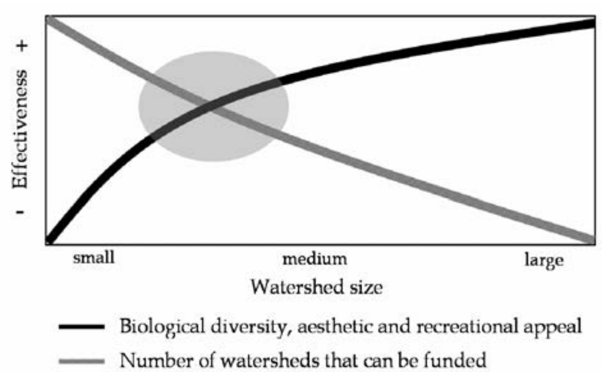

Figure 13-3. Conceptual illustration of tradeoffs in various aspects of program effectiveness across a range of watershed sizes. The gray oval indicates the size that maximizes the average effectiveness across criteria.

Table 13-1. Comparative watershed size and number.

USGS 8-digit HUCs	Wisconsin DNR Watersheds	WBI Watersheds
N = 42	N = 334	1598
3400 km^2	434 km^2	47.4 km^2
1312 mi^2	167 mi^2	18.1 mi^2

shed Program, the program that was the focus of the re-design effort, was based on subdividing these 42 USGS watersheds into 334 watersheds, each of an average size of approximately 434 km^2.

The WBI watersheds are hydrologically complete and developed on all the third-order and some fourth-order streams dominated by agricultural activities in Wisconsin. This resulted in 1598 watersheds being delineated, with an average size of approximately 47 km^2. As noted, the decision was based on selecting a size at which it would be feasible to determine if the watershed responded to the implementation of buffer systems, being small enough that the watershed would be viewed as manageable by local staff and residents, and being congruent with available data and other salient information. All watershed boundaries were identified in a GIS layer for further analysis.

Criteria for Ranking WBI Watersheds

The next question faced in the WBI process was "responsive to what?" That is, before deciding on appropriate watershed tools, it is first necessary to determine what types of degradation will be the focus of the intervention effort. It was explicitly acknowledged in the WBI discussions that the selection of specific forms of degradation is a political decision. An important distinction raised in these discussions was that the targeting would not be based on the level of degradation, but on the probability of a positive response to the implementation of buffer technology. In short, the goal was not to maximize the number of buffer systems installed, but to install buffer systems where they would have the greatest impact on the chosen forms of degradation. For the WBI, three different forms of degradation, or criteria, were finally selected to screen the 1598 watersheds. These were sediment and phosphorus (P) loads, protecting and enhancing native biological communities, and the trophic status of lakes, reservoirs, or impoundments down-gradient from the watershed. Other criteria were proposed (e.g., biodiversity, wildlife habitat, etc.), but political consensus could only be achieved on the three listed above. A spatially specific analysis was then conducted for each of these criteria for each of the 1598 watersheds.

Weighting of each individual WBI watershed was based on the following calculations:

1. Predicting potential reduction in nutrient and sediment loads was based on a regression model developed around land use and watershed loading data derived from the USGS and other monitoring sources. In each of these watersheds, there was an attempt to quantify sources where buffer-based conservation steams would significantly reduce P and sediments. Sources of P and sediments associated with non-agricultural areas (urban or suburban) or sources associated with stream characteristics (bank slumping and stream bed erosion) had to be estimated and subtracted from the total watershed load (see the WBI Final Report for technical details at http://bombadil.lic.wisc.edu/WBI/index.htm).

2. The potential response of biological communities to conservation systems was developed around sediment-sensitive fish species. The aquatic biologists who participated in the WBI reported that sediment-sensitive fish species are a good indicator for a wide range of other aquatic organisms. This response was calculated for each of the 1598 watersheds by examining trends in the counts of 19 sediment-sensitive fish species. Other factors associated with stream temperatures and cover was also considered. These data were used to predict potential species distributions and to assess the potential for biological community response to sediment reductions.

3. Most rivers and streams in Wisconsin flow into or through a lake, impoundment, or reservoir. Because of this fact, the watersheds were also ranked based on the capacity of the lake, impoundment, or reservoir to receive additional P and sediments relative to its trophic status. A rating was assigned to each WBI watershed by calculating the potential for attenuation or prevention of eutrophication based on current water body conditions, monitoring data, and the likely response to reductions in P from contributing streams. Water bodies that were closer to the threshold of moving from eutrophic to a hyper-eutrophic state were rated higher than those below or above this point. Again, significantly degraded or exceptional water bodies received a lower score than those near this threshold.

Each of these three ranking criteria was then integrated in a GIS layer representing a composite ranking for each of the WBI watersheds. This allowed for a rank order listing of all 1598 watersheds in the state, from those most likely to respond to implementation of buffers as part of a conservation system to those least likely to respond (figure 13-4).

It can be argued that the analytical procedures used to produce this ranked list of watersheds increased the spatial congruence of nonpoint program objectives relative to the selected degradation processes occurring in these watersheds. Any resulting political or administrative decision regarding the amount of funds, distribution criteria, or the types of policy mechanisms to be employed (i.e., voluntary versus regulatory) can be structured on the basis of the rank order of the "probability of response" to intervention efforts. The next step in enhancing the spatial congruence was to specify implementation procedures within selected watersheds.

Planning and Implementation at the Local Level

As just discussed, the WBI developed a ranked list of 1598 predominantly agricultural watersheds based on the probability of a positive water quality response to riparian buffers as part of a larger conservation system. This ranking was based on three criteria for which there was consensus among the members of the WBI advisory committee. The WBI collaboration also had to develop a set of procedures and tools that could be used in any of the watersheds. Because of the diversity of interests associated with the WBI process, it was decided that a series of conditions should be addressed in deciding what tools need to be deployed in whatever watersheds are selected. The consensus was that whatever tools are selected should incorporate local knowledge and build on local expertise and experience. The watershed tools to be selected cannot solely be a top-down, science-driven set of procedures, but must address

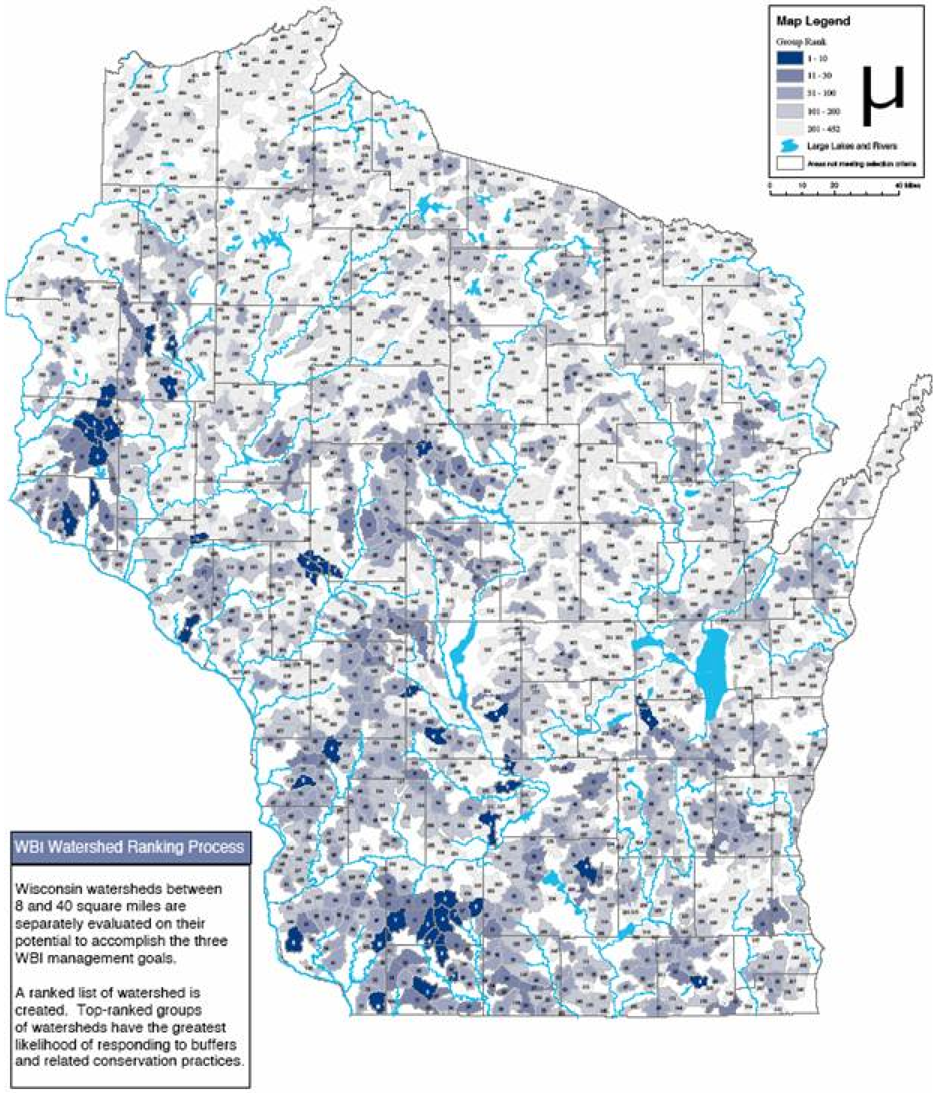

Figure 13-4. The WBI ranked watersheds.

indigenous knowledge and local capabilities. Moreover, the selected tools need to recognize that these efforts are not occurring in a resource management vacuum. Instead, they need to be compatible with ongoing conservation and nutrient management planning efforts. In essence, the agreement was that the state agencies would not come into the selected watersheds and implement the re-designed program, but complementary tools and procedures would be developed that allowed local interests to also address areas of concern that were identified as part of the WBI process.

These decisions led to seeking out databases that would be universally available at the local level, and would involve activities that would be familiar to local conservation staff. Initially, this resulted in four common sets of information requirements: digital elevation data, digitized soils data, land use information, and stream loading data. All of this information would be made available in a web-based format that could be accessed by local officials and staff. This internet mapping site would be used to convey analysis results, support "what if" analyses, and provide data access (prototype available at: http://144.92.119.47/website/opener.htm).

There is a significant amount of variation in the resolution of digital elevation data across Wisconsin, and therefore it was decided that the next step within the watershed should be based on the universally available 30 m digital elevation models (DEM). The USDA-NRCS SSURGO digital soils data are also universally available. Digital land use data were also deemed to be readily available from such diverse sources as recent satellite imagery available through the University of Wisconsin, USDA-FSA offices, or local initiatives associated with local government (e.g., planning and zoning departments). The stream data are more widely variable, as USGS data are only available in selected locations, and the monitoring that accrued as a result of the previous Nonpoint Priority Watershed Program is also variable.

All these data are to be used to determine priority areas within the selected watersheds; that is, those locations that have the highest levels of vulnerability to disturbance, based on soils, slope, and land cover. Assessment of the need for buffers would begin in these limited areas by evaluating the appropriateness of land user behavior occurring on these sites. Thus, the first level of spatial targeting was to establish procedures that would allow the ranking of small-scale watersheds in Wisconsin based on the probability of a positive response to the implementation of buffer systems. The second level of spatial targeting was designed to address potential disproportionate contributions from specific areas within the watershed, where local staff and citizens would initially concentrate their efforts. Limiting the area within any targeted watershed would allow local staff to focus resources and efforts on those areas where there is the highest probability for degradation and the greatest probability of a meaningful response when a buffer system is installed.

An example of this spatial targeting within a WBI watershed is illustrated in figure 13-5. Figure 13-5 illustrates that available data can be used to identify the most biophysically vulnerable areas in a watershed. The WBI suggested that the implementation of buffer systems would be most efficient if initial analyses for inappropriate land management occur in these high-vulnerability areas before proceeding to the rest of the watershed. This initial map will be reviewed by local conservation staff, who may be aware of local efforts or situations that are not represented in the initial representations of potential priority areas.

The next critical question addressed in the WBI process was: what tools will be used to judge what constitutes an appropriate or inappropriate behavior? It was decided to use a field-scale assessment tool (see chapter 12). In Wisconsin, this assessment will be built around the SNAP+ planning tool. This tool incorporates a phosphorus index (PI) with erosion calculations to provide a series of management options to

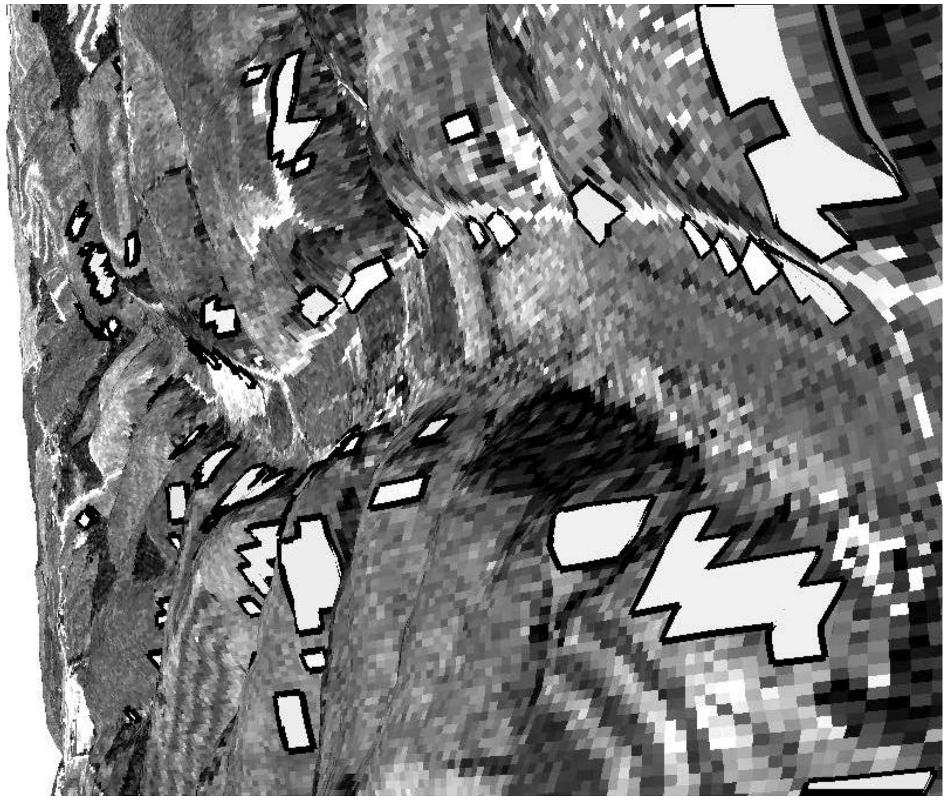

Figure 13-5 (see inside back cover). Example of priority areas for assessment within a WBI watershed (from a manuscript in preparation by L. W. Good and J. T. Maxted: "Estimating soil and hosphorus loss potential in a small agricultural watershed").

the land owner. Fields selected through this initial screening process will be further evaluated by obtaining soil test, crop rotation, and manure/fertilizer management data. The PI portion of this tool will have an important function to play in this field assessment, as a PI value greater than 6 implies that intervention is needed (e.g., inappropriate behavior is occurring). Only on those fields with a PI greater than 6 will riparian buffers, as part of a larger conservation system, be considered. Moreover, the land owner will have options within the SNAP+ that will allow them to change current practices (e.g., tillage, tilling on the contour, rotation, changes in manure distribution patterns), thereby reducing the PI below the value of 6 and negating the need for a buffer system. This "what if" planning capability is an important part of the selected buffer implementation strategy, as it gives the land owner a number of options to meet the phosphorus or soil erosion standards.

This has proven to be a contentious point in the WBI process, as some conservation staff wanted an objective buffer standard (i.e., mandate a standard 30-foot buffer). Yet the logic of the WBI recommendation is that any riparian buffer has to be part of a larger conservation system to be effective. Installing a vegetative strip adjacent to a stream without addressing what is happening in the upland contributing area increases the probability of buffer failure. Consequently, the WBI process recommended that the upland contributing area must meet existing PI and soil erosion standards. If this is not possible, then practices have to be implemented in the contributing area to the stream reach under consideration to reduce these values as much as is possible. Only then will a riparian buffer be designed to address any remaining runoff processes. This has come to be referred to as a strategy in which riparian buffers are viewed as the "last line of defense" in a systems approach, rather than the one-size-fits-all "only line of defense."

Placement and Design of Riparian Buffers

A riparian buffer will be designed only in those circumstances where the PI is greater than 6, erosion rates exceed the soil loss tolerance value within a contributing area to a specific segment of a stream reach, and all feasible changes in current farming practices have occurred. An important contribution of the WBI process was that buffers will be designed to explicitly address the contributing area, rather than the current NRCS field office guidelines (i.e., 393 Standard). The importance of the contributing area was developed in the WBI process through the application of the Precision Application Landscape Modeling System (PALMS). PALMS research in Wisconsin on Discovery Farms (Molling et al., 2005) demonstrated that standard-width buffers are highly vulnerable to breaching by concentrated flow in select locations along the buffer. Consequently, the conservation system begins in the upland contributing area, and the buffer itself must be designed to prevent concentrated flow from developing in the contributing area. This will require applying the conservation systems perspective up across the landscape, possibly considering neighboring fields.

The design and placement of these buffer-based conservation systems is based on diffusing water and energy in the higher areas of the landscape rather than trying to control and mitigate this energy in the riparian zone. The WBI recommendation thereby becomes a constellation of practices organized by topographic features. Realization that the classic "ribbon model" of riparian buffers would not achieve the goals of the WBI evolved from recognition that the effectiveness of any watershed tool is highly site-specific. Focusing the placement of these buffer-based conservation systems in areas of the agricultural landscape that have the greatest likelihood of causing degradation specifically addressed the charge from the Natural Resources Board to consider effectiveness and efficiency.

Conclusions

The field-specific design and placement of buffer technology within ranked Wisconsin agricultural watersheds is a direct result of a policy process that asked where across the Wisconsin landscape riparian buffers were needed to achieve water quality objectives. The responses to this question emerged from a three-year civic science process. This collaboration was guided by a spatial congruency process; that is, it was believed that the effectiveness and efficiency of the resulting WBI recommendations will be directly related to the spatial congruence within (1) political and administrative decisions that encourage variability in program implementation, (2) the ranking of salient degradation processes within Wisconsin watersheds, and (3) the ability of locally implemented tools to identify and address the most biophysically vulnerable locations where inappropriate land user behaviors are occurring.

Perhaps the most important lesson is that agricultural scientists and the scientific process should not be limited to designing watershed tools based on the best available science, and then go looking for an application situation. Development of any watershed tool is explicitly a political process. As the WBI process demonstrated, determining which forms of water quality degradation should be addressed can be the outcome of a collaborative process involving both scientists and representatives of a wide range of vested interests. A critical stage in this process occurs when scientists characterize the magnitude and spatial and temporal features of the form of degradation selected. While it may be mundane in scientific circles to say that "space and time matter," it is quite another matter to bring this knowledge into the political process. Establishing and ranking watershed boundaries, which can be used for program implementation and that incorporates both an estimate of the relative contribution of pollutants and the probability of a positive response to intervention efforts, can only occur if scientists contribute to the political process.

Agricultural scientists are often frustrated when the popular media or other nonagricultural interests characterize agriculture as the major source of some form of environmental degradation. While this gross characterization may be valid, scientists also recognize that the origins of this degradation are highly variable in space and time, and in the underlying social-economic causes. Hence, the challenge that faced the WBI process was to determine how to best estimate this variability in a way that could be used in designing and implementing remediation efforts. This required WBI scientists to take complex disciplinary knowledge and present (i.e., simplify) it in a way that would allow targeting across several spatial scales in an effort to achieve spatial congruency. The challenge was to find techniques of communicating science that allowed all WBI participants to recognize that not all agricultural watersheds are equal in terms of the state's nonpoint objectives, not all portions of any selected watershed have equal vulnerability to the processes that may contribute to degradation (an appropriate behavior in one part of a field may be inappropriate in another), and that there is significant variability in the functioning of any riparian buffer unless one also addresses the contributing area.

For a layperson, the spatial congruency hypothesis simply states that programs need to be designed so as to place the remedial actions at the most significant sources of the problem. Yet translating this "common sense" approach into the realm of agricultural nonpoint-source pollution was a challenge for both the scientists and laypersons involved in the WBI collaboration.

The final political decision will be acceptance of the WBI recommendations, and the degree to which implementation efforts are funded. The point being made is that watershed tools are not purely an artifact of the latest scientific advances. The question of whether there will even be an opportunity to even use analytical, mechanical, structural, or behavioral techniques to pursue a conservation objective is a political decision. Politics is not just a funding source or the point of origin for new conservation policies. The WBI collaboration demonstrated that scientists can address political considerations, from the federal down to the local, in the selection, design, or use of any watershed tool.

Another lesson reaffirmed in the WBI process is that we will probably never create any watershed tool that does not contain significant levels of uncertainty. Hence the need for an adaptive management approach to any application of watershed tools. Implementation, monitoring, and adjustment form a process that determines our ability to answer questions such as: where and what types of intervention are needed, to what extent did we achieve our goals, or how did surprises or extreme events influence the performance of our tools? Adaptive management is a logical response to uncertainty. Feedback mechanisms can be used to assess both program and tool performance. A long-standing obstacle to this approach has been the costs associated with monitoring. The WBI addressed this situation by selecting relatively small watersheds where there is a greater likelihood of measuring changes associated with the installation of buffer systems. The sophistication associated with this monitoring effort is still being discussed by WBI participants and involved agencies. Various designs are being explored relative to optimizing the data needed to address three objectives: (1) collect data in a scientifically rigorous and valid fashion, (2) minimize personnel, equipment, and laboratory expenses, and (3) be capable of demonstrating to local land owners that the conservation practices actually work and have a positive impact on water quality.

The final step in the WBI process currently underway is to test the set of recommendations in a matched watershed experiment. Two relatively high-ranked WBI watersheds have been selected that are in close geographical proximity to each other. Monitoring equipment has been installed in both watersheds, with one designated as a reference watershed while the other will be asked to follow WBI recommendations. Negotiations with local staff are underway, with the initial assessment for inappropriate behaviors scheduled to begin post-harvest in the fall of 2007.

References

Albeverio, S., V. Jentsch, and H. Kantz, eds. 2006. *Extreme Events in Nature and Society*. Springer: Berlin, Germany.

Bazzaz F., G. Ceballos, M. Davis, R. Dirzo, P. R. Ehrlich, T. Eisner, S. Levin, J. H. Lawton, J. Lubchenco, P. A Matson, H. A. Mooney, P. H Raven, J. E. Roughgarden, J. Sarukhan, D. G. Tilman, P. Vitousek, B. Walker, D. H. Wall, E. O. Wilson, G. M. Woodwell. 1998. Ecological science and the human predicament. *Science* 282: 879.

Good, L. W., and J. T. Maxted. Estimating soil and phosphorus loss potential in a small agricultural watershed (manuscript in preparation).

Molling, C. C., J. C. Strikwerda, J. M. Norman, C. A. Rodgers, R. Wayne, C. L. S. Morgan, G. R. Diak, and J. R. Mecikalski. 2005. Distributed runoff formulation designed for a precision agricultural-landscape modeling system. *J. American Water Resources Assoc*. 41(6): 1289-1313.

Nowak, P., and P. E. Cabot. 2004. The human dimension of resource management programs. *J. Soil Water Cons*. 59(6): 128A-135A.

Nowak, P., S. Bowen, and P. E. Cabot. 2006. Disproportionality as a framework for linking

social and biophysical systems. *Society and Natural Resources* 19: 153-173.
Palmer, M., E. Bernhardt, E. Chornesky, S. Collins, A. Dobson, C. Duke, B. Gold, R. Jacobson, S. Kingsland, R. Kranz, M. Mappin, M. L. Martinez, F. Micheli, J. Morse, M. Pace, M. Pascual, S. Palumbi, O. J. Reichmann, A. Simons, A. Townsend, M. Turner. 2004. Ecology for a crowded planet. *Science* 304: 1251-1252.
Shepard, R., 2000. Nitrogen and phosphorus management on Wisconsin farms: Lessons learned for agricultural water quality programs. *J. Soil and Water Cons.* 55(1): 63-68.

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Pp. 189-212 in UMRSHNC (Upper Mississippi River Sub-basin Hypoxia Nutrient Committee). 2008. *Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop.* St. Joseph, Michigan: ASABE.

Limitations of Evaluating the Effectiveness of Agricultural Management Practices at Reducing Nutrient Losses to Surface Waters

14

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Hypoxia in the Gulf of Mexico is a serious problem. Excessive transport of nitrogen and phosphorus to the gulf from the Mississippi River contributes to growth of the hypoxic area (Goolsby et al., 1999). The Gulf of Mexico Hypoxia Task Force set a goal for a 30% reduction in the area of the hypoxic zone, and a variety of ways to reduce the area of the hypoxic zone by 30% have been proposed (Mitsch et al., 2001). These measures include reductions in nitrogen applications to cropland, restoration of wetlands, installation of riparian buffer strips, and improvements in nitrogen treatment processes at wastewater treatment plants. Ultimately, the effectiveness of management practices at reducing hypoxia in the Gulf of Mexico will be determined from repeated annual measurements of the hypoxic zone area.

Upstream of the Gulf of Mexico, state and federal agencies have been working to identify and remediate impaired lakes and rivers through the framework of total maximum daily loads (TMDLs). A significant proportion of the TMDLs are for eutrophication of water bodies from excess nutrients such as nitrogen and phosphorus (USEPA, 2007). TMDLs typically represent the maximum mass of phosphorus or nitrogen that can enter the water body without violating water quality standards.

Nonpoint-source pollution is responsible for a large proportion of the impairments in surface water bodies throughout the upper Midwestern region. Sediment, phosphorus, or nitrogen lost from agricultural fields can collect in reservoirs and other waterbodies to produce significant water quality impairments. A variety of best management practices (BMPs) have been developed to reduce the losses of sediment, phosphorus, and nitrogen from agricultural fields. A BMP can be defined as a practice or combination of practices that is technologically and economically effective in reducing pollutant loads generated by nonpoint sources to a level that meets water quality goals (USEPA, 1980). Typically, a BMP both reduces the pollutant load and maintains agricultural productivity. Methods are needed to evaluate the impact of BMP implementation on water quality in the upper Midwestern region.

Tracking Implementation of Agricultural BMPs

The USDA has several ongoing programs designed to reduce the impact of agricultural management practices on water quality and improve wildlife habitat. These include the Conservation Reserve Program (CRP), Environmental Quality Incentives Program (EQIP), and Wetlands Reserve Program. A new program called the Conservation Security Program (CSP) rewards farmers for existing BMPs. States have developed a myriad of complementary programs to help fund the implementation of BMPs. For example, Minnesota has the Conservation Reserve Enhancement Program (CREP), the Reinvest in Minnesota (RIM) program for wetland restoration, and the Agricultural BMP Loan program. As a result of these programs, BMPs have been implemented on thousands of acres of farmland, leading to large increases in the acreage of conservation tillage, better manure management practices, improved nutrient management practices, retirement of highly erodible farmland, and restoration of wetlands.

Questions have been raised regarding the effectiveness of the money used to fund these programs, and what impacts these programs have had on water quality and wildlife habitat. In response to questions raised about the need for soil conservation programs, the USDA initiated the statistically rigorous National Resources Inventory (NRI) in 1982 to track land use changes, soil erosion rates, and wetland areas at five-year intervals (USDA-NRCS, 1997). From 1982 to 1997, the NRI documented a 30% reduction in erosion rates from wind and water. From 1992 to 1997, NRI data showed that 48,400 acres of wetlands had been restored in the upper Midwestern region, while 74,200 acres of wetlands were lost during the same period. About half of the wetland losses were attributed to agricultural practices.

Many states track implementation of BMPs to assess the effectiveness of conservation programs. For example, in Minnesota, the Board of Water and Soil Resources (BWSR) has developed an extensive database of BMP implementation (BWSR, 2005). This database lists the type of BMP implemented, the location of the BMP, the acreage or area affected, the cost of the project, and the predicted reduction in pollutant load where appropriate. Predicted reductions for erosion are based on the Revised Universal Soil Loss Equation (RUSLE) and do not represent reductions in sediment delivery. From 1998-1999, Minnesota had almost 6000 projects implemented at a cost of $26 million. Reductions in soil erosion were estimated at 777,000 tons year^{-1}, and reductions in phosphorus loss were estimated at 438,000 lb year^{-1}. In contrast, a recent study of phosphorus export to Minnesota surface waters under average climatic conditions estimated that 3.9 million lb of phosphorus is exported from agricultural land every year, while 14.9 million lb of total phosphorus is exported to surface waters from all point and nonpoint sources (Barr Engineering, 2004). Comparing these assessments, we conclude that implementation of BMPs has at most reduced phosphorus export from agricultural land by 11%. Thus, the effects on water quality in a given watershed are generally limited and would be difficult to quantify using water quality monitoring.

Statistical surveys of agricultural commodities at the county level have been conducted by the U.S. Census Bureau and the USDA National Agricultural Statistics Service (NASS) since 1840. The USDA currently tracks several agricultural indicators that could be potentially related to water quality. These include crop acreages, animal production numbers, crop harvests, chemical and fertilizer usage patterns, land market values, and farm income patterns. These data have been used to estimate inputs of nitrogen and phosphorus on agricultural lands in the upper Midwest for regional assessments of areas with the greatest potential for export of nutrients to surface water bodies. Some types of data that have a significant impact on agricultural exports of nitrogen are not tracked by NASS. The most notable are areas of land improved by subsurface tile drainage, timing of nitrogen fertilizer application, and methods of manure application. Another limitation is that data do not provide information about variations in rate of fertilizer application at the county level or finer.

Water Quality Monitoring, Analysis, and Interpretation

Independently, the U.S. Environmental Protection Agency (USEPA) has tracked the status of water quality in lakes and rivers since 1992 at two-year intervals through the National Water Quality Inventory (305b) process. These assessments are not based on statistical sampling strategies; rather, they are based on summaries of water quality monitoring or survey data when they exist. In 1992, siltation and nutrients were responsible for 45% and 37% of the miles of impaired reaches, respectively (USEPA, 1992). By 2000, the USEPA reported that 39% of assessed rivers and 46% of assessed lakes were impaired. Due to inconsistencies in the methods used to gather data, assessments from different years and across states cannot be reliably compared, so it is difficult to determine whether or not there are trends in the extent of impaired water bodies.

The USEPA has historically used water quality monitoring data to identify impaired water bodies. Various sampling techniques are used, ranging from observations taken by volunteers, to regular grab samples taken regardless of flow regime, to sophisticated automated sampling programs that are actuated by storm events. As such, aggregating results for meaningful interpretation is difficult when methods are not standardized and sample resolution is variable both temporally and spatially. Thus, the usefulness of water quality data for determining pollutant loads and water quality characteristics varies. Much of the sediment and phosphorus loads transported to rivers and lakes in the upper Midwest is delivered during high-intensity rainfall events that represent a small proportion of the collected water quality samples (Birr and Mulla, 2005). If these events are not sampled, then the pollutant loads cannot be estimated reliably. A much larger proportion of nitrate transport occurs during baseflow than for sediment and phosphorus transport. Thus, water quality monitoring programs for nitrate must involve samples collected during both storm events and baseflow.

A strength of existing water quality monitoring programs for determining effectiveness of BMPs is that they are often focused at the scale of 8-digit hydrologic unit code watersheds or larger. This allows for regional assessments of spatial variations in water quality for large watersheds. These regional patterns have been modeled using Spatially Referenced Regressions on Watershed Attributes (SPARROW) in an attempt to predict the relationships between water quality and factors such as land management practices and stream channel characteristics (Smith et al., 1997). SPARROW models are useful for identifying which 8-digit hydrologic unit watersheds are the largest sources of nitrogen and phosphorus export to the Mississippi River basin.

Another innovative use of long-term regional-scale water quality monitoring data is the statistical modeling work of McIsaac et al. (2001, 2002). They found a strong statistical correlation between annual nitrate flux to the Gulf of Mexico and factors such as net anthropogenic nitrogen inputs in the Mississippi River basin and annual river discharge. This statistical model was used to infer that the effects on water quality of reductions in N fertilizer on the landscape would not be completely realized until a lag time of nine years. The greatest impact of reductions in fertilizer use would occur within the first two to five years, with secondary impacts lagging by six to nine years. Recent work by Mulvaney et al. (2001) has suggested that organic N forms such as amino sugars may partially explain these lags, but more work is needed to better understand the long-term dynamics of organic N in Midwestern agricultural soils. Soils in the Upper Mississippi River basin have large amounts of organic matter, and therefore large pools of organic N and P. The cycling of N and P is greatly influenced by recent and long-term management effects, but improving management practices may lead to a slow improvement in water quality due to these relatively stable organic nutrient pools.

The large area of watersheds (typically several hundred thousand acres or more) monitored through existing federal and state programs can, however, also be considered a weakness when it comes to evaluating BMP implementation impacts on water quality. BMPs are typically implemented at very low density at this scale, making it difficult to quantify measurable impacts of BMPs on water quality. In addition, there may be considerable variations within watersheds in landscape, climatic, and soil factors that control the effectiveness of BMPs. Additional effort should be made to conduct detailed water quality monitoring studies in smaller watersheds (several thousand acres or less) with more homogeneous soil, landscape and climatic characteristics, or to use nested water quality sampling strategies to better separate out these effects.

Sophisticated statistical tools are needed to evaluate trends in water quality over time due to implementation of BMPs. Trends in water quality can arise from other causes as well, including long-term increases in precipitation, increases in the amount of land that is tile drained, changes in land use, expansion of urban developments, improved crop varieties or changes in crop rotation. Separating these effects from the effects of BMPs is difficult. Further complication is added at watershed scales, since watersheds typically involve implementation of multiple BMPs, rather than a single BMP in isolation from other BMPs. Thus, watersheds are, by their very nature, confounded and challenging to assess. Multiple analysis techniques are needed. Trend analysis, regression, simulation modeling and statistical analysis of variance (ANOVA) approaches all have specific strengths and weaknesses. For these reasons, the effectiveness of BMPs has traditionally been evaluated under more controlled smaller-scale conditions.

Evaluating BMPs on Small Research Plots

When new approaches are first developed to reduce nonpoint-source pollution, these potential BMPs are typically evaluated using research on small plots with statistically rigorous experimental designs that involve randomization and replication. An example of such research is small tile-drained plots with a continuous corn crop that receives a wide range of nitrogen application rates. Drainage water is collected from the plots, and the effluent is analyzed for nitrate-nitrogen. After harvest, grain yield and nitrogen losses from the plots are summarized and analyzed using standard statistical methods. Results from the experiment can be used to determine the optimum rate of nitrogen fertilizer that reduces nitrogen losses while maintaining crop productivity.

An experiment such as this is scientifically rigorous. It adequately defines the nitrogen BMP for the site and time period where the experiment was conducted. Yet, it leaves some questions unanswered. For example, the following questions are relevant for this and other similar BMPs:

- How widely must the BMPs from this experiment be implemented to make significant contributions to reducing the hypoxic zone area by 30%?
- How does the effectiveness of the BMPs vary in response to spatial and temporal variations in climate, landscape, soils, and proximity to surface waters?
- How many years do BMPs need to be installed before benefits are observed?
- What will be the N losses for a corn-soybean rotation?

Evaluating BMPs in Field-Scale Experiments

Farmers often question the applicability of plot-scale research on BMPs for implementation on their farms. They view their farms as differing from the experimental plots in area, diversity of soils and landscapes, and management practices. The availability of GIS, GPS, and computers has allowed researchers and farmers, particularly those with an interest in precision agriculture, to conduct experiments to evaluate the effectiveness of BMPS at the field scale. Most often, these experiments focus on evaluations of crop productivity rather than water quality impacts.

These experiments often involve use of commercial farm equipment to apply treatments, often using long strips across the landscape. For best results, these treatments should be randomized and replicated. Farmer-owned combines and implements equipped with yield monitoring and chemical application systems and global positioning systems (GPS) are typically used to collect yield and chemical application data used in assessing the effects of BMPs. Advanced statistical techniques are needed to evaluate the effectiveness of BMPs in these field-scale experiments. Some of the more promising tools include nearest-neighbor analysis, analysis of covariance, mixed-model forms of ANOVA, spatial autoregressive models, and special experimental designs.

Advantages of these experiments include implementation on many field sites, better representation of soil and climatic diversity, and greater farmer acceptance of results. Disadvantages include more factors that can confound the interpretation of treatment effects, including spatial variability of soil properties and precipitation, differences in planting dates and cultivars, farmer management errors, inaccurate harvest data, and uneven weed and pest infestations. In addition, it is often difficult to measure water quality impacts of BMPs at the farm scale due to difficulties in measuring runoff, erosion, and nutrient loss from large areas with numerous hydrological outlets.

Evaluating BMPs at the Regional Scale

During the 1990s, the USDA-ARS and USDA-CSREES provided funding for Management Systems Evaluation Areas (MSEA) and Agricultural Systems for Environmental Quality (ASEQ). The goal of the MSEA program was to develop and promote agricultural management systems that reduced the impact of farming on ground and surface water quality. MSEA sites (plot, field, and small watershed scales) were located in five states: Ohio, Missouri, Minnesota, Iowa, and Nebraska (Ward et al., 1994). Extensive evaluation of the water quality impacts of farming systems were conducted at these sites. The scope and timeline of the research were extended beginning in 1996 with ASEQ, which had research sites located in Missouri, Ohio, and Indiana.

Numerous BMPs were evaluated at the sites for their relative effect on water quality. Water quality modeling was used to predict effects at watershed and regional scales. Such analysis predicted that water quality would improve with reduced applications of phosphorus or nitrogen fertilizers and increases in the adoption of soil conservation practices. However, actual empirical evidence for improvements in watershed-scale water quality as a result of these projects was largely absent, presumably due to inadequate extent of BMP implementation and system time lags.

Evaluating BMPs at the Watershed Scale

The National Research Council reports that one of the primary needs of the TMDL program is information on the effectiveness of BMPs and the related processes of system recovery (USEPA, 2002). A 1998 report by the USEPA stated that BMP effectiveness research ranks second among the USEPA's priorities for science and tool development. TMDL plans require reasonable assurance that implemented BMPs will meet load reduction goals. Moreover, an understanding of the processes and time scales involved in the restoration process is also needed in order to verify water quality improvement (USEPA, 2002).

There have been few long-term evaluations of the effectiveness of BMPs at the watershed scale. To address this knowledge gap, the USDA-NRCS and USDA-CSREES have recently started the Conservation Effects Assessment Project (CEAP). CEAP has two components (USDA-NRCS, 2005). The first component is to use ongoing statistically based farm-scale data collected through the National Resources Inventory (NRI) to document trends in conservation practice adoption nationwide. The second component is to evaluate the effectiveness of BMP implementation in selected watersheds with a long record of water quality monitoring data (Mausbach and Dedrick, 2004). These studies are designed to address the effectiveness of BMPs for erosion control and nutrient management over a wide range of soil, landscape, climate, and land use characteristics. These studies will also be used to test the accuracy of computer model predictions (e.g., the Soil Water Assessment Tool, or SWAT) on the effectiveness of

BMPs. Finally, the studies will be used to evaluate the impacts of BMPs on wildlife populations and on soil and air quality.

The major strength of CEAP is the detailed study of water quality trends in relatively small watersheds with a long history of water quality monitoring data. The major weakness is the lack of detailed long-term information in many of these watersheds concerning BMP implementation. Another difficulty of conducting such watershed-scale studies is the difficulty of convincing a significant number of farmers within the watershed to simultaneously implement BMPs on their farms.

Published Research on Effectiveness of BMPs at the Watershed Scale

There are four common approaches to determining the impacts of BMPs on water quality at the watershed scale (Spooner et al., 1995). The first approach is studying trends in water quality over time without detailed knowledge of BMP implementation within the watershed. An example of this is a study conducted by Richards and Baker (2002) for four watersheds in Ohio. They studied log-transformed water quality data from 1975 to 1995 using analysis of covariance with time and seasonality as covariates. Significant decreases were observed in total phosphorus and total suspended solids, but not nitrate-N. Without detailed tracking of BMP implementation within the watersheds, there was no definitive way of identifying the cause of the water quality changes, although statistical measures suggested that the changes were due to improvements in nutrient management and conservation tillage.

A second approach is water quality monitoring upstream and downstream of the area where BMPs were implemented. Water quality downstream of BMPs can be compared with water quality upstream to determine if there have been any improvements. This approach is of limited value if the upstream monitoring station collects water from a very large area, since it will be difficult to detect small changes in water quality due to implementation of BMPs downstream. A third approach is multi-year monitoring of multiple watersheds where BMPs have been implemented. This approach is limited due to the variability in flow that typically occurs in space and time. It is difficult to separate the influences of flow variation due to climatic variability from the effects of BMPs. The most rigorous approach involves paired watershed comparisons. Paired watersheds have been used extensively in the field of forest management to study the effectiveness of BMPs. A paired watershed experiment involves two nearby watersheds with similar climate, landscape, soils, and management. BMPs are installed in one of the watersheds, and no changes are made in the other (control). Water quality monitoring should take place in both watersheds for at least one to three years before implementing BMPs in the treated watershed. Water quality monitoring should then continue for a minimum of another three to five years in both watersheds after implementation of BMPs.

Examples of Unpaired Watershed Assessments

Davie and Lant (1994) studied the impact of CRP implementation on sediment loads in two Illinois watersheds. They found that CRP enrollments on 15% and 27% of cropland decreased estimated erosion rates by 24% and 37%, respectively, but sediment loads at the mouths of the watersheds decreased by less than 1%. They attributed these small overall impacts to poor targeting of CRP to lands in close proximity to streams and to a time delay in sediment transport from the field edge to the mouth of the watershed. The estimated erosion reductions occurred only in the third year of their three-year study.

Schuler (1996) described the restoration of Lake Shaokatan in southwestern Minnesota. This lake was heavily impaired by excessive nitrogen and phosphorus levels, and had nuisance algal blooms and algal toxins that occasionally caused the death of cattle and dogs that drank from the lake. It was determined that a significant proportion of the nutrient load to the lakes was generated by three swine operations and one dairy farm. After corrective measures were taken on these operations in 1993, the lake water quality improved significantly. From 1994 to 1996, the average lake total phosphorus concentrations decreased from 270 ppb to less than 160 ppb. Noxious algal blooms and algal scums also disappeared.

Edwards et al. (1997) evaluated the effect of BMPs on two tributaries of the Lincoln Lake watershed located in northwest Arkansas draining 1800 and 800 ha, respectively. Monitoring was conducted over a period of approximately 2.5 years, with BMP implementation conducted simultaneously. By the end of the monitoring period, BMPs had been implemented on 39% of the available area in one of the watersheds and on 65% of the available land in the other watershed. Reductions ranging from 23% to 75% per year were observed in concentrations and mass transport of nitrate-nitrogen (NO_3-N), ammonia nitrogen (NH_3-N), total Kjeldahl nitrogen (TKN), and chemical oxygen demand (COD) based on trend analysis. Major BMPs implemented included nutrient management, pasture and hayland management, waste utilization, dead poultry composting, and waste storage structure construction (Edwards et al., 1997).

Garrison and Asplund (1993) studied the effect of reducing phosphorus loadings on lake water quality in a 1216 ha Wisconsin watershed. Phosphorus losses from animal waste storage facilities were reduced by 46% and from cropland runoff by 19%, but these improvements had a negligible impact on water quality of a lake at the river mouth. Total phosphorus levels in the lake increased from 29 ppb before implementation of pollution control measures to 44 ppb 15 years after implementation. Chlorophyll-a levels increased from 9 to 13 ppb over the same time period. The increased impairment of the lake after reductions in phosphorus losses was attributed to a failure to control cropland runoff adequately, which accounted for 76% of the phosphorus loading.

Inamdar et al. (2001) evaluated agronomic and structural BMPs on the 1463 ha mixed-use Nomini Creek watershed in Virginia. In the seven years during and following BMP implementation, average annual loads and flow-weighted concentrations of nitrogen were reduced by 26% and 41%, respectively. The largest reductions were observed for dissolved ammonium-N, soluble organic N, and particulate N. The authors did not observe statistically significant reductions in phosphorus loads and concentrations. Total phosphorus loads were reduced by 4% due to reductions in particulate P.

Graczyk et al. (2003) studied the effects of BMPs on two watersheds (14.0 and 27.2 km^2) in southern Wisconsin using monitoring data collected from 1984 to 1998. The post-BMP monitoring data were collected eight years after BMP implementation began. BMPs included animal waste management, streambank protection, and upland erosion and nutrient management strategies. Significant reductions in NH_3-N load during storm flows were observed in the larger watershed based on regression residuals. For the smaller watershed, significant decreases in both total phosphorus and NH_3-N storm loads were observed based on residuals from a regression analysis of TP and ammonia-N loads on total rainfall, antecedent precipitation, and serial date of storms for the pre- and post-BMP periods.

Examples of Paired Watershed Assessments

The basic premise of a paired watershed design is that there is a quantifiable, statistically significant relationship between paired water quality data for two watersheds. The water quality values do not need to be equal between the two watersheds, but rather the relationship must be consistent over time, except for the influence of BMP implementation in the treatment watershed (Clausen and Spooner, 1993).

The advantage of a paired watershed approach is that watershed differences and year-to-year climatic differences can be accounted for in the analysis. With a paired watershed approach, the study area is a collection of fields, and the watersheds do not need to be identical. Disadvantages of this approach include: minimal change in the control watershed is permitted, short calibration time may result in serially correlated data, and response to the treatment may be gradual over time (Clausen and Spooner, 1993).

Clausen et al. (1996) applied a paired watershed approach to two agricultural watersheds in west-central Vermont to evaluate tillage effects on runoff, sediment, and pesticide losses. Over a 30-month treatment period, reduced tillage decreased runoff by 64% and sediment losses by 99%. Bishop et al. (2005) also used a paired watershed approach to evaluate nutrient and sediment loading attributable to BMPs implemented on a 65 ha dairy farm watershed in New York. They found that manure management BMPs and rotational grazing reduced total phosphorus loads by 29% relative to the control watershed.

Gallichand et al. (1998) attributed 90% of the point-source pollution in the Belair River watershed near Quebec to leaking liquid manure tanks and manure piles. Improved manure storage facilities and septic tanks, and electric fences near streams were installed throughout a 531 ha experimental watershed to improve water quality. In addition, fertilizer applications were reduced, fall application of manure was reduced from 70% to 13%, and spring and summer applications were split. No improvements were made in an adjacent control watershed. Maximum concentrations of total phosphorus and dissolved phosphorus decreased significantly in the experimental watershed, but not in the control watershed, during two years of monitoring after improvement. Fecal bacteria counts were not measurably affected by the watershed improvements. In spite of the improvements, total phosphorus concentrations in the improved experimental watershed still exceeded critical levels (0.03 mg L^{-1}) for protection of aquatic life 94% of the time.

Udawatta et al. (2002) used field-scale paired watersheds to study the effects of grass and agroforestry contour buffer strips on runoff, sediment, and nutrient losses on highly erodible claypan soils of northern Missouri. Both watersheds employed conservation tillage, so runoff and yields were fairly low to begin with. After a seven-year calibration period, grass and agroforestry strips were initiated and found to reduce total phosphorus by 8% and 17% during the first three years. Only in the third year was total nitrogen reduced (between 24% and 37%) by the buffer strips. During the same period, buffer strip treatments only reduced water runoff by about 9%.

Jaynes et al. (2004) worked with eight producers in a 400 ha tile-drained subwatershed of the Walnut Creek watershed in Iowa to reduce nitrogen fertilizer applications through use of a late-spring nitrogen test (LSNT). Water quality data collect from this and an adjacent watershed since 1997 showed a 41% reduction in nitrate-N losses from the watershed where the LSNT approach was used relative to losses in the control watershed. Corn yields in the two watersheds were similar in three out of four years.

Birr and Mulla (2005) implemented conservation tillage on 70% of the moldboard plowed acreage for three years in an 1100 ha watershed in southern Minnesota. No changes in tillage were made in an adjacent watershed. Although these changes resulted in a 40% reduction in erosion for the treated fields, and an estimated 20% reduction in sediment load delivered to the mouth of the watershed, statistical comparisons of water quality monitoring data in the treated and control watersheds failed to show any improvements in water quality in the treated watershed, probably due to: (1) the effects of climatic variability, (2) the lag times for transport of pollutants from the field to the watershed scale, and (3) the need for more than three years of water quality monitoring data to identify trends.

Modeling

Knowledge of BMP effectiveness has been increasingly studied using process-based models at the watershed scale (Phillips et al., 1993; Hamlett and Epp, 1994; Keith et al., 2000; Mostaghimi et al., 1997; Osei et al., 2000; Walter et al., 2001). The use of models for watershed-scale assessments of BMPs is warranted due to the challenges associated with implementing these assessments in the field such as: the large range of management practices and physiographic conditions, confounding effects of implementing multiple BMPs at varying extents and locations within a watershed, time periods required to measure a response to BMPs, and the impact of BMPs applied under conditions differing from those that were tested (Walker, 1994; Sharpley et al., 2002).

Types of Questions Models Can Answer

Substantial advances have been made in using simulation models in the prediction of agricultural chemicals in the environment. These models help to estimate the time required for natural processes to remove chemicals already in the soil and groundwater, to predict the movement and persistence of chemicals in soil, and to predict the fate of agricultural chemicals to assist farmers in designing effective crop, soil, and chemical management strategies (Wagenet and Hutson, 1986). Models can aid in evaluating alternative rates and timing of chemical application, the use of alternative chemicals with different properties, and optimum management practices for soil, water, and chemicals. They have proved to be effective and efficient tools for water resource management decision support, and are increasingly being used to estimate the impacts of BMPs on TMDL goals (Dalzell et al., 2004; Gowda et al., 2007).

Models are useful for studying scenarios that cannot be investigated using actual experimentation. For example, models can be used to estimate the effectiveness of BMPs under various climate change scenarios. Models can also separate the impacts on water quality when multiple changes in management are made. For example, if a large dairy feedlot is established in a watershed, which previously had agricultural fields in a corn-soybean rotation, the model can be used to evaluate the water quality benefits from increases in the acreage of alfalfa versus the negative impacts of increased rates of manure application on cropland.

Models are also useful for estimating the best locations for implementation of BMPs within a watershed, and how much area these BMPs should cover to attain predetermined water quality improvements. The accuracy of models in making these predictions depends on the availability of accurate model input data. The most critical data are typically those related to topography (slope steepness, runoff curve number), the hydrologic properties of soil horizons across the landscape (hydraulic conductivity, moisture characteristics), and the variability in agricultural management practices for different fields (White and Chaubey, 2005).

Models are also useful for estimating the uncertainty in impacts on water quality of BMPs. Uncertainty can be estimated by varying critical model input parameters one at a time to determine their effect on predicted transport of pollutants. This is typically referred to as a sensitivity analysis. Practices that have a high certainty of improving water quality, despite uncertainty in model input parameters, are more likely to be effective than practices that have a high uncertainty.

Limitations of Modeling

Models are not designed to answer every question. They are designed to answer specific questions for widely varying conditions and with different levels of accuracy. When selecting a model for a particular application, it is critical that the selected model was designed and has sufficient accuracy to answer the question that the modeler needs to answer. For example, models such as SWAT, which employ daily time steps, are fairly accurate with respect to predicting average annual sediment or nutrient losses and comparing differences in alternative management practice scenarios, but they are much less accurate in predicting daily losses and concentrations that may be necessary to determine compliance with instantaneous water quality standards. They are particularly limited when the accuracy or availability of input data are limited. The accuracy of models that have been calibrated and validated can be quantified, but the accuracy of uncalibrated models cannot. In general, models should not be used if they do not accurately represent the processes and pathways for transport of pollutants within the field or watershed that is being studied. Models are often not useful under extreme conditions, including extremely intense storms, very steep slopes, or excessively high application rates of manure. There is an appropriate scale for each type of model. Using a model developed to estimate nitrate leaching at the field scale to estimate nitrate losses at the watershed scale is inappropriate and beyond the scope of what the model was designed for.

Models use a variety of approaches to represent the effects of BMPs on water quality. Some of these approaches are deterministic, others are statistical, and a few are based on empirical build-up and wash-off or export coefficients. Models that use empirical representations of BMP effects on water quality generally have lower predictive ability to examine alternative management scenarios than models that use statistical or deterministic representations.

Comparison of Models

A variety of models are available to evaluate the effectiveness of BMPs at reducing transport of sediment, phosphorus, and nitrates to surface waters. These include the HSPF, GLEAMS, DRAINMOD, SWAT, EPIC, RZWQM, and ADAPT models (see descriptions below). Each of these models has strengths and weaknesses. Each is designed to operate at a different scale. A comprehensive listing of these and many other models is available at http://eco.wiz.uni-kassel.de/ecobas.html. Borah and Bera (2003) compared several of the most common watershed-scale hydrologic nonpoint-source models. Parsons et al. (2004) compared and contrasted agricultural nonpoint-source water quality models. The reader is encouraged to consult these references for more detailed information about any of the models described below.

HSPF (Hydrological Simulation Program Fortran) is a watershed-scale model, and is not designed to operate at the field scale (Bicknell et al., 1997). It is a sophisticated hydrologic and water quality model that has the ability to simulate runoff and water quality from pervious and impervious land areas, as well as in-stream and reservoir processes. The main weakness of HSPF in agricultural settings is that it does not explicitly account for fertilizer application rates, different types of mechanical tillage operations, or tile drainage management systems. HSPF is also a lumped parameter model at the subwatershed scale and cannot represent the impacts of spatial variations in BMP placement at the subwatershed scale. HSPF time steps can be as small as desired, and it can thus represent instantaneous instream pollutant concentrations.

SWAT (Soil and Water Assessment Tool) is a watershed-scale agricultural water quality model linked to existing nationwide soil and climatic databases (Arnold and Fohrer, 2005). SWAT computes runoff based on the curve number approach, and handles channel and reservoir routing. It has routines for agricultural management practices pertaining to fertilizer, manure, tillage, and crop growth and uptake. It accounts for leaching, runoff, erosion, and drainage losses. Erosion rates are based on RUSLE. The main weakness of SWAT is inflexibility in defining hydrologic response units based on factors other than land use or soil map unit boundaries. This is particularly problematic in small watersheds. Like HSPF, SWAT is also a lumped parameter model and cannot represent spatial variability within subwatersheds. SWAT's daily timestep also precludes its use for estimating instream concentrations during storm events.

GLEAMS (Groundwater Loading Effects of Agricultural Management Systems) is a field-scale model operating at a daily time step that is designed to estimate pollutant losses at the edge of field and below the root zone (Leonard et al., 1987). Much of the code from this model was used as the basis for SWAT. GLEAMS has detailed algorithms for a diverse range of agricultural management operations. It is not designed to address spatial variability of soils, management, or precipitation within a field. It does not have explicit algorithms to simulate effects of tile drainage. The maximum depth simulated is limited to five soil horizons and 1.5 m.

DRAINMOD (Drainage Model) is a water table management model developed for poorly drained soils with parallel networks of subsurface drains or surface drainage ditches (Skaggs, 1982). It estimates water flow and nitrate losses to drains or ditches. DRAINMOD does not estimate losses of sediment or phosphorus, and is weak on surface runoff processes. DRAINMOD does not estimate impacts of nitrogen stress on crop growth.

ADAPT (Agricultural Drainage and Pesticide Transport) is a combination of GLEAMS and DRAINMOD (Ward et al., 1993). It has the capability to be run at the field or watershed scales, and simulates losses of sediment, phosphorus, and nitrates through surface and subsurface transport processes. ADAPT is limited by an inability to simulate effects of nitrogen fertilizer management on crop growth and subsequent impacts of crop growth on evapotranspiration and the water balance.

EPIC (Erosion Productivity Impact Calculator) was designed to simulate the impacts of agricultural management practices on erosion and crop productivity (Williams et al., 1984). It can estimate losses of sediment, phosphorus, and nitrates to surface and ground waters. EPIC is not designed to be operated at the watershed scale, nor does it have the ability to explicitly simulate impacts of tile drainage on water flow or water quality.

RZWQM (Root Zone Water Quality Model) is a process-based one-dimensional model that simulates transport of water, nitrates, and pesticides by runoff and leaching (Ahuja et al., 2000). It accounts for plant growth and uptake, including root growth. A myriad of agricultural management practices can be represented using RZWQM. RZWQM can simulate leaching losses to a depth of 30 m, and includes the effects of macropore flow. RZWQM requires numerous input parameters, and can be challenging to calibrate.

Phosphorus index models are increasingly being used by many states to estimate the risk of phosphorus transport to surface waters. The matrix (Lemunyon and Gilbert, 1993) and pathway (www.mnpi.umn.edu/#summary) versions of the phosphorus index exemplify two typical approaches used in estimating phosphorus loss risks. Each of these has strengths and weaknesses, but neither is able to account for the impacts of climatic variations or detailed mechanistic considerations. In general, they are not designed to estimate the actual losses of phosphorus; rather, they give a risk estimate relative to a set of baseline conditions. These models are typically applied at the field scale, although a few studies have examined the phosphorus index models at the regional scale (Birr and Mulla, 2001).

Model Calibration and Validation

The basic protocol for hydrologic modeling, regardless of the scale of the problem, has been summarized by Anderson and Woessner (1992). The essential steps include defining the purpose of the study and specifying the questions that need to be answered, selecting an appropriate model for the specified questions, using existing field data or collecting new field data to calibrate the model, validating the model using data not used in calibration, using the model to predict a future response at the experimental site or for the surrounding region, presenting and interpreting the results of model predictions, and conducting a post-audit evaluation of the model.

Once a model is selected, one of the first steps in using it is to determine the value of input parameters needed by the model. Deterministic distributed parameter models will require thousands of input parameters, quasi-distributed models require the estimation of hundreds of parameters, and lumped or empirical models typically require far fewer input parameters. Selecting input parameters is often termed parameter estimation or parameterization. There are five major approaches for selecting input parameters (Addiscott et al., 1995). These are by direct measurement, by pedotransfer function, by direct fitting with model expression, by indirect fitting with whole model, and by fitting model to the data (calibration). The approach actually taken depends to some extent on the scale at which the model is to be applied, and the availability of measured data and/or pedotransfer function models for the region studied.

Data needs for evaluating nitrogen management with fate and transport models must typically address multiple pathways. As the scale of study becomes coarser, it becomes more difficult to obtain accurate information concerning nitrogen transport pathways and processes. For example, what is the spatial and temporal variability in denitrification at the scale of a major watershed? Does it matter if we use a spatially or temporally average denitrification value across the entire watershed? Similar uncertainty exists in estimating other model inputs, including soil hydraulic conductivity and stream travel times.

In addition to the processes and pathways, it is essential to have sufficiently accurate information concerning the inputs of nitrogen from fertilizer, manure, atmospheric deposition, and fixation. For plots, hillslopes, and fields, these inputs can be reasonably controlled through management. At the scale of watersheds and large regions, we must increasingly rely on statistical survey information on fertilizer sales, number and species of farm animals, average rates of manure production and nutrient content, and types of confinement, storage, or land application methods. A major data gap often exists for farm nutrient management practices. What is the spatial variation in rates of nitrogen applied from fertilizer and manure across the watershed? Does it matter if we use the average rate for modeling watershed-scale losses of nitrate? Other useful information includes land use, dates of crop planting and harvest, and residue cover. At the watershed scale, there is often considerable uncertainty in input data about these management practices.

Information concerning spatial and temporal variations in precipitation becomes important at the watershed scale. During a particular storm, one area of the watershed may experience more intense precipitation than the rest of the watershed. If this is critical for the questions being asked, then an adequate network of precipitation gauges is important.

Results of modeling at the scale of plots, hillslopes or fields are often not accurate when extrapolated to the scale of minor or major watersheds. New processes occur at the watershed scale that are not simulated at the plot or hillslope scales, including ground water baseflow, nutrient transformations in ditches, streams, lakes, and wetlands, uptake by grass and trees, and streambank erosion. The accuracy of the model depends on its ability to account for these processes.

This process of selecting input parameters to optimize the fit between model "predictions" and observed data is often referred to as model calibration. It is typically followed by a second independent step termed validation. Validation differs from calibration in two essential ways. First, model parameters are not adjusted during validation (Addiscott et al., 1995). Second, the performance of the model is evaluated using a different data set, preferably independent, from the training subset used in calibration. This data set may be a subset of the experimental measurements used for calibration of the model, or it may be data from the same type of experiment conducted at a different location or time. The accuracy of the model is evaluated against the experimental data subset during the validation phase using statistical and graphical techniques, and prediction errors can be quantified (Loague and Green, 1991). Rigorous calibration and validation of a watershed-scale model typically requires the availability of four to ten years of water quality monitoring data, along with the associated climatic, soil, and management input data. Datasets of this nature are scarce.

As shown in the previous sections, modeling results are highly dependent on the model, adequacy of model input data, and assumptions made during the model application process. Modeling standards and protocols are needed to help reduce this variability. Automation of modeling processes would reduce this variability, and help improve the rigor of decision support systems used for conservation planning.

Targeting BMPs to Critical Areas

Export of pollutants, such as sediment and sediment-bound nutrients such as phosphorus, does not occur with spatial or temporal uniformity within an agricultural watershed. Critical source areas exist within these watersheds that are hydrologically active during storm events and transport a majority of the pollutant load observed at the watershed outlet (Walter et al., 2000; Gburek et al., 2002). Pionke et al. (2000) found that 98% of the algal-available phosphorus measured in a 25.7 ha agricultural watershed in Pennsylvania came from 6% of the watershed area. It makes sense to implement BMPs in the regions of a watershed that are most likely to be the greatest source of water quality impairment. These areas generally have direct transport pathways to surface water bodies, and may have soil or landscape characteristics that make them vulnerable to generating nonpoint-source pollution. The optimization of BMP type and placement (location) using models has rarely been done due to the inability of most models to simulate spatial placement of BMPs and the computational requirements and complexity of models that can. This is an area of significant opportunity.

The most cost-effective reduction of nonpoint-source pollution loads at the watershed outlet in agricultural settings is dependent on the implementation of effective BMPs in critical source areas of nonpoint-source pollution that transport a majority of the pollutant load to the watershed outlet (Maas et al., 1985; Ice, 2004). Defining critical source areas in agricultural watersheds is a challenge due to the hydrologic complexity and natural variability that occurs across the landscape. However, studies show that topographic indices can be used to assist water resource managers in targeting areas where the implementation of BMPs would be most effective (Gowda et al., 2003; Moore and Nieber, 1989; Tomer et al., 2003).

Topographic indices utilize individual and combinations of topographic attributes to describe complex hydrological processes in the landscape using simplified estimates of the spatial distribution of hydrologic variables in the landscape. The index approach sacrifices physical sophistication to allow simple calculations using key factors to develop estimates of soil moisture patterns in the landscape. The advantage of using a terrain-based index approach for identifying critical source areas of nonpoint-source pollution at a watershed scale is that the input requirements are consistent with the level of data available to water resource managers and is appropriate for the precision with which many management questions need to be and can be answered (Barling et al., 1994).

The accuracy of terrain indices is dependent on several factors including: (1) the sampling location and density of elevation data, as well as the techniques used to collect the data; (2) the horizontal resolution and vertical precision used to represent the elevation data; (3) the algorithms used to calculate the terrain attributes; and (4) the topography of the landscape being represented (Theobald, 1989; Chang and Tsai, 1991; Florinksy, 1998). The interpretation of terrain indices must account for each of these factors such that the application of the data is appropriate given the limitations each of these factors presents for the data.

Some of the most useful terrain indices for targeting BMPs to critical areas include slope steepness, compound terrain index (Moore et al., 1991; Gallant and Wilson, 2000), and stream power index (Moore et al., 1993). To be effective, each of these must be considered in relation to the potential for transport to a nearby stream or lake. This potential is largely based on the direction of flow paths across the landscape and on the proximity of a given area to a stream or lake. To date, there have been few attempts to use quantitative techniques in terrain analysis in conjunction with simulation models in order to estimate the impacts on water quality of BMPs that are targeted to critical areas on the landscape (Lyon et al., 2004).

The term “precision conservation” has recently been coined (Berry et al., 2003) to reflect the overall process of targeting conservation practices to the most vulnerable portions of the landscape. Precision conservation ties efforts across scales (zones within field to between fields to watershed and basin management) and is a key tool in achieving conservation goals. Precision conservation involves the application of global positioning systems (GPS), remote sensing (RS), terrain analysis, and geographic information systems (GIS) in conjunction with existing spatial databases to examine spatial relationships using modeling, spatial data mining, and map analysis. It is an extension of the ideas of precision agriculture, which use knowledge of spatial and temporal variability to tailor management (Mulla, 1991). The goals of optimizing management using precision information should simultaneously consider both profitability and conservation. However, past studies in this area have either focused on the one or the other, but not both. To achieve sustainable food production systems, precision agriculture technologies and practices need to be integrated into conservation planning and assessment, in order to deal with the complexity of spatial heterogeneity of farmlands (Berry et al., 2003).

Impacts of Complexity, Non-linearity, and Feedback Loops on BMPs

A given BMP does not have the same effectiveness at improving water quality across all soil types, landscape positions, climatic regions, or management systems. A sediment BMP differs in effectiveness depending on slope steepness, distance from a surface water body, and frequency of intense storms. A nitrogen BMP varies in effectiveness in response to factors such as soil organic matter content, amount and timing of fertilizer applied before the BMP was implemented, manure management practices, and extent of subsurface tile drainage. To complicate matters further, the effectiveness of a nitrogen BMP may depend on what other types of management practices are in place. The effect on water quality of reducing nitrogen fertilizer application rate may depend on the amount of crop residue for erosion control, and on the type of tillage practiced. Greater amounts of residue may tie up more nitrogen through immobilization, thereby reducing leaching losses. The reduced tillage practices associated with increased crop residue coverage may, however, lead to greater infiltration. Greater infiltration may increase the risk of nitrate leaching. So, reduced tillage systems may either increase or decrease the effectiveness of nitrogen BMPs. These types of interactions involve both complexity and feedback loops.

Another type of interaction is non-linearity. When BMPs are implemented, their effect on water quality may depend on other factors. This type of behavior is often dependent on thresholds or critical values. For instance, decreasing phosphorus fertilizer application rates may have little impact on water quality if soil-test phosphorus levels are excessive, yet the same decreases may have an important impact if implemented on another soil with moderate soil-test phosphorus levels.

Factors that Offset the Effectiveness of BMPs

Benefits of implementing BMPs may be offset by several factors over time. Greater annual precipitation has been observed in the upper Midwestern region since the 1960s. This tends to increase the erosivity of rainfall, leading to greater erosion without any changes in management. It also tends to increase the fraction of water drained by subsurface tiles, leading to greater nitrate losses, all other factors being constant. Many BMPs lose their effectiveness over time (Brackmort et al., 2004) as a result of degradation, damage, neglect, or removal. Crop residue cover declines due to biological and physical degradation. Grassed waterways and riparian filter strips lose effectiveness as they become damaged by sediment deposition and concentrated flow. Terraces can be damaged by large storms. These effects are typically not considered when evaluating the long-term effectiveness of BMPs.

The effectiveness of BMPs for nitrogen leaching can also be offset by increasing amounts of land that are artificially drained, and by increases in the fraction of land in a continuous corn rotation as opposed to a corn-soybean rotation. Thus, the level of implementation of BMPs that is sufficient for water quality improvements will change depending on trends in climate, land use, and agricultural management systems.

Another example of offsetting factors can be given for phosphorus losses. If phosphorus losses from rainfall runoff are controlled by reduced tillage, then the rates of phosphorus loss during snowmelt runoff may increase due to greater trapping of snow and solubilization of phosphorus from crop residues. Finally, if rates of erosion decrease due to the implementation of BMPs, but there are no corresponding decreases in total volume of runoff, there may not be any decreases in sediment load at the mouth of the watershed because of increased rates of streambank erosion (Sekely et al., 2002).

Other Issues

Environmental management involves reducing the impact of multiple pollutants on the soil, water, air in both terrestrial and aquatic habitats. Marine environments are more sensitive to nitrogen enrichment, while freshwater environments are more sensitive to phosphorus enrichment. BMPs that reduce phosphorus losses to surface waters through reduced runoff may increase nitrate leaching losses. BMPs that reduce nitrate leaching losses through increased denitrification may increase losses of nitrous oxide to the atmosphere. Clearly, there must be clear directives about which pollutants are most important and what part of the environment it is most desirable to improve.

Government support for university Extension Service activities is in serious decline. The Extension Service has traditionally played an important role in conducting farm demonstrations that help to evaluate the effectiveness of BMPs at the field scale. New paradigms are needed for field testing of BMPs. These new paradigms could include on-farm trials with collections of farmers (grower learning groups). Farmers increasingly have the ability to establish experiments across their fields using GPS and yield monitors. The data from these experiments could be sent to researchers in industry or at universities for statistical analysis. Industry may have to play a larger role than in the past with regards to testing and promoting BMPs, and this includes BMPs for nitrogen, phosphorus, and sediment.

Government support for evaluating effectiveness of BMPs is critically needed, especially from the USDA and USEPA, Currently, most nonpoint-source monitoring funded by the USEPA is for watershed-scale assessment projects that do not include support for research to measure the effectiveness of individual BMPs. In the interest of improving the efficiency with which BMPs are implemented to improve water quality, the USDA and USEPA should consider earmarking a proportion of their funding to evaluating the effectiveness of BMPs.

Conclusions

Water quality impairments arising from sediment, phosphorus, and nitrogen are widespread throughout the upper Midwestern region. Hypoxia in the Gulf of Mexico, arising from excess nutrients transported down the Mississippi River, is a serious problem. There is increasing public pressure to improve water quality through implementation of BMPs on agricultural land in the upper Midwestern region. There is also increasing pressure to document water quality benefits of federal and state programs that pay farmers to implement BMPs.

A variety of methods are in place to document the implementation of BMPs, including the USDA National Resources Inventory (NRI) survey and farm statistical data collected by the USDA National Agricultural Statistics Service (NASS). The NRI has documented a 30% reduction in soil erosion on agricultural lands since 1982 due to the implementation of conservation tillage methods and the Conservation Reserve Program. Independent of these efforts, the USEPA tracks the status of water quality through the National Water Quality Inventory (NWQI). Due to lack of consistency in the reporting methods on which NWQI is based, it is difficult to relate the USDA tracking of BMPs with trends in water quality. More appropriate data, sophisticated statistical tools, and computer models are needed to quantitatively separate the effects on water quality of implementing specific BMPs from other influential factors such as a wetter climate, increases in the proportion of land that is tile drained, or reductions in the amount of pasture. The USDA is currently undertaking a new effort, the Conservation Evaluation Assessment Project (CEAP), to directly study the impacts of implementing BMPs on water quality in selected watersheds across the nation. However, even this effort devotes less than 1% of program funding to measuring improvements or changes in water quality.

The effectiveness of new and existing BMPs can be evaluated at a variety of scales using a variety of techniques. Traditionally, BMPs are first evaluated at the scale of small research plots. However, skepticism about the relevance of this research at coarser scales has led to increasing use of on-farm research. Research at the farm scale is often more focused on documenting the effects of BMPs on crop productivity than the effects on water quality. A few scientists have studied the effects of implementing BMPs on water quality at the small watershed scale. Results from some of these studies show that water quality is improved, while the remaining studies show no changes or a worsening in water quality. The studies that failed to show improvements in water quality often attributed the failure to an insufficient water quality monitoring record, a failure to implement BMPs that correct the most important sources of pollution, or a failure to implement BMPs in the most critical areas of the watershed. More emphasis is needed on long-term watershed-scale projects to evaluate impacts of BMPs on water quality, especially projects that involve paired watersheds. In addition, more focus is needed to evaluate the effectiveness of BMPs targeted to portions of the landscape that contribute most to water quality degradation.

Computer modeling is widely used to evaluate the impact of BMPs on water quality at a variety of scales. The accuracy of model results depends on selecting the right model to answer the desired questions, the ability of the model to simulate the desired pollutants and pollutant transport mechanisms at the appropriate spatial and temporal scales, the quality of the input data used to parameterize the model, and the availability of long-term hydrologic and water quality data to calibrate and validate the model. Model accuracy is typically better at plot or field scales, where input data are more reliable, than at watershed scales. Models can be used to assess the optimum rate of fertilizer or the impacts of fertilizer quantity and timing, crop rotations, and conservation tillage on water quality. More importantly, models have the potential to identify the portions of fields or watershed that are most critical for control of nonpoint-source pollution, as well as estimate the area that must be treated with a particular BMP or combination of BMPs in order to attain a desired level of improvement in water quality.

Models can also be used to evaluate impacts of BMPs on water quality under scenarios that would be difficult, if not impossible, to study experimentally. These "what if" scenarios include impacts of BMPs under conditions of changing land use and climate, or the effectiveness of BMPs under a wide range of soil and landscape characteristics. Caution must be used to avoid applying models to conditions for which they were not intended and are thus inappropriate, including extreme storm events, spatial scales for which the model was not intended, or watersheds in which inadequate or inaccurate input data are available for calibration and validation.

Summary

Water quality impairments are widespread throughout the Upper Mississippi River basin due in large part to agricultural production. Many agencies have worked with landowners to implement various agricultural best management practices (BMPs) to reduce nutrient and sediment losses to streams and rivers. However, it has been difficult to document the effectiveness of these practices at field and watershed scales. This is due to a number of factors, including the following:

Variability in weather, runoff and drainage lead to highly variable nutrient and sediment exports from one day or month to another and from one year to another. Without long-term data, it is difficult to know how much of a change in nutrient and sediment export has resulted from a change in management. Long-term water quality data sets of sufficient monitoring intensity are generally not available, and short-term (1 to 5 years) data sets can give false impressions of the response. Finally, long-term, baseline, monitoring data are needed before the agricultural practices are altered, given the variability typically found, and these data are seldom available.

There have been few scientifically rigorous studies of BMP effectiveness at the scale of small watersheds or larger. More long-term paired watershed studies, the most rigorous experimental design at the watershed scale, are needed in order to compare water quality in watersheds where BMPs are widely implemented to water quality in nearby watersheds where no BMPs are implemented.

Responses to changes in management require long lag times. Because of large soil pools of N and P, response to implementation of BMPs can take years to decades in some cases. In addition, stream and river responses may be obscured by previous accumulation and transport of in-stream sediments and nutrients that mask reduced export from fields.

Most management practices achieve sparse, non-targeted implementation at the watershed scale. Most conservation programs at the watershed scale only involve a small percentage of the land area and often do not target the most critical areas. Many studies have shown that a majority of the sediment or phosphorus that enters surface waters is generated by a small proportion of poorly managed land that is in close proximity to surface waters. Better tools are needed to identify these critical areas and improve them with appropriate BMPs.

Modeling limitations, including uncertainty in many parameters (e.g., soil hydraulic properties, denitrification, mineralization rates, and biological N_2 fixation), incomplete representations of field and watershed processes, and limited data for validation, can make projections uncertain. Because of limited long-term data sets, modeling is often used to project responses to management, but there are many difficulties in this approach.

References

Addiscott T. M., J. Smith, and N. Bradbury. 1995. Critical evaluation of models and their parameters. J. Environ. Qual. 24: 803-807.

Ahuja, L. R., K. W. Rojas, J. D. Hanson, M. J. Shaffer, and L. Ma. 2000. The Root Zone Water Quality Model. Highlands Ranch, Colo.: Water Resources Publications.

Anderson, M. P., and W. W. Woessner. 1992. Applied Groundwater Modeling: Simulation of Flow and Advective Transport. New York, N.Y.: Academic Press.

Arnold, J. G., and N. Fohrer. 2005. SWAT 2000: Current capabilities and research opportunities in applied watershed modeling. Hydrological Processes 19(3): 563-572.

Barling, R. D., I. D. Moore, and R. B. Grayson. 1994. A quasi-dynamic wetness index for characterizing the spatial distribution of zones of surface saturation and soil water content. Water Resour. Res. 30: 1029-1044.

Barr Engineering. 2004. A detailed assessment of phosphorus sources to Minnesota watersheds. St. Paul, Minn.: Minnesota Pollution Control Agency. Available at: www.pca.state.mn.us /hot/legislature/reports/phosphorus-report.html.

Berry, J. K., J. A. Delgado, R. Khosla, and F. J. Pierce. 2003. Precision conservation for environmental sustainability. J. Soil Water Conserv. 58: 332-339.

Bicknell, B. R., J. C. Imhoff, J. L. Kittle Jr., A. S. Donigian Jr., and R. C. Johanson. 1997. Hydrological Simulation Program - Fortran, User's manual for version 11. EPA/600/R-97/080. Athens, Ga.: U.S. EPA, National Exposure Research Laboratory.

Birr, A. S., and D. J. Mulla. 2001. Evaluation of phosphorus site index in watersheds at the regional scale. J. Environ. Quality 30: 2018-2025.

Birr, A. S., and D. J. Mulla. 2005. Paired watershed studies for nutrient reductions in the Minnesota River basin. Available at: www.usawaterquality.org/integrated/MNRiver. html.

Bishop, P. L., W. D. Hively, J. R. Stedinger, M. R. Rafferty, J. L. Lojpersberger, and J. A. Bloomfield. 2005. Multivariate analysis of paired watershed data to evaluate agricultural best management practice effects on stream water phosphorus. J. Environ. Qual. 34: 1087-1101.

Borah, D. K., and M. Bera. 2003. Watershed-scale hydrologic and nonpoint-source pollution models: Review of mathematical bases. Trans. ASAE 46: 1553-1566.

Bracmort, K. S., B. A. Engel, and J. R. Frankenberger. 2004. Evaluation of structural best management practices 20 years after installation: Black Creek Watershed, Indiana. J. Soil and Water Conserv. 59: 191-196.

BWSR. 2005. eLINK. St. Paul, Minn.: Board on Water and Soil Resources. Available at: www.bwsr.state.mn.us/outreach/eLINK/index.html.

Chang, K. T., and B. W. Tsai. 1991. The effect of DEM resolution on slope and aspect mapping. Cart. Geog. Info. Sys. 18: 69-77.

Clausen, J. C., and J. Spooner. 1993. Paired watershed study design. EPA 841-F-93-009. Washington, D.C.: U.S. EPA, Office of Water.

Clausen, J. C., W. E. Jokela, F. I. Potter III, and J. W. Williams. 1996. Paired watershed comparisons of tillage effects on runoff, sediment, and pesticide losses. J. Environ. Qual. 25: 1000-1007.

Dalzell, B. J., P. H. Gowda, and D. J. Mulla. 2004. Modeling sediment and phosphorus losses in an agricultural watershed to meet TMDLs. J. American Water Resources Assoc. 40: 533-543.

Davie, D. K., and C. L. Lant. 1994. The effect of CRP enrollment on sediment in two Illinois streams. J. Soil Water Conserv. 49: 407-412.

Edwards, D. R., T. C. Daniel, H. D. Scott, P. A. Moore Jr., J. F. Murdoch, and P. F. Vendrell. 1997. Effect of BMP implementation on storm flow quality of two northwestern Arkansas streams. Trans. ASAE 40: 1311-1319.

Florinksy, I. V. 1998. Accuracy of local topographic variables derived from digital elevation models. Intl. J. Geog. Info. Sci. 12: 47-61.

Gallant, J. C., and J. P. Wilson. 2000. Primary topographic attributes. In Terrain Analysis: Principles and Applications, 51-85. J. P. Wilson and J. C. Gallant, eds. New York, N.Y.: John Wiley and Sons.

Gallichand, J., E. Aubin, P. Baril, and G. Debailleul. 1998. Water quality improvement at the watershed scale in an animal production area. Canadian Agric. Eng. 40: 67-77.

Garrison, P. J., and T. R. Asplund. 1993. Long-term (15 years) results of NPS controls in an agricultural watershed upon a receiving lake's water quality. Water Sci. Tech. 28: 441-449.

Gburek, W. J., C. C. Drungil, M. S. Srinivasan, B. A. Needelman and D. E. Woodward. 2002. Variable source area controls on phosphorus transport: Bridging the gap between research and design. J. Soil Water Conserv. 57: 534-543.

Goolsby, D. A., W. A. Battaglin, G. B. Lawrence, R. S. Artz, B. T. Aulenbach, R. P. Hooper, D. R. Keeney and G. T. Stensland. 1999. Topic 3 Report: Flux and sources of nutrients in the Mississippi-Atchafalaya River basins. Decision Analysis Series No. 17. Silver Spring, Md.: NOAA Coastal Ocean Program.

Gowda, P. H., D. J. Mulla and B. J. Dalzell. 2003. Examining the targeting of conservation tillage practices to steep vs. flat landscapes in the Minnesota River basin. J. Soil Water Conserv. 58: 53-57.

Gowda, P. H., B. J. Dalzell, and D. J. Mulla. 2007. Model-based nitrate TMDLs for two agricultural watersheds in southern Minnesota. J. American Water Resources Assoc. 43: 256-263.

Graczyk, D. J., J. F. Walter, J. A. Horwatich and R. T. Bannerman. 2003. Effects of best management practices in the Black Earth Creek priority watershed, Wisconsin, 1984-98. USGS Water Res. Investigations Rept. 03-4163. Washington, D.C.: U.S. Geological Survey.

Hamlett, J. M., and D. J. Epp. 1994. Water quality impacts of conservation and nutrient management practices in Pennsylvania. J. Soil Water Conserv. 49: 59-66.

Ice, G. 2004. History of innovative Best Management Practice development and its role in addressing water quality limited waterbodies. J. Environ. Eng. 130: 684-689.

Inamdar, S. P., S. Mostaghimi, P. W. McClellan, and K. M. Brannan. 2001. BMP impacts on sediment and nutrient yields from an agricultural watershed in the coastal plain region. Trans. ASAE 44: 1191-1200.

Jaynes, D. B., D. L. Dinnes, D. W. Meek, D. L. Karlen, C. A. Cambardella, and T. S. Colvin. 2004. Using the late spring nitrate test to reduce nitrate loss within a watershed. J. Environ. Qual. 33: 669-677.

Keith, G., S. Norvell, R. Jones, C. Maquire, E. Osei, A. Saleh, P. Gassman, and J. Rodecap. 2000. Livestock and the environment: CEEOT-LP modeling for the Upper Maquoketa River watershed, Iowa. Stephenville, Texas: Texas Institute for Applied Environmental Research.

Lemunyon, J. L., and R. G. Gilbert. 1993. The concept and need for a phosphorus assessment tool. J. Prod. Agric. 6: 483-486.

Leonard, R. A., W. G. Knisel, and D. A. Still. 1987. GLEAMS: Groundwater loading effects of agricultural management systems. Trans. ASAE 30(5): 1403-1418.

Loague, K., and R. E. Green. 1991. Statistical and graphical methods for evaluating solute transport models: Overview and application. J. Contaminant Hydrol. 7: 51-73.

Lyon, S. W., M. T. Walter, P. Gerard-Marchant, and T. S. Steenhuis. 2004. Using a topographic index to distribute variable source area runoff predicted with the SCS curve number equation. Hydrol. Proc. 18: 2757-2771.

Maas, R. P., M. D. Smolen, and S. A. Dressing. 1985. Selecting critical areas for nonpoint-source pollution control. J. Soil Water Conserv. 40: 68-71.

Mausbach, M. J., and A. R. Dedrick. 2004. The length we go: Measuring environmental benefits of conservation practices. J. Soil Water Conserv. 59: 96-103. Available at: ftp://ftp-fc.sc.egov.usda.gov/NHQ/nri/web/swsc-CEAP-Reprint.pdf.

McIsaac, G. F., M. B. David, G. Z. Gertner, and D. A. Goolsby. 2001. Nitrate flux to the Gulf of Mexico. Nature 414: 166-167.

McIsaac, G. F., M. B. David, G. Z. Gertner, and D. A. Goolsby. 2002. Relating net nitrogen input in the Mississippi River basin to nitrate flux in the lower Mississippi River: A comparison of approaches. J. Environ. Qual. 31: 1610-1622.

Mitsch, W. J., J. W. Day Jr., J. W. Gilliam, P. M. Groffman, D. L. Hey, G. W. Randall, and N. Wang. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River basin: Strategies to counter a persistent ecological problem. BioScience 51: 373-388.

Moore, I. D., and J. L. Nieber. 1989. Landscape assessment of soil erosion and nonpoint-source pollution. J. Minnesota Acad. Sci. 55: 18-24.

Moore, I. D., R. B. Grayson, and A. R. Ladson. 1991. Digital terrain modeling: A review of hydrological, geomorphological, and biological applications. Hydrol. Processes. 5: 3-30.

Moore, I. D., P. E. Gessler, G. A. Nielsen, and G. A. Peterson. 1993. Soil attribute prediction using terrain analysis. SSSA J. 57: 443-452.

Mostaghimi, S., S. W. Park, R. A. Cooke, and S. Y. Wang. 1997. Assessment of management alternatives on a small agricultural watershed. Water Res. 31: 1867-1876.

Mulla, D. J. 1991. Using geostatistics and GIS to manage spatial patterns in soil fertility. In Automated Agriculture for the 21st Century, 336-345. G. Kranzler, ed. St. Joseph, Mich.: ASAE.

Mulvaney, R. L., S. A. Khan, R. G. Hoeft, and H. M. Brown. 2001. A soil organic nitrogen fraction that reduces the need for nitrogen fertilization. SSSA J. 65: 1164-1172.

Osei, E., P. W. Gassman, R. D. Jones, S. J. Pratt, L. L. M. Hauck, L. J. Beran, W. D. Rosenthal, and J. R. Williams. 2000. Economic and environmental impacts of alternative practices on dairy farms in an agricultural watershed. J. Soil Water Conserv. 55: 466-472.

Parsons, J. E., D. L. Thomas, and R. L. Huffman. 2004. Agricultural nonpoint-source water quality models: Their use and application. Southern Coop. Series Bulletin No. 398. Raleigh, N.C.: North Carolina State University. Available at: www3.bae.ncsu.edu/Regional-Bulletins/Modeling-Bulletin/modeling-bulletin.html.

Phillips, D. L., P. D. Hardin, V. W. Benson and J. V. Baglio. 1993. Nonpoint-source pollution impacts of alternative agricultural management practices in Illinois: A simulation study. J. Soil Water Conserv. 48: 499-456.

Pionke, H. B., W. J. Gburek, and A. N. Sharpley. 2000. Critical source area controls on water quality in an agricultural watershed located in the Chesapeake basin. Ecol. Eng. 14: 325-335.

Richards, R. P., and D. B. Baker. 2002. Trends in water quality in LEASEQ rivers and streams (northernwestern Ohio), 1975-1995. J. Environ. Qual. 31: 90-96.

Schuler, D. J. 1996. Lake Shaokatan restoration project: Final report. St. Paul, Minn.: Minnesota Pollution Control Agency, Yellow Medicine River Watershed District.

Sekely, A. C., D. J. Mulla, and D. W. Bauer. 2002. Streambank slumping and its contribution to the phosphorus and suspended sediment loads of the Blue Earth River, Minnesota. J. Soil Water Conserv. 57(5): 243-250.

Sharpley, A. N., P. J. A. Kleinman, R. W. McDowell, M. Gitau, and R. B. Bryant. 2002. Modeling phosphorus transport in agricultural watersheds: Processes and possibilities. J. Soil Water Conserv. 57: 425-439.

Skaggs, R. W. 1982. Field evaluation of a water management simulation model. Trans. ASAE 25(3): 666-674.

Smith, R. A., G. E. Schwarz, and R. B. Alexander. 1997. Regional interpretation of water-quality monitoring data. Water Resources Research 33: 2781-2798.

Spooner, J., D. E. Line, S. W. Coffey, D. L. Osmond, and J. A. Gale. 1995. Linking water quality trends with land treatment trends. Available at: www.bae.ncsu.edu/programs/extension/wqg/brochures/ten.html.

Theobald, D. M. 1989. Accuracy and bias issues in surface representation. In Accuracy of Spatial Databases, 99-106. M. Goodchild and S. Gopol, eds. Bristol, Pa.: Taylor and Francis.

Tomer, M. D., D. E. James, and T. M. Isenhart. 2003. Optimizing the placement of riparian practices in a watershed using terrain analysis. J. Soil Water Conserv. 58: 198-206.

Udawatta, R. P., J. J. Krstansky, G. S. Henderson, and H. E. Garrett. 2002. Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. J. Environ. Qual. 31: 1214-1225.

USDA-NRCS. 1997. National Resources Inventory: Highlights. Available at: www.nrcs.usda.gov/TECHNICAL/land/pubs/97highlights.html.

USDA-NRCS. 2005. Conservation Effects Assessment Project. Available at: www.nrcs.usda.gov/technical/nri/ceap.

USEPA. 1980. An approach to water resources evaluation on non-point silvicultural sources. EPA-600/8-80-012. Athens, Ga.: U.S. EPA, Environmental Research Laboratory.

USEPA. 1992. The quality of our nation's water. Available at: www.epa.gov/305b/92report/92summ.pdf.

USEPA. 2002. The twenty needs report: How research can improve the TMDL program. EPA 841-B-02-002. Washington, D.C.: U.S. EPA, Office of Water.

USEPA. 2007. Total maximum daily loads: Causes of impairment. Available at: http://oaspub.epa.gov/waters/national_rept.control.

Wagenet, R. J., and J. L. Hutson. 1986. Predicting the fate on nonvolatile pesticides in the unsaturated zone. J. Environ Quality 15: 315-322.

Walker, J. F. 1994. Statistical techniques for assessing water quality effects of BMPs. J. Irrig. Drainage Eng. 120: 334-347.

Walter, M. T., M. F. Walter, E. S. Brooks, T. S. Steenhuis, J. Boll, and K. Weiler. 2000. Hydrologically sensitive areas: Variable source area hydrology implications for water quality risk assessment. J. Soil Water Conserv. 55: 277-284.

Walter, M. T., E. S. Brooks, M. F. Walter, T. S. Steenhuis, C. A. Scott, and J. Boll. 2001. Evaluation of soluble phosphorus transport from manure applied fields under various spreading strategies. J. Soil Water Conserv. 56: 329-336.

Ward, A. D., E. Desmond, N. R. Fausey, T. J. Logan, and W. G. Logan. 1993. Development studies with the ADAPT water table management model. 15th International Congress on Irrigation and Drainage. The Hague, The Netherlands.

Ward, A. D., J. L. Hatfield, J. A. Lamb, E. E. Alberts, T. J. Logan, and J. L. Anderson. 1994. The management systems evaluation areas program: tillage and water quality research. Soil Tillage Res. 30: 49-74.

White, K. L., and I. Chaubey. 2005. Sensitivity analysis, calibration and validations for a multisite and multivariable SWAT model. J. American Water Res. Assoc. 41: 1077-1089.

Williams, J. R., C. A. Jones, and P. T. Dyke. 1984. A modeling approach to determining the relationship between erosion and soil productivity. Trans. ASAE 27: 129-144.