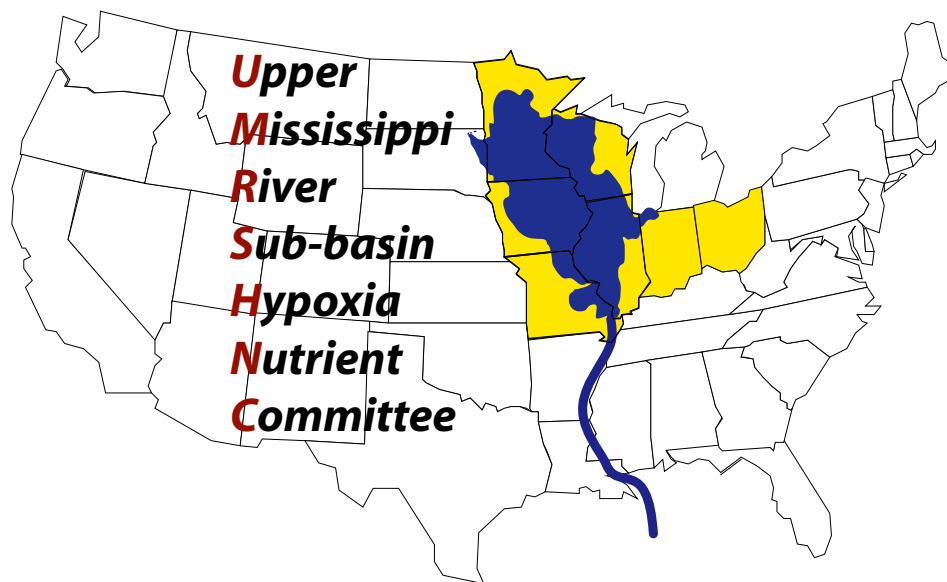


Gulf Hypoxia and Local Water Quality Concerns Workshop

A workshop assessing tools to reduce agricultural nutrient losses to water resources in the Corn Belt



September 26 – 28, 2005
Iowa State University
Ames, Iowa

Gulf Hypoxia and Local Water Quality Concerns Workshop

September 26 – 28, 2005

Iowa State University
Ames, Iowa

Table of Contents

Workshop program and agenda.....	6
Scheman Building room diagram	9
Map and directions to Reiman Gardens (<i>Monday evening reception</i>)	10
Session 1 Understanding Nutrient Fate and Transport, Including the Importance of Hydrology in Determining Losses, and Potential Implications on Management Systems to Reduce Those Losses <i>James L. Baker, Iowa State University; Mark B. David, University of Illinois; Dean W.Lemke, Iowa Department of Agriculture and Land Stewardship</i>	11
Session 2 Drainage Water Management: A Practice for Reducing Nitrate Loads From Subsurface Drainage Systems <i>Richard A. Cooke, Associate Professor, Department of Agricultural and Biological Engineering, University of Illinois at Urbana Champaign; Gary R. Sands, Associate Professor, Department of Biosystems and Agricultural Engineering, University of Minnesota; Larry C. Brown, Professor, Department of Food, Agricultural and Biological Engineering, The Ohio State University</i>	27
Session 3 Potential of Wetlands to Reduce Agricultural Nutrient Export to Water Resources in the Corn Belt <i>William G. Crumpton, Iowa State University, Ames, IA</i>	35
Session 4 Buffers and Vegetative Filter Strips <i>Matthew J. Helmers, Assistant Professor Department of Agricultural and Biosystems Engineering, Iowa State University; Thomas Isenhart, Associate Professor Department of Natural Resource Ecology and Management, Iowa State University; Jeffrey Strock, Associate Professor Department of Soil, Water, and Climate and Southwest Research and Outreach Center, University of Minnesota, St. Paul</i>	45
Session 5 Nitrogen Rates <i>John E. Sawyer, Associate Professor and Soil Fertility Extension Specialist, Department of Agronomy, Iowa State University; Gyles W. Randall, Soil Scientist and Professor, Southern Research and Outreach Center, University of Minnesota</i>	59

Session 6	Nitrogen Application Timing, Forms and Additives <i>Gyles Randall, Soil Scientist and Professor, Southern Research and Outreach Center, University of Minnesota; John Sawyer, Associate Professor and Soil Fertility Extension Specialist, Dept. of Agronomy, Iowa State University.....</i>	73
Session 7	Agronomic and Environmental Implication of Phosphorus Management Practices <i>Antonio P. Mallarino, Department of Agronomy, Iowa State University, Ames; Larry G. Bundy, Department of Soil Science, University of Wisconsin-Madison</i>	85
Session 8	Using Manure as a Fertilizer for Crop Production <i>John A. Lory and Ray Massey, Associate Professors of Extension, University of Missouri; Brad Joern, Professor, Purdue University.....</i>	105
Session 9	Effects of Erosion Control Practices on Nutrients Losses <i>George F. Czapar, University of Illinois; John M. Laflen, Iowa State University; Gregory F. McIsaac, University of Illinois; Dennis P. McKenna, Illinois Department of Agriculture</i>	115
Session 10	Potential and Limitations of Cover Crops, Living Mulches and Perennials to Reduce Nutrient Losses to Water Sources from Agricultural Fields <i>T.C. Kaspar, Plant Physiologist, USDA-ARS, National Soil Tilth Laboratory; E.J. Kladivko, Professor of Soil Physics, Agronomy Dept., Purdue University; J.W. Singer, Research Agronomist, USDA-ARS, National Soil Tilth Laboratory</i>	125
Session 11	Sustaining Soil Resources While Managing Nutrients <i>Dan B. Jaynes, National Soil Tilth Laboratory, Ames, IA; Douglas L. Karlen, National Soil Tilth Laboratory, Ames, IA</i>	141
Session 12	Field-Scale Tools for Reducing Nutrient Losses to Water Resources <i>Larry G. Bundy, Department of Soil Science, University of Wisconsin-Madison; Antonio P. Mallarino, Department of Agronomy, Iowa State University, Ames</i>	151
Session 13	Watershed-Scale Tools <i>Pete Nowak, Professor and Chair, Gaylord Nelson Institute for Environmental Studies, University of Wisconsin-Madison; John Norman, Professor, Department of Soil Science, University of Wisconsin-Madison; Dave Mulla, Professor and W.E. Larson Chair for Soil & Water Resources, Department of Soil, Water and Climate, University of Minnesota</i>	163
Session 14	Evaluating the effectiveness of agricultural management practices at reducing nutrient losses to surface waters <i>D. J. Mulla and A. S. Birr, Univ. Minnesota, St. Paul, Minnesota; N. Kitchen, USDA-ARS, Columbia, Missouri; M. David, University of Illinois, Urbana, Illinois.</i>	171
Session 15	Where do we go from here? <i>Dean W. Lemke, Iowa Department of Agriculture and Land Stewardship; Dennis P. McKenna, Illinois Department of Agriculture</i>	195

Workshop Program

Monday, September 26

11:00 am Registration desk opens

12:00 pm Opening luncheon - Room 167-179, Scheman Building

Remarks by Secretary of Agriculture Patty Judge, Iowa Department of Agriculture and Land Stewardship, and Wendy Wintersteen, Interim Dean, College of Agriculture, Iowa State University.

Afternoon sessions – Benton Auditorium

Moderator: Rick Cruse, Iowa State University

1:00 Session 1 – Understanding Nutrient Fate and Transport, Including the Importance of Hydrology in Determining Losses, and Potential Implications on Management Systems to Reduce Those Losses

Authors: James Baker, Iowa State University; Mark David, University of Illinois; Dean Lemke, Iowa Department of Agriculture and Land Stewardship

Panelists: Gyles Randall, University of Minnesota and Dan Jaynes, USDA National Soil Tilth Laboratory

2:00 Session 2 – Drainage Water Management: A Practice for Reducing Nitrate Loads From Subsurface Drainage Systems

Authors: Richard Cooke, University of Illinois; Larry Brown, Ohio State University; Gary Sands, University of Minnesota

Panelists: Norman Fausey, USDA Agricultural Research Service and Don Pitts, Illinois Natural Resources Conservation Service

3:00 Break

3:30 Session 3 – Potential of Wetlands to Reduce Agricultural Nutrient Export to Water Resources in the Corn Belt

Author: William Crumpton, Iowa State University

Panelists: David Kovacic, University of Illinois; Don Hey, The Wetlands Initiative; Mary Skopec, Iowa Geological Survey Bureau; Gary Sands, University of Minnesota

4:30 Session 4 – Buffer and vegetative filter strips

Authors: Matt Helmers, Iowa State University; Jeff Strock, University of Minnesota; Tom Isenhardt, Iowa State University

Panelist: Mike Dosskey, USDA National Agroforestry Center and Seth Dabney, USDA Agricultural Research Service

5:30 Workshop adjourns

6:00 Dinner and reception at Iowa State University Reiman Gardens

A brochure describing the Gardens is included in the cover pocket of this notebook. A map showing the location of the Gardens is on page 10 of this notebook.

Tuesday, September 27

Morning sessions – Benton Auditorium

Moderator: Dennis Keeney, Iowa State University

- 7:30 am Morning refreshments
- 8:00 Session 5 – Nitrogen rates (combined with Session 6)
 Session 6 – Nitrogen Application Timing, Forms and Additives
Authors: John Sawyer, Iowa State University and Gyles Randall, University of Minnesota
Panelists: Robert Hoeft, University of Illinois; Peter Scharf, University of Missouri; Alfred Blackmer, Iowa State University; Sylvie Brouder, Purdue University; Eileen Kladienko, Purdue University; Larry Bundy, University of Wisconsin; James Baker, Iowa State University
- 9:45 Break
- 10:00 Session 7 – Agronomic and Environmental Implication of Phosphorus Management Practices
Authors: Antonio Mallarino, Iowa State University and Larry Bundy, University of Wisconsin
Panelists: Brad Joern, Purdue University; Robert Hoeft, University of Illinois; Peter Scharf, University of Missouri
- 11:00 Session 8 – Using Manure as a Fertilizer for Crop Production
Authors: John Lory, University of Missouri; Brad Joern, Purdue University; Ray Massey, University of Missouri
Panelists: Carrie Laboski, University of Wisconsin; Stewart Melvin, Curry-Wille & Associates.
- 12:00 pm Lunch - Room 167-179, Scheman Building
Remarks by Diane Regas, Director of Office of Wetlands, Oceans and Watersheds, U.S. Environmental Protection Agency

Afternoon sessions – Benton Auditorium

Moderator: Tom Hunt, University of Wisconsin

- 1:00 Session 9 – Effects of Erosion Control Practices on Nutrients Losses
Authors: George Czapar, University of Illinois; Greg McIsaac, University of Illinois; Dennis McKenna, Illinois Department of Agriculture; John Laflen, Iowa State University
Panelist: Chi-hua Huang, Purdue University
- 2:00 Session 10 – Potential and Limitations of Cover Crops, Living Mulches and Perennials to Reduce Nutrient Losses to Water Sources from Agricultural Fields
Authors: Tom Kaspar, USDA National Soil Tilth Laboratory; Eileen Kladienko, Purdue University; Jeremy Singer, USDA National Soil Tilth Laboratory
Panelist: Dale Mutch, Michigan State University; Steve Morse, University of Minnesota
- 3:00 Break
- 3:30 Session 11 – Sustaining Soil Resources While Managing Nutrients
Authors: Doug Karlen, USDA National Soil Tilth Laboratory and Dan Jaynes, USDA National Soil Tilth Laboratory
Panelists: William Crumpton, Iowa State University; Michael Russelle, USDA Agricultural Research Service; Lowell Gentry, University of Illinois
- 4:30 **Evening social – cash bar and light hors d'oeuvres – 1st Floor Lobby, Scheman Building**
- 5:30 UMRSHNC stakeholder advisory group meeting – Room 150-154, Scheman Building

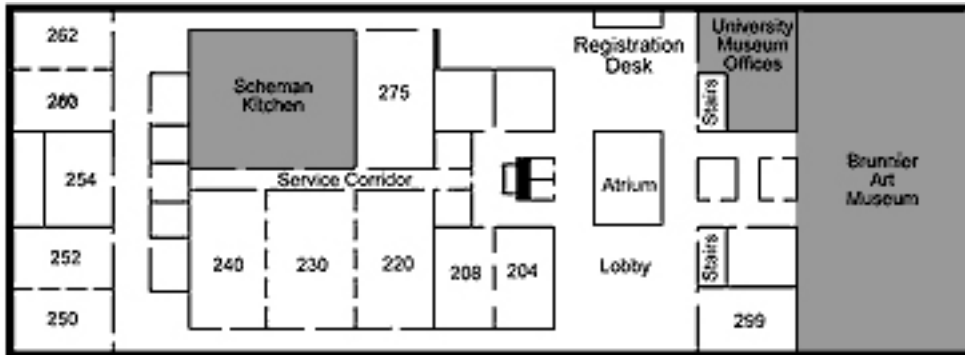
Wednesday, September 28

Morning sessions – Benton Auditorium

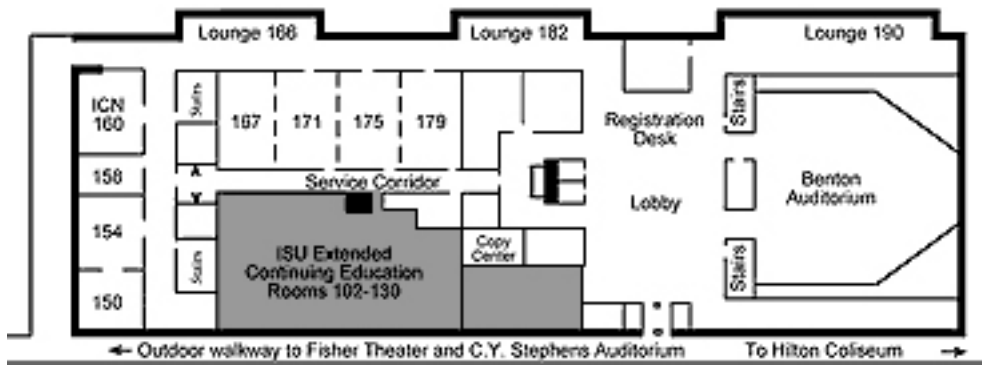
Moderator: William Kurtz, University of Missouri

- 7:30 am Morning refreshments
- 8:00 Session 12 – Field-Scale Tools for Reducing Nutrient Losses to Water Resources
Authors: Larry Bundy, University of Wisconsin and Antonio Mallarino, Iowa State University
Panelists: John Moncrief, University of Minnesota; Wes Jarrell, University of Illinois; Laura Good, University of Wisconsin
- 9:00 Session 13 – Watershed-scale tools
Authors: Peter Nowak, University of Wisconsin; John Norman, University of Wisconsin; David Mulla, University of Minnesota
Panelists: Max Schnepf, Soil and Water Conservation Society and Mike Burkart, USDA National Soil Tilth Laboratory
- 10:00 Break
- 10:30 Session 14 – Evaluating the effectiveness of agricultural management practices at reducing nutrient losses to surface waters
Authors: David Mulla, University of Minnesota; Newell Kitchen, USDA Agricultural Research Service; Mark David, University of Illinois
Panelists: Bernie Engel, Purdue University and Greg McIsaac, University of Illinois
- 11:30 Session 15 – Where do we go from here?
Authors: Dean Lemke, Iowa Department of Agriculture and Land Stewardship and Dennis McKenna, Illinois Department of Agriculture
Panelist: Mike Sullivan, National Resources Conservation Service; Jim Gulliford, U.S. Environmental Protection Agency; Wayne Anderson, Minnesota Pollution Control Agency; Charles Burney, Wisconsin Department of Natural Resources; Joe Engeln, Missouri Department of Natural Resources
- 12:30 pm Workshop adjourns
- 12:45 Author and panelist lunch and meeting - Room 175-179 Scheman Building

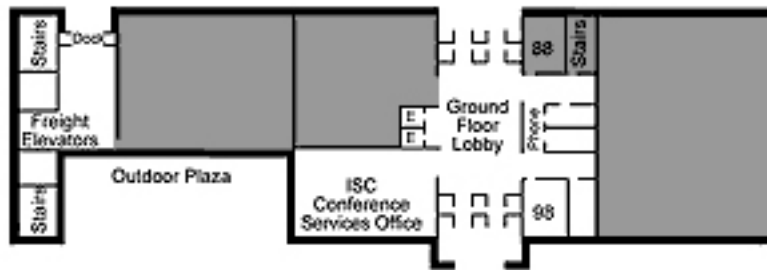
Scheman Building Layout



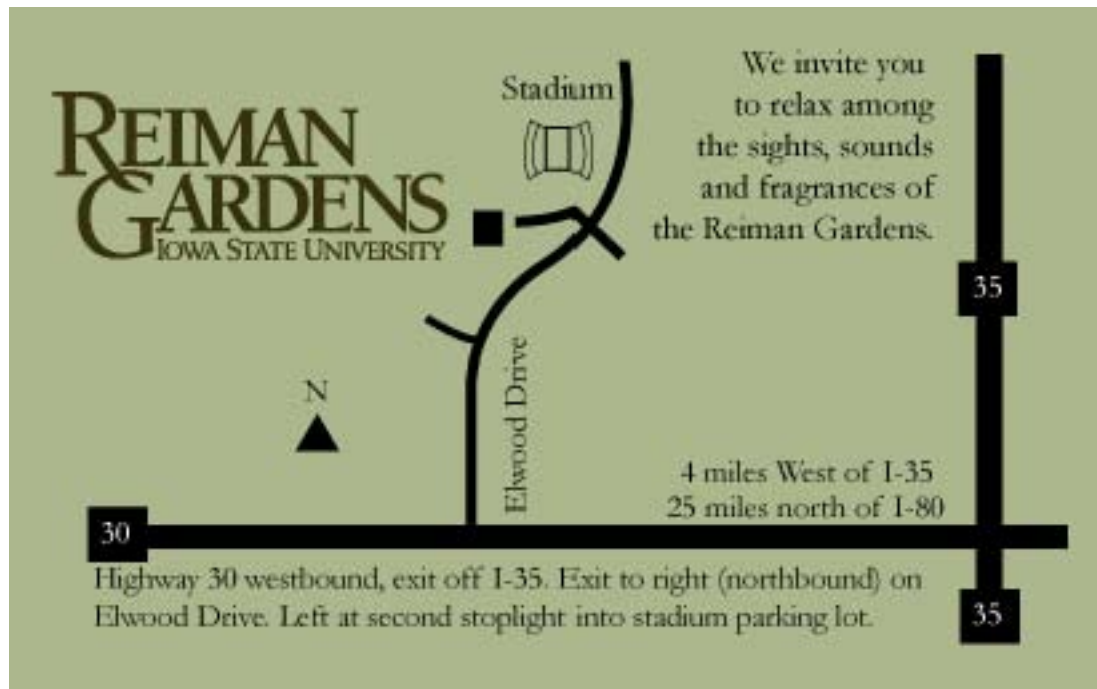
Second Floor



First Floor



Ground Floor



Reiman Gardens is conveniently located just south of Jack Trice Stadium (the ISU football stadium) in Ames.

When traveling west on Highway 30, exit to the right (northbound) on Elwood Drive. You can see the Gardens immediately by looking north. Then take a left at the second stoplight into the stadium parking lot.

Parking

Reiman Gardens is located directly south of Jack Trice Stadium, the home football field for the Iowa State Cyclones. During football Saturdays, Reiman Gardens is open, but parking is not available. At all other times, there is free parking in the lot marked for Reiman Gardens' visitors.

Understanding Nutrient Fate and Transport, Including the Importance of Hydrology in Determining Losses, and Potential Implications on Management Systems to Reduce Those Losses

James L. Baker, Iowa State University

Mark B. David, University of Illinois

Dean W. Lemke, Iowa Department of Agriculture and Land Stewardship

Introduction

Losses of the major nutrients, nitrogen (N) and phosphorus (P), from agricultural lands 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 lakes and streams.

At this time it is not clear on whether or how other cost and benefit factors such as production economics, sustainability and carbon sequestration/soil quality, and grain quality will be taken into account in the development of the criteria. Because of current water quality concerns and use impairments, and the expectation that the criteria, when developed and implemented, will lead to additional water bodies being listed as impaired, 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 right practices/systems to effectively reduce nutrient losses (i.e. to do the “right thing”). However, it is probably equally as important, if not more so, to use that understanding to not do the “wrong thing.” 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 - 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.

First and foremost in a “performance-based” approach is the question of whether there are even any practices/systems capable of reaching the stated criteria. Given that there are (and for drainage from a landscape driven by nature’s highly variable weather, it is not a given that there always will be), one advantage is that producers should have some flexibility in the choice of practices/systems to use. 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) that the number of legitimate choices of practices/systems available to producers will likely be fairly limited based on current

economic constraints, 2) being able to accurately predict the nutrient reductions (i.e. outcomes) expected for practices/systems under a standard or hypothetical set of homogenous 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 outcomes, makes predictions for realistic field/watershed conditions even more difficult, and 4) the high cost and effort needed to accurately monitor what the outcomes were, 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 NRI), as well as an acceptable mathematical modeling approach to quantify outcomes on a temporal basis (although 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. 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.

Transport Mechanisms

Losses of nutrients can occur dissolved in surface runoff water and leaching water and attached to eroded soil/sediment. These three transport mechanisms, or nutrient carriers, are illustrated in Figure 1. Concern is focused on the inorganic ions ammonium-nitrogen ($\text{NH}_4\text{-N}$), nitrate-nitrogen ($\text{NO}_3\text{-N}$), and phosphate-phosphorus (reactive-P), as well as total N and total P both in solution and associated with sediment (with P, there may need to be an additional delineation; that of “bioavailable-P”). Regional guidelines proposed for standing waters are 0.7 mg N/L and 0.035 mg P/L; regional guidelines for flowing waters are still in development. National guidelines for flowing waters for the western Corn Belt ecoregion are 2.18 mg N/L and 0.076 mg P/L.

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 represent 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 and/or concentration in that carrier reduces loss with that carrier. However, the overall impact will be determined by the summation for all three carriers, as well as the consideration of individual nutrient forms. 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 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

On the large-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 and Baker et al., 2004).

On the small-scale, 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 into the soil, one is gravity, and the other is the “suction” of water by dry soil. Since water infiltration causes the soil to become wetter, the wetting front advances down into the soil with time. During the early stages of infiltration (at the beginning of a rainfall event), the suction forces predominate over the force of gravity and the infiltration rate is at its highest. As time goes on, the infiltration rate decreases because the wetting front moves down into the soil, and the suction forces decrease. When the rainfall rate is less than the initial infiltration rate of the soil 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 rainfall intensity (both of which can change by the minute, and which makes measurement and prediction so difficult), that determines the volume and timing of surface runoff; and by subtraction the volumes of water that enters the soil to be stored for later evapotranspiration or movement from the root zone via percolation (either to groundwater or back to surface water resources through natural or artificial subsurface drainage). In general, with the exception of possibly more nitrate-nitrogen ($\text{NO}_3\text{-N}$) leaching, the higher the infiltration rate and more the infiltration, the lower the field losses of all other nutrient/forms.

Infiltration rate can also play a role in determining concentrations as well as in the masses of carriers. As shown in Figure 1, there is a thin “mixing zone” at the soil surface that interacts with and releases sediment and nutrients to rainfall 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, affects the amount of a particular nutrient form remaining in the “mixing zone” (illustrated in Figure 1 as having a thickness of 1 cm) potentially available to be lost. During a rainfall event the amount of chemical remaining in this mixing zone decreases with movement of water either over or down through this zone. Obviously the higher the rate of infiltration, the longer it is before runoff begins and the lower the chemical concentration in runoff water (and to some degree in sediment derived from soil in the mixing zone).

A second important small-scale hydrologic factor that affects nutrient leaching loss is the route of infiltration (whether the infiltrating water moves through the whole soil matrix, or whether it finds “macropores” or preferential flow paths through which to move quickly deeper, thereby, “bypassing” much of the soil). This is also illustrated in Figure 1. If the chemical of concern is within soil aggregates, flow through macropores can by-pass the chemical and leaching will be reduced. However, if the chemical is on the soil surface and dissolves in infiltrating water that is moving through macropores, leaching will be greater, quicker, and deeper than otherwise expected.

Antecedent soil moisture content is one of the important 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). All these 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 2 shows the reduction in wetland area in Iowa over the years, and Figure 3 shows the trend in drain tile produced in Illinois during the period of more intense drainage activity. Figure 4 shows the trends in U.S. cropping, with the current cropping of oats almost totally eliminated by the advent of soybeans.

Analysis of streamflow data over the second half of the 20th century (Schilling, 2005) indicated that generally baseflow and baseflow percentage have increased in that time frame and are “significantly related to increasing row crop intensity.” The subsurface drainage that has come in conjunction with more row-cropping 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 (Helmert 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 71-73% of the NO₃-N leaching losses occurred in those three months. Kladivko et al. (2004) in a 15-year study in Indiana showed most of the flow and NO₃-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.

While farming the tile-drained landscape presents an environmental challenge with respect to NO₃-N leaching, the challenge would seem to be less than the multiple challenges for the undrained landscapes that have better drainage and more surface runoff because of their sloping/rolling topography. Use of subsurface drainage (under most designs) generally reduces losses of pollutants in surface runoff (because of not only reduced surface runoff volume but also reduced pollutant concentrations in the surface runoff water). Thus to meet this challenge and preserve use of these drained and very good lands to produce food, special attention will need to be paid to in-field soil (and cropping) and nutrient (N) management to better protect NO₃-N against leaching, and/or use of additional practices such as improved water management and wetlands in the overall system design.

Nutrient Forms/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. A fourth property, the vapor pressure of ammonia ($\text{NH}_3\text{-N}$) over the soil as affected by that present and that added as manure and/or ammoniacal N fertilizers, will be a factor in how much N volatilizes.

Table 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 are given to represent the annual averages for the row-crop planted, corn rotated with soybeans, in much of the Corn Belt.

As shown for N in the soil water, $\text{NO}_3\text{-N}$ dominates over ammonium-nitrogen ($\text{NH}_4\text{-N}$); while in the solid soil itself, organic-N dominates. When comparing what is in the soil water with what is in runoff water, the stronger adsorption of $\text{NH}_4\text{-N}$ compared to $\text{NO}_3\text{-N}$ “traps” the $\text{NH}_4\text{-N}$ nearer the soil surface so the reduction is less for $\text{NH}_4\text{-N}$ (part of the reduction between concentrations in soil water at equilibrium, and what is in runoff, is due to dilution as well as incomplete mixing of rainfall-runoff with the surface soil during runoff). On the other hand, that same adsorption is what causes the relative concentrations in subsurface drainage, both relative to what is in soil water, and between $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$, to be lower for $\text{NH}_4\text{-N}$. The ratios of $\text{NH}_4\text{-N}$ and organic-N concentrations for sediment compared to their respective values for in-place soil are over unity which is due to the selective erosion process where the more chemically active, smaller, and less dense (with greater organic matter content) soil particles are preferentially transported. Organic-N in runoff water is usually less than 2 mg/L.

As shown for P in soil water, and in surface runoff and subsurface drainage, reactive-P (dissolved inorganic or molybdenum-reactive-P, sometimes termed $\text{PO}_4\text{-P}$ or ortho-P) generally makes up more than 60% of the total soluble P. In the soil, total (organic plus inorganic) P dominates over what is classified as plant “available” P determined by one of several soil P tests (in this case a Bray-1 or Mehlich-3 extractant). As with $\text{NH}_4\text{-N}$, reactive-P is somewhat trapped on the soil surface, 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 the in-place soil because of the selective erosion process. Given the very high adsorption and low solubility for total P, and realizing the ratio of the mass of surface runoff water to sediment can be as small as 100 to 1 for some rainfall-runoff events, P loss for row-cropped fields is often dominated by that lost with sediment (depending on the degree of erosion).

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 (see Figure 5). At the four sites, measurements of flow and

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 15,890 ha (in 1998, 40.3% corn, 27.2% soybeans, 11.2% oats/hay/alfalfa, 10.5% pasture, and 8.8% forest). The three intrabasin sites range from 232 ha (designated 2; 81.7% corn, 12.5 soybeans, and 5.4% pasture) to 1315 ha (designated 3; 57.4% corn, 26.0% soybeans, 13.2% oats/hay/alfalfa, and 1.8% pasture) to 1733 ha (designated 1; 44.0% corn, 40.0% soybeans 10.1% oats/hay/alfalfa, and 4.2% pasture).

Figure 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 site (number 4). These three events, preceded and separated by flow periods of only subsurface drainage, were caused by rainfalls of 30 to 40 mm, and are representative of growing-season events. Figures 7, 8, and 9 show $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$, and total-N (including N associated with sediment) concentrations versus time for the same period. Figures 10 and 11 show reactive-P and total-P (including P associated with sediment) concentrations versus time; and Figure 12 shows COD concentrations versus time, again for the same period. In agreement with the previous discussion, $\text{NO}_3\text{-N}$ concentrations decrease during a surface runoff event, while all other concentrations increase.

Tables 2, 3, and 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 losses (kg/ha) are given for both soluble forms and N and P lost with sediment. As shown in the Tables, the N losses were dominated by $\text{NO}_3\text{-N}$, which ranged from 20 to 65 kg/ha. Ammonium-nitrogen losses were less than 1 kg/ha, and soluble organic N loss (organic-N minus $\text{NH}_4\text{-N}$) and N lost with sediment were about the same, in the range of 1 to 13 kg/ha. As a percent of total N lost in all forms, that lost as $\text{NO}_3\text{-N}$ was at least 80% of the total for all four sites. As shown in the Tables, total soluble P losses were less than 1 kg/ha, 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.

Flow-weighted annual average $\text{NO}_3\text{-N}$ concentrations ranged from 9.6 to 17.5 mg/L for the three years. The maximum contaminant level (MCL) for $\text{NO}_3\text{-N}$ is 10 mg/L, which was exceeded much of the time at all four sites. $\text{NH}_4\text{-N}$ concentrations, which when above 2 mg/L can be harmful to fishes, never exceeded 1 mg/L and averaged less than 0.25 mg/L. Total N concentrations including $\text{NO}_3\text{-N}$ as well as $\text{NH}_4\text{-N}$, soluble organic-N, and sediment-N were all above 10 mg/L, which is more than four times higher than a proposed regional water quality standard of 2.2 mg/L. Soluble P concentrations averaged about 0.15 mg/L, and when sediment P was added, total P concentrations in stream flow averaged from 0.25 to 0.50 mg/L. These concentrations are three to seven times higher than a proposed regional water quality standard of 0.076 mg/L.

Management Practices/Systems

In discussion of nutrient losses, and practices/systems 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; one, under the conditions and assumptions of mass balances being made by Corn Belt states for the corn-soybean rotation, there are no “excess nutrients,” and two, in order that sufficient nutrients are available to

the plants to obtain economic 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 grain, roots, and stover, at least 180 lb N/ac need to be taken up with about 18 inches of transpiring water (about 4 million lb/ac) to produce 165 bu/acre; therefore, the ratio of $\text{NO}_3\text{-N}$ to water is 45 mg/L. Even if only half the N was taken up passively with water, the average concentration in soil water available to corn roots during the growing season would have to be over 22 mg/L to obtain economically viable yields.

Management practices/systems for the nearly flat, tile-drained areas of Iowa need to be more focused on N because of $\text{NO}_3\text{-N}$ leaching losses (Baker, 2001 and 2003). Management practices/systems 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 Lafflen, 1983 and Baker, 1987). The Iowa P index addresses this issue (Mallarino et al., 2002).

In summary, discussion to follow in this workshop on current technology in the way of in-field best management practices/systems (including fertilizer and manure management and erosion control) will show there is potential but also limitations in terms of how much reduction in nutrient losses can be achieved for row-crops. Off-site practices such as wetlands (for reduction in $\text{NO}_3\text{-N}$ transport) and vegetated filter/buffer strips (for reduction in sediment and sediment-P transport) will be discussed relative to their considerable potential to add to in-field practices/systems. Predictions that alternative cropping, in the way of small grains and more and longer sod-based rotations (including cover crops), if more economically feasible, could have a major impact on reducing nutrient losses will be made. The benefits of using field- and watershed-scale tools to design more efficient systems of practices based on site-specific conditions will be presented. And the difficult problem of how to assess the reduction in nutrient losses of implementing new practices/systems will be addressed. The questions now (and possibly for a future workshop) are how much reduction is necessary, and who should pay for the implementation of alternative practices when they do not pay for themselves.

References

- Baker, J.L. and J.M. Lafflen. 1983. Water quality consequences of conservation tillage. *J. Soil Water Conserv.* 38:186-193.
- Baker, J.L. 1987. Hydrologic effects of conservation tillage and their importance relative to water quality. Chapter 6, In: *Effects of Conservation Tillage and Groundwater Quality*, T.J. Logan, J.M. Davidson, J.L. Baker, and M.R. Overcash, eds., Lewis Publishers, Inc., Chelsea, MI.
- Baker, J.L. 1997. The critical nature of infiltration in determining the water quality of agriculture drainage. In: *Proc. Third USA/CIS Joint Conference on Environmental Hydrology and Hydrogeology-Water: Sustaining a Critical Resource*, Tashkent, Uzbekistan, Sept. 22-27, 1996, Am. Instit. Hydrol., Minneapolis, MN, p. 172-181.

- Baker, J.L. 2001. Limitations of improved nitrogen management to reduce nitrate leaching and increase use efficiency. In: *Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection; Proceedings of the 2nd International Nitrogen Conference on Science and Policy*. 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*, Albuquerque, NM, Nov. 8-12; ASAE, St. Joseph, MI, p. 74-79.
- 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: *Eighth International Drainage Symposium Proc.*, ASAE, March 21-24, Sacramento, CA; p.39-50.
- Gilliam, J.W., J.L. Baker, and K.R. Reddy. 1999. Water quality effects of drainage in humid regions. Chapter 24, In: *Drainage Monograph*, Am. Soc. Agron., Madison, WI.
- Helmers, M.J., P.A. Lawlor, J.L. Baker, S.W. Melvin, and D. W. Lemke. Temporal subsurface flow patterns from 15 years in north-central Iowa. Presented at the 2005 International Meeting of ASAE, July 17-20, Tampa, FL. ASAE Paper No. 05-2234.
- Iowa Ag Statistics. 2003. Compiled by Iowa Agricultural Statistics in cooperation with USDA-NASS and the Iowa Farm Bureau; Des Moines, IA; 137 p.
- 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.
- 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.
- 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: *Eighth International Drainage Symposium Proc.*, ASAE, March 21-24, Sacramento, CA; p.17-24.
- 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.
- Schilling, K.E. 2005. Relation of baseflow to row crop intensity in Iowa. *Ag. Ecosys. . Environ.* 105:433-438.

Table 1. Example concentrations of the nutrient forms in soil or soil water, and in surface runoff, subsurface drainage, and sediment.

Nitrogen (N)

<u>soluble</u>	Soil ¹ water	surface runoff	subsurface drainage
	-----mg/L-----		
NH ₄ -N	1.0	0.5	0.1
NO ₃ -N	50.0	4.0	15.0
<u>solid/adsorbed</u>	Soil ¹	sediment	
	-----ppm-----		
NH ₄ -N	15	20	
NO ₃ -N	0	0	
organic-N	1500	2000	

Phosphorus (P)

<u>soluble</u>	Soil ¹ water	surface runoff	subsurface drainage
	-----mg/L-----		
Reactive-P	0.6	0.2	0.050
total-P	0.9	0.3	0.075
<u>solid/adsorbed</u>	Soil ¹	sediment	
	-----ppm-----		
available-P	30	40	
total-P	600	800	

¹Top 12 inches of soil; 3% organic matter.

Table 2. Total rainfall, runoff, and losses (kg/ha) of suspended sediments and nutrients for the Upper Maquoketa Watershed in 1999.

Watershed/site	1	2	3	4
Rainfall (mm)	853	853	847	847
Runoff (mm)	429	276	499	396
NH ₄ -N	0.9	0.4	0.6	0.2
NO ₃ -N	44.9	39.2	65.4	40.7
Organic-N	2.8	1.5	2.5	2.0
Reactive-P	0.5	0.3	0.4	0.4
Total soluble P	0.8	0.4	0.6	0.7
Sediments	406	666	1,539	440
N with sediments	2.3	5.8	13.2	1.7
P with sediments	0.4	0.8	3.5	0.4

Table 3. Total rainfall, runoff and losses (kg/ha) of suspended sediments and nutrients for the Upper Maquoketa Watershed in 2000.

Watershed/site	1	2	3	4
Rainfall (mm)	790	790	830	830
Runoff (mm)	293	201	325	315
NH ₄ -N	0.4	0.2	0.7	0.3
NO ₃ -N	38.7	34.9	54.7	38.2
Organic-N	2.8	1.3	3.1	2.0
Reactive-P	0.5	0.3	0.3	0.4
Total soluble P	0.7	0.3	0.5	0.5
Sediments	136	149	2,410	884
N with sediments	0.9	0.7	5.9	1.8
P with sediments	0.8	0.3	3.1	0.8

Table 4. Total rainfall, runoff, and losses (kg/ha) of suspended sediments and nutrients for the Upper Maquoketa Watershed, 2001.

Watershed/site	1	2	3	4
Rainfall (mm)	889	889	890	890
Runoff (mm)	226	138	210	332
NH ₄ -N	0.5	0.1	0.3	0.4
NO ₃ -N	25.4	19.8	36.8	36.2
Organic-N	2.1	0.6	1.3	36.2
Reactive-P	0.4	0.2	0.2	0.4
Total soluble P	0.5	0.2	0.2	0.6
Sediments	41	44	266	326
N with sediments	0.1	0.4	1.5	2.1
P with sediments	0.0	0.7	0.4	0.5

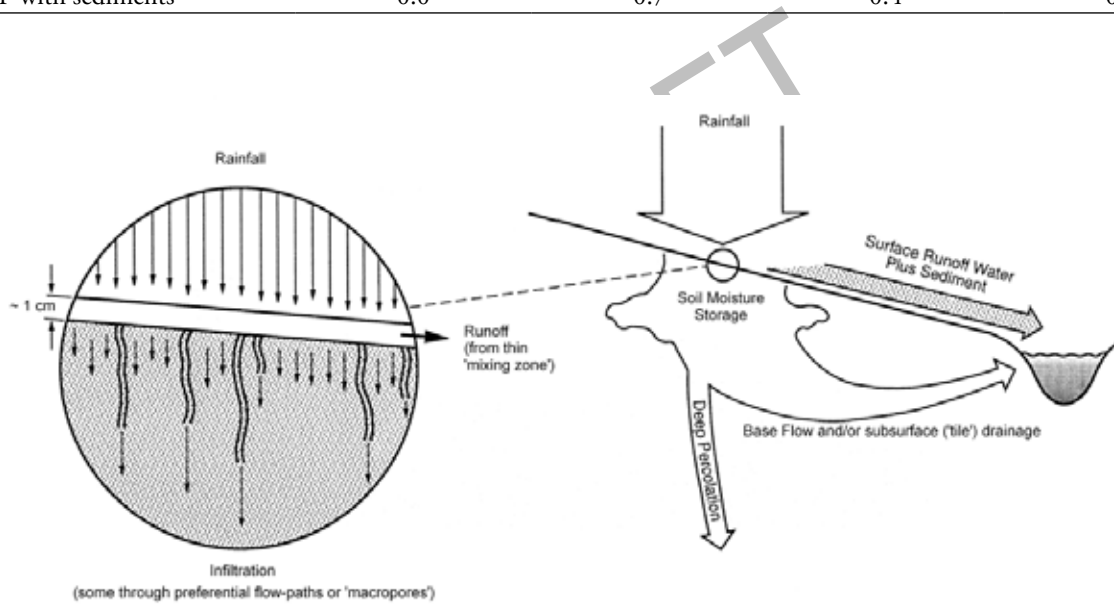


Figure 1. Schematic of transport processes and the “thin mixing zone.”

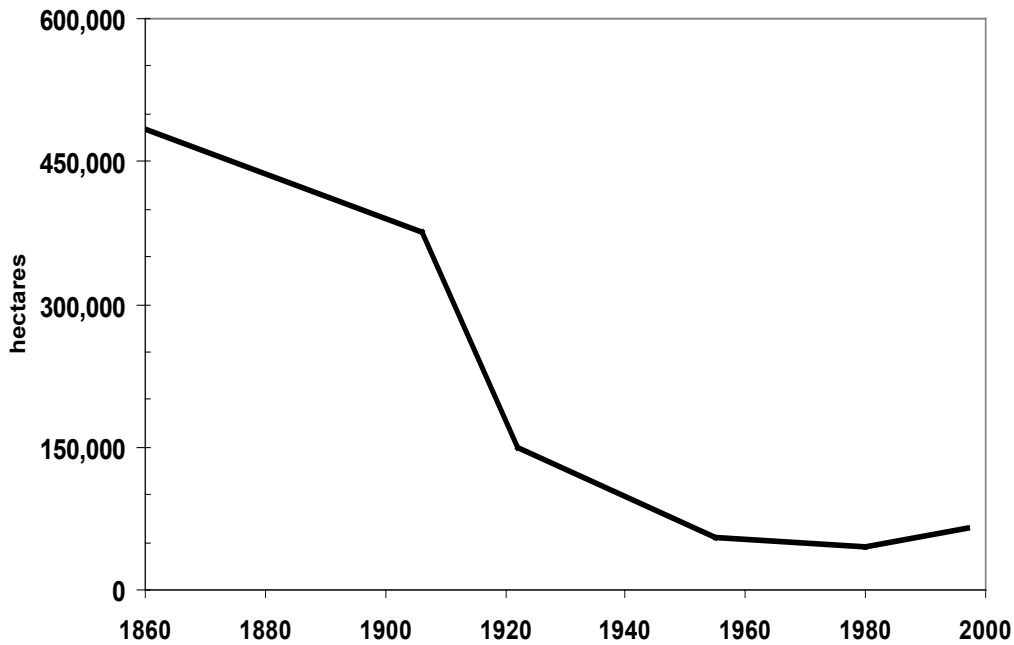


Figure 2. Trends in total wetland area in Iowa with time.

Illinois Drain Tile History

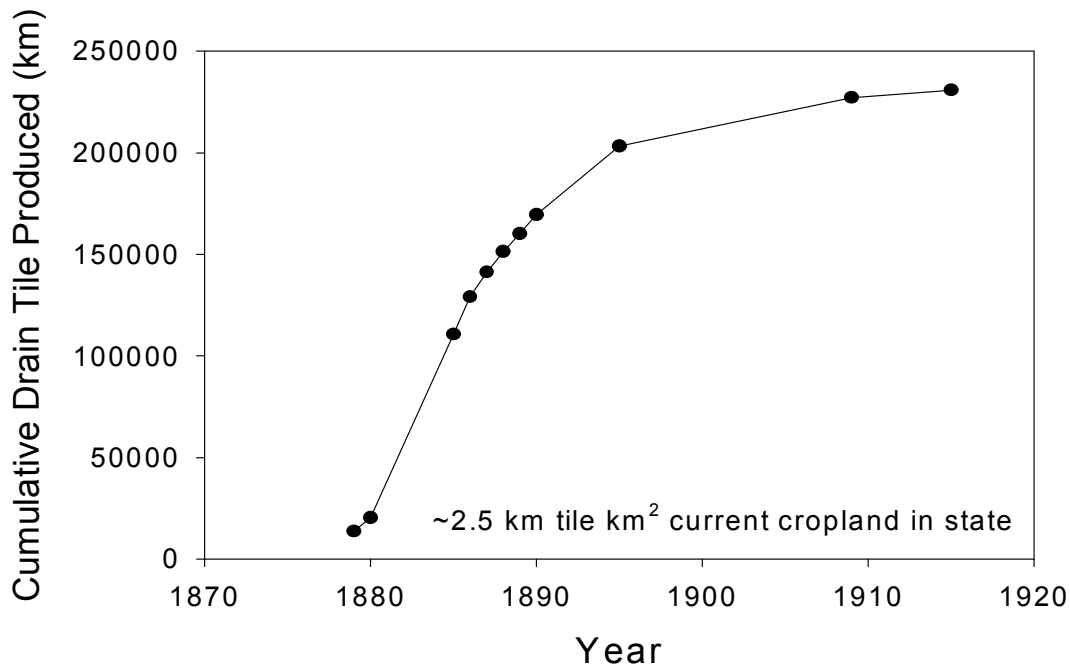


Figure 3. Trends of drain tile produced with time for the state of Illinois (most of Illinois was tiled during the 1880s, and records of tile produced were not kept after 1913, because so little was being produced).

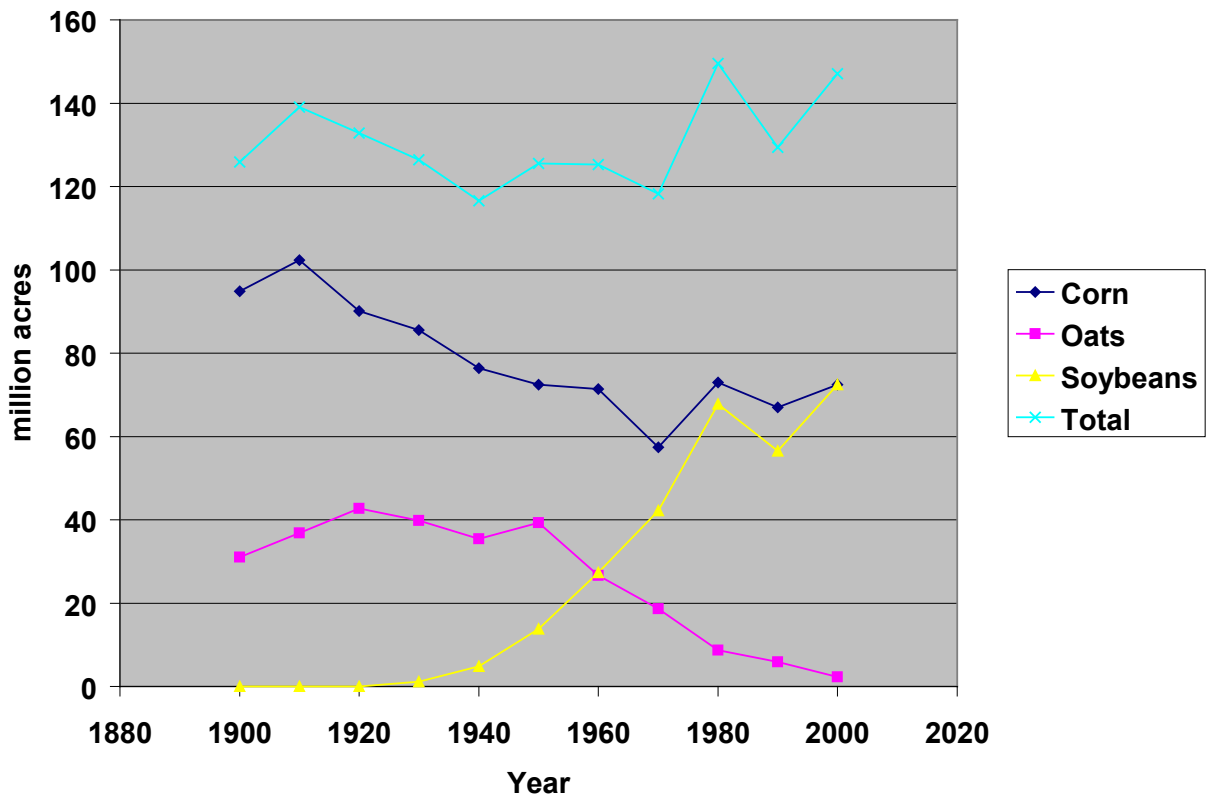


Figure 4. Trends in annual harvested acres in the U.S. (total equals corn plus soybeans plus oats).

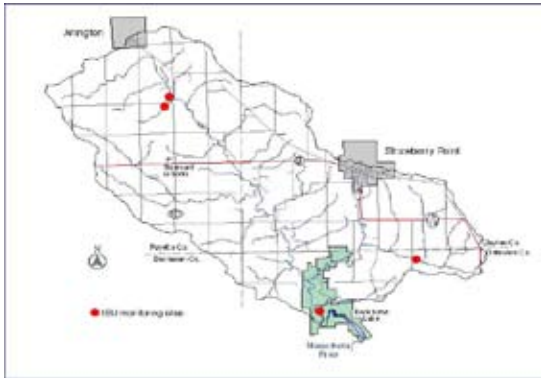


Figure 5. Maquoketa watershed monitoring locations.

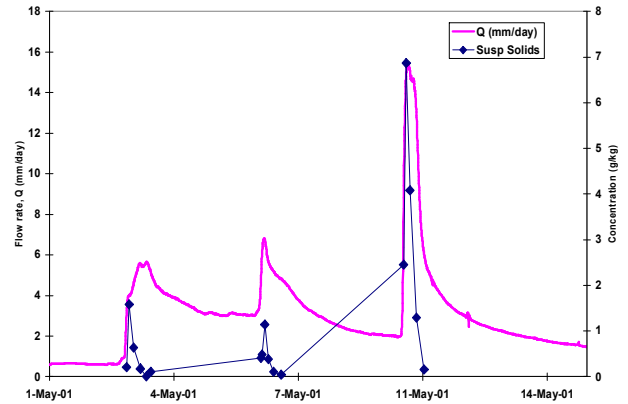


Figure 6. Suspended solids at Site 4.

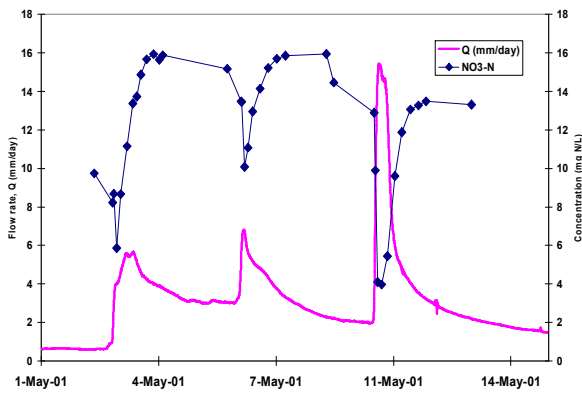


Figure 7. Nitrate-nitrogen at Site 4.

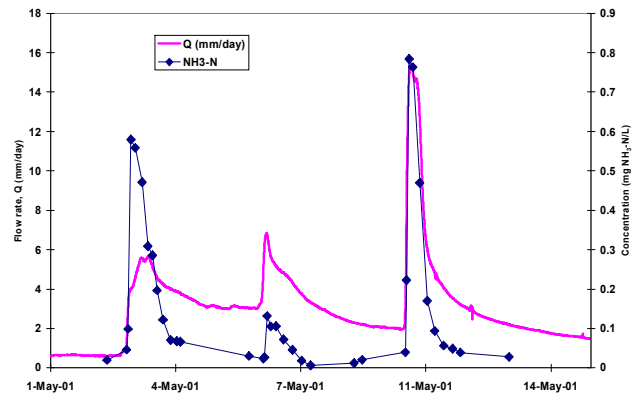


Figure 8. Ammonia-nitrogen at Site 4.

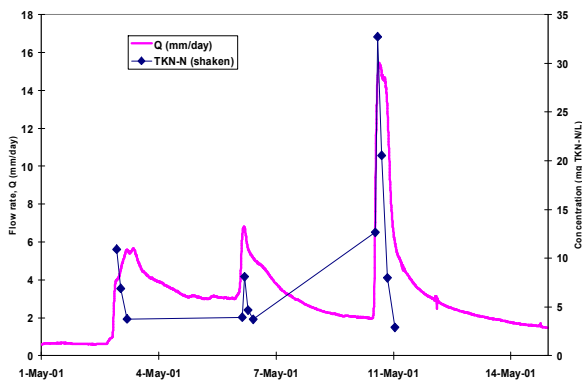


Figure 9. Organic nitrogen (shaken) at Site 4.

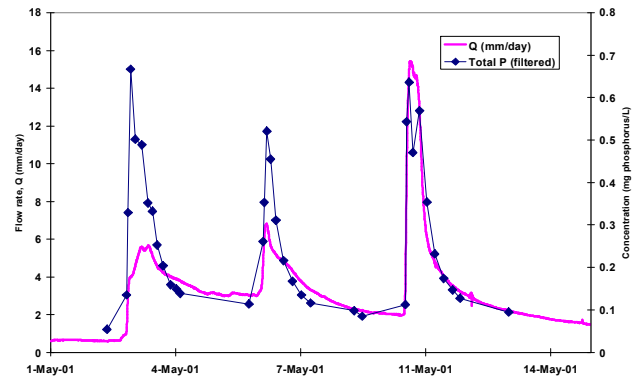


Figure 10. Total phosphorus (filtered) at Site 4.

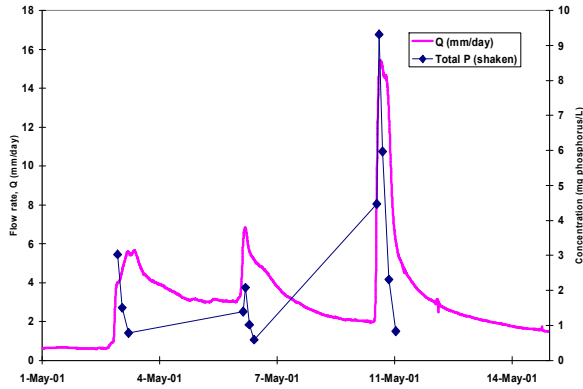


Figure 11. Total phosphorus (shaken) at Site 4.

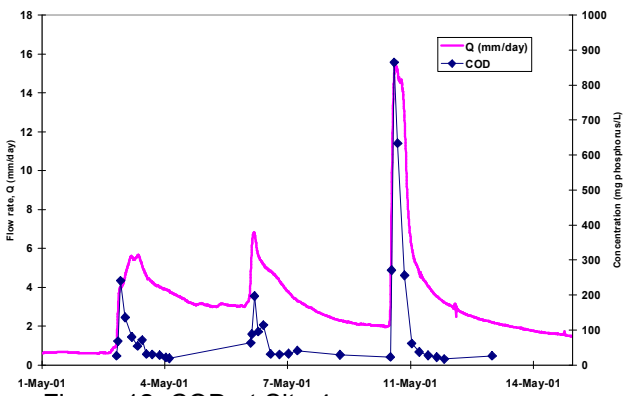


Figure 12. COD at Site 4.

DRAFT

DRAFT

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

Richard A. Cooke, Associate Professor, Department of Agricultural and Biological Engineering, University of Illinois at Urbana Champaign;

Gary R. Sands, Associate Professor, Department of Biosystems and Agricultural Engineering, University of Minnesota

Larry C. Brown, Professor, Department of Food, Agricultural and Biological Engineering, The Ohio State University

Introduction

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

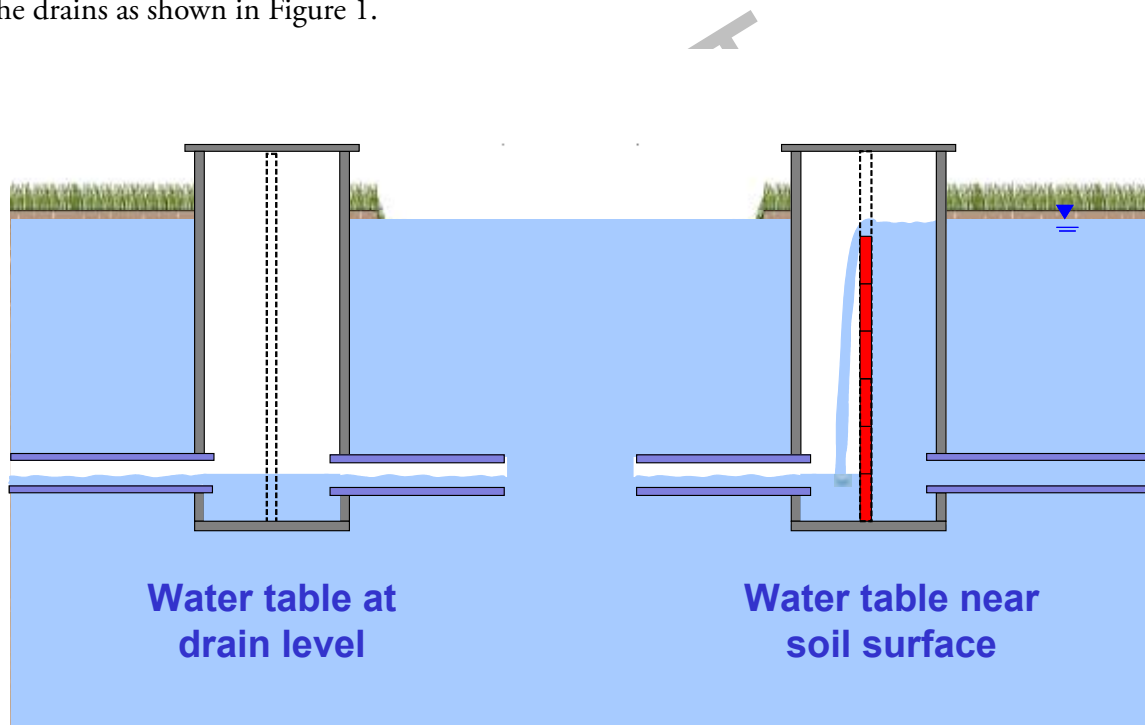


Figure 1. Using control structures to manipulate water table levels.

Two types of structures in common usage are shown in Figure 2. Water table level is controlled with these structures by adding or removing “stop logs” or using float mechanisms to regulate the opening/closing of a flow valve. There are many variations in the shapes and sizes of structures. Gate structures may either be manually operated or automated to adjust the water table level 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 can be set at or close to the soil surface between growing seasons to recharge the water table, thereby temporarily retaining contaminated water in the soil profile where it will be subjected to attenuating and nitrate transforming processes. In addition, it is possible to raise the outlet 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 weather conditions. In some soils, water may even be added during very dry periods to reduce crop loss from drought, and this practice is termed subirrigation. However, the drain spacing for subirrigation has to be half the recommended value for drainage in order for the addition of water to be very effective.

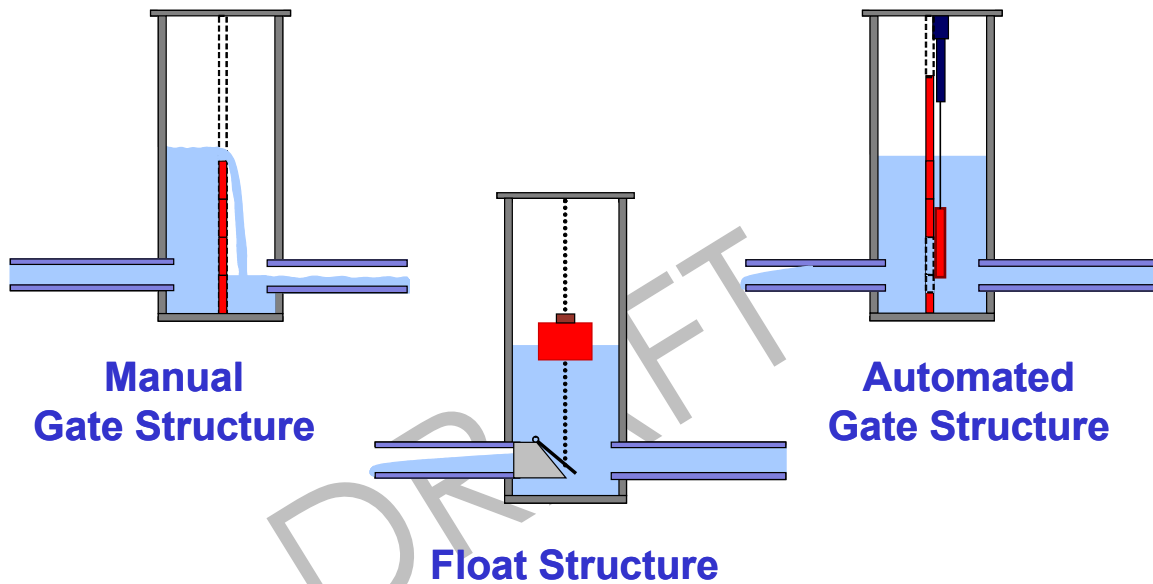


Figure 2. Types of water table control structures

In recent studies, no significant differences in nitrate concentrations have been measured in paired fields with conventional and drainage water management systems. The general consensus is that the dominant process leading to reductions in nitrate loads is a reduction in drain outflow. There is less water leaving the field through the drain pipe, and therefore, less nitrate flowing out of the drain, even if there is no change in nitrate concentration. There may be instances, however, where the implementation of drainage water management leads to increased denitrification.

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

Various researchers have found that drainage water management leads to reductions in chemical transport from agricultural fields. Munster et al. (1995) reported a decrease in aldicarb concentrations when drainage water management was implemented on cultivated fields in North Carolina. 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 drainage water management 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 holding the water table 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 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 not drained and water was added to maintain a constant water table level. 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.

A conservative estimate by consensus of drainage researchers is that drainage water management can lead to a 30% reduction in average annual nitrate loads in regions where appreciable drainage occurs in late fall and winter. This figure is based on results from North Carolina where the practice has been implemented for close to two decades, and preliminary research results in the Midwest. Measured average annual nitrate-N concentrations from drained fields in Illinois range from 8 to 19 mg/L depending on cropping practice and the timing of fertilizer application, while average annual nitrate losses 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.

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.

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%. 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 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 maintain an elevated water table, and 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 by elevating the water table will be insignificant and consequently, drainage water management systems may not be effective.

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. Figure 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 two percent). 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 making retrofitting expensive, and that the practice is economically challenging on some slopes greater than 1%.

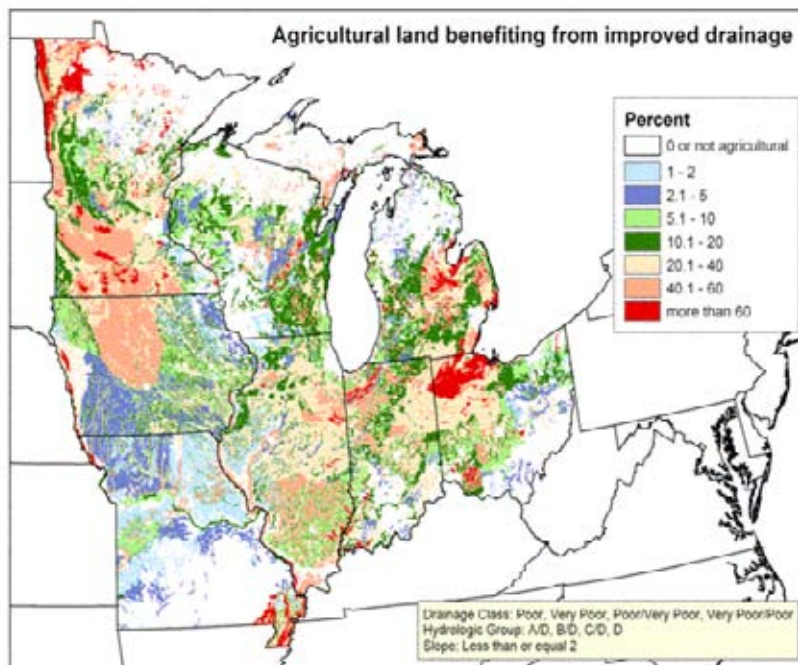


Figure 3. Agricultural land in the Midwest with the potential to benefit from Drainage Water Management

Existing drainage systems can be retrofitted for drainage water management by installing control structures at a cost of \$50 to \$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 4 are two possible drainage systems that could be installed on the same field. 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 \$50 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, the annualized cost for nitrate amelioration with drainage water management systems range from \$1.45 - \$2.05 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.

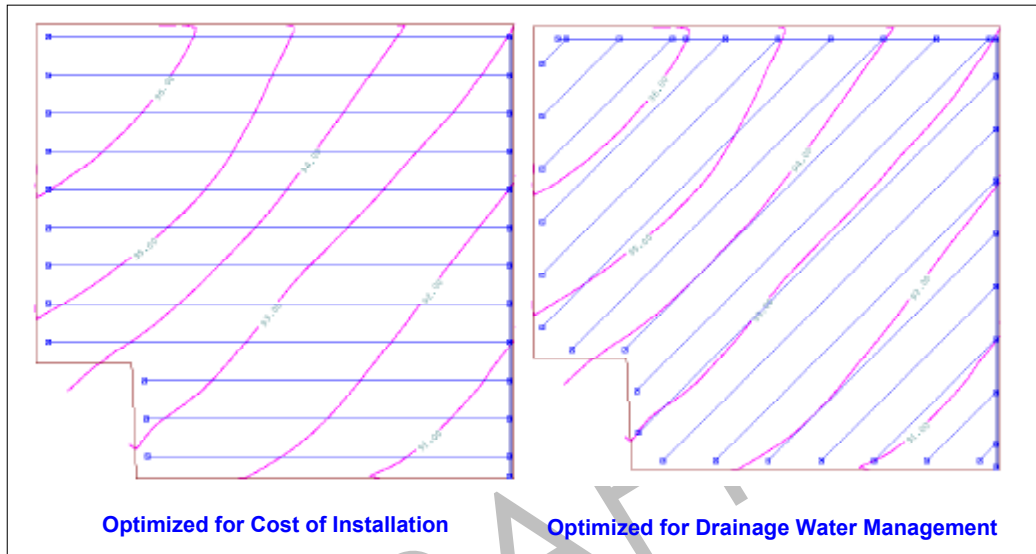


Figure 4. Effect of design objective on drainage system layout.

Because drainage water management systems are normally managed during the fallow period when there are no crops on the field, there is but little potential for yield loss. However, the systems can be used to store water in the soil during the growing season (Figure 5), provided there is adequate rainfall. 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 risk of reduced yield in very wet years.

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

One limitation to determining the efficacy of drainage water management stems from the difficulty in characterizing all the pathways by which water, and the dissolved nitrate by extension leaves a field when the water table is elevated. 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 surface runoff which might result in increased sediment and phosphorus transport from the field.

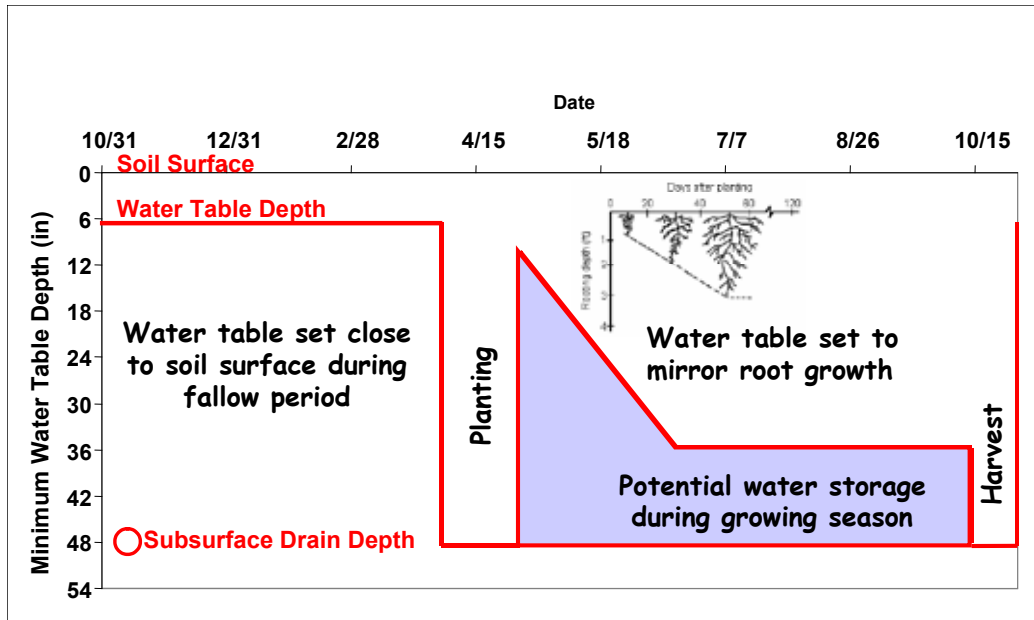


Figure 5. Drainage water management system operated for both water quality and yield benefits.

Summary

The Agricultural Drainage Management Systems Task Force was formed in 2003 in recognition of the potential for DWM to have an impact on the transport of nitrates 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” ADMSTF (2005). There is also a companion group made up of industry representatives known as the Agricultural Drainage Management Coalition that 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 members of the Agricultural Drainage Management Systems Task Force have 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 actual fact while the water table is regulated, water in excess of what is required to elevate the water table to the set level, can flow out of the soil profile (Figure 1). Other concerns, such as those relating to the drowning of earthworms, the destruction of soil structure, or excessive pressure on and freezing of subsurface drains, are also being addressed through research or educational activities.

The practice is also being used to benefit wildlife by allowing the water table in some fields to rise above of the soil surface during the fallow period. This mode of operation creates mini-wetlands that provide suitable habitats for migrating birds.

As mentioned above, drainage water management systems can be managed to store water in the soil profile and enhance yields. In the 2004 crop year, farmers in Illinois reported yield increases of 0.3 to 0.6 metric tons/hectare for corn, and 0.2 to 0.4 metric tons/hectare for soybean due to the implementation of drainage water management. However, these are but anecdotal reports; research on the yield benefits of this practice is in the early stages.

Research is being conducted in several Midwestern states to resolve many questions relating to the practice of drainage water management. In order to assess the benefits of this practice, more information is needed on the 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. In Illinois for example, detailed soil water characteristics are only available for 20 of the 244 soils that require drainage. 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

- Agricultural Drainage Management Systems Task Force. 2005. <http://extension.osu.edu/~usdasdr/ADMS/ADMSindex.htm>
- Algozany , A.S.. P. K. Kalita , J. K. Mitchell , R. A. C. Cooke , M C. Hirschi . 2005. A long-Term Monitoring of Agricultural Chemical Transport From A Flat Tile-Drained Watershed. American Society of Agricultural Engineers, St. Joseph, Michigan. Paper No. 05-2255
- 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.
- 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. *Journal of Environmental Quality* 25:317-324.
- Kalita, P. K. and R. S. Kanwar. (1993). Effect of water table management practices on the transport of nitrate-N to shallow groundwater. *Transactions of the 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. *Agricultural Water Management* 29:187-199.
- Mitchell, J.K. G.F. McIsaac, S.E. Walker, and M.C. Hirschi. (2000). Nitrate in river and subsurface flows from an east central Illinois watershed. *Transactions of the ASAE*, Vol. 43(2): 337-342.
- Munster, C.L., R.W. Skaggs, V.R. Pemmireddy. (1995). Effect of water table management on the fate of the pesticide aldicarb. *Transactions of the ASAE*. Vol. 39(1):55-66.
- Natural Resources Conservation Service. Drainage Water Management. Conservation Practice Standard Code 554. <ftp://ftp-fc.sc.egov.usda.gov/NHQ/practice-standards/standards/554.pdf>

DRAFT

Potential of Wetlands to Reduce Agricultural Nutrient Export to Water Resources in the Corn Belt

William G. Crumpton, Iowa State University, Ames, IA

Introduction and nature of the problem

Agricultural applications of fertilizers and pesticides have increased dramatically since the middle 1960s and the impact of agrochemicals on water quality has become a serious environmental concern. Nitrate (NO_3^-) is a particular concern (1) because of the potential adverse impacts on both public health and ecosystem function, (2) because of the high mobility of nitrate in surface and groundwater, and (3) because of the widespread use of nitrogen in modern agriculture. Application of fertilizer-N in the U.S. has grown from a negligible amount prior to World War II to approximately ten million metric tons of N per year (Terry and Kirby 1997), and significant amounts the fertilizer nitrogen applied to cultivated crops may be lost in agricultural drainage water, primarily in the form of nitrate (Neely and Baker 1989). The impacts of chemical intensive agriculture are a special concern in the U.S. Corn Belt. The region is characterized by intensive row crop agriculture (Figure 1) and by correspondingly high stream nitrate concentrations (Figure 2). Agricultural nitrogen loads to surface waters in the Corn Belt are among the highest in the Mississippi River Basin. In addition to impacts on water quality within the region, these nitrogen loads are considered a primary source of nitrate contributing to hypoxia in the Gulf of Mexico.

Potential of wetlands as solution to the problem

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. Nitrate is transported from crop land primarily in subsurface drainage, especially in extensively tile drained areas like the Corn Belt. 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 reduce nitrate loads by 40-90%, and this approach is particularly promising for heavily tile drained areas like the Corn Belt. This region was historically rich in wetlands, and in many areas, farming was made possible only as a result of extensive drainage (Figure 3). 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. However, wetland restorations have been motivated primarily by concern over waterfowl habitat loss, and in most cases, criteria for wetland restorations have not adequately addressed water quality functions. Most of the 500 wetland restorations surveyed by Galatowitsch (1993) in Iowa and southern Minnesota were located fairly high in the landscape and received drainage from relatively small areas. Although such restorations have significant habitat value, they do not intercept sufficient contaminant loads to significantly affect the water quality of receiving streams (Crumpton 2001). This does not negate the potential of wetlands for water quality improvement in agricultural watersheds. However, it does underscore the need for explicitly considering contaminant loading rates when planning wetland

restorations for water quality benefits (Crumpton 2001). If wetlands are to remove significant quantities of contaminants, they must first receive significant quantities of contaminants.

Performance-based approach to wetland restoration

For the past 10 years, the Iowa State University Wetlands Research Group has specifically addressed the hydrologic and water quality functions of wetlands in agricultural watersheds of the upper Midwest. This interdisciplinary research effort uses mass balance analysis and ecosystem modeling to integrate the work over spatial and temporal scales ranging from short term process studies in experimental wetland mesocosms to long term analysis and modeling of watersheds. A major objective of this effort has been to extend the application of performance forecast models to siting, design, and assessment of wetland restorations in agricultural watersheds (Crumpton and Baker 1993; Crumpton et al. 1995; Baker et al. 1997; Crumpton 2001). Results from laboratory and field mesocosms were used to develop a general model of nitrate loss for wetlands receiving non-point source nitrate loads. This model was calibrated and validated against field data for research sites in Illinois and Iowa. The nitrate loss model was then combined with a hydraulic loading model to simulate nitrate loading and loss for wetlands in agricultural watersheds. The model's predictive capability is illustrated in Figure 4 by a comparison to observed data for a restored wetland intercepting water from an agricultural drainage district in north central Iowa.

The combined model was integrated into a watershed scale framework for evaluating the hydrologic and water quality benefits of alternative wetland restoration scenarios and provided the research foundation for the Iowa Conservation Reserve Enhancement Program (CREP). The Iowa CREP is a performance-based wetland restoration program created by the Iowa Department of Agriculture and Land Stewardship, in partnership with USDA, to establish wetlands strategically located and designed to remove nitrate from agricultural tile-drainage water. Performance forecast models were used to guide the development of program criteria for the Iowa CREP based on a nitrate reduction target of 40-90%. Since the Iowa CREP is a performance-based program, nitrate reduction targets are not simply assumed based on acres of wetland enrolled but is actually calculated based on the measured performance of wetlands. Model simulations suggest that relatively small areas of wetlands can remove more than 50% of the nitrate in tile drainage water. These model predictions were tested by comparison of measured and modeled nitrate mass removal efficiencies for a series of wetlands for which monitoring data were available. The wetlands were chosen so as to encompass the 0.5% - 2.0% range of wetland/watershed ratios approved for Iowa CREP wetlands and to span a broad range of incoming nitrate concentrations. The wetlands thus span a broad range in those factors most affecting nitrate removal rates: residence times, nitrate concentrations, and nitrate loading rates. Despite spanning a broad range of loading rates and nitrate concentrations, the wetlands behaved predictably with respect to nitrate mass balance. A comparison of nitrate removal efficiency illustrates good correspondence between the measured and modeled performance of the wetlands (Figure 5).

Assessing the potential of wetland restoration to reduce agricultural nitrate loads to surface waters in the Corn Belt

We used the same mass balance modeling approach described above to estimate the nitrate reduction that could be achieved by strategically restoring wetlands in tile-drained regions across the Corn

Belt. First we developed a spatially explicit estimate of nitrate mass loading across the region (Figure 6). Next, we developed a spatially explicit estimate of hydraulic loading rates for wetlands for specific restoration scenarios. For example, wetlands could occupy 1% of the total area of their watershed and intercept 50% of the flow within the watershed. Next we developed a spatially explicit estimate of the nitrate concentrations expected in flow contributed to the wetlands (Figure 7). These distributions of hydraulic loading rates and nitrate concentrations were combined with temperature data and wetland characteristics to drive 10 year mass balance simulations for wetland restoration scenarios. The results of these simulations were used to develop a spatially explicit estimate of mass nitrate removal by wetlands (Figure 8).

The results demonstrate that wetland restorations have significant potential to reduce agricultural nitrate loads to receiving waters and that this potential is much greater in some areas than others. If wetlands are to remove significant quantities of contaminants, they must first receive significant quantities of contaminants. As a result, the greatest potential benefit for mass reduction will be found in those extensively row cropped and tile drained areas of the Corn Belt where the nitrate loading rates are highest (cf Figures 1, 4, 5 and 7). However, removal rates are not only driven by nitrate loading rates. This is because several primary determinants of wetland performance vary longitudinally across the region, including volume and timing of “runoff”, nitrate concentration, and temperature. For example, removal rates for areas in the western Corn Belt are higher than might be expected based just on nitrate loading rates because wetlands in these areas would have lower hydraulic loading rates and longer residence times. This also underscores the problem with simply extrapolating wetland performance from criteria such as the wetland/watershed area ratio criteria developed for the Iowa CREP. These criteria can not simply be extrapolated to other areas. We need to develop guidance for transferable criteria that can be adapted to different geographic areas with for example different volume and timing of “runoff”, nitrate concentration, and temperature.

References

- 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*, J.C. Hatfield, D.B. Buhler, and B.A. Stewart (ed.), CRC Press, Boca Raton, pp. 135-160.
- Crumpton, W.G. 2001. Using wetlands for water quality improvement in agricultural watersheds: the importance of a watershed scale perspective. *Water Science and Technology*. 44: 559-564.
- Crumpton, W.G. and J.L Baker. 1993. Integrating wetlands into agricultural drainage systems: Predictions of nitrate and loss in wetlands receiving agricultural subsurface drainage. Pages 118-126 in *Proceedings, International Symposium on Integrated Resource Management and Landscape Modification for Environmental Protection*, American Society of Agricultural Engineers, Dec. 13-14, Chicago, IL.
- Crumpton, W.G., J.L. Baker, J. Owens, C. Rose, and J. Stenback. 1995. Wetland and streams off-site sinks for agricultural chemicals. Pages 49-53 in *Clean Water-Clean Environment- 21st Century, Volume I: Pesticides*, American Society of Agricultural Engineers publication 2-95.

Galatowitsch, S.M. (1993). Site selection, design criteria and performance assessment for wetland restorations in the prairie pothole region. Ph.D. thesis, Iowa State University.

Neely, R.K. and J.L. Baker. 1989. Nitrogen and phosphorous dynamics and the fate of agricultural runoff. In: A.G. van der Valk (Ed.), Northern Prairie Wetlands, Iowa State University Press, Ames, Iowa. 400 pp.

Terry, D.C. and B.J. Kirby. (1997). Commercial Fertilizers 1997. Report prepared by AAPFCO and The Fertilizer Institute, Wash., DC, USA.

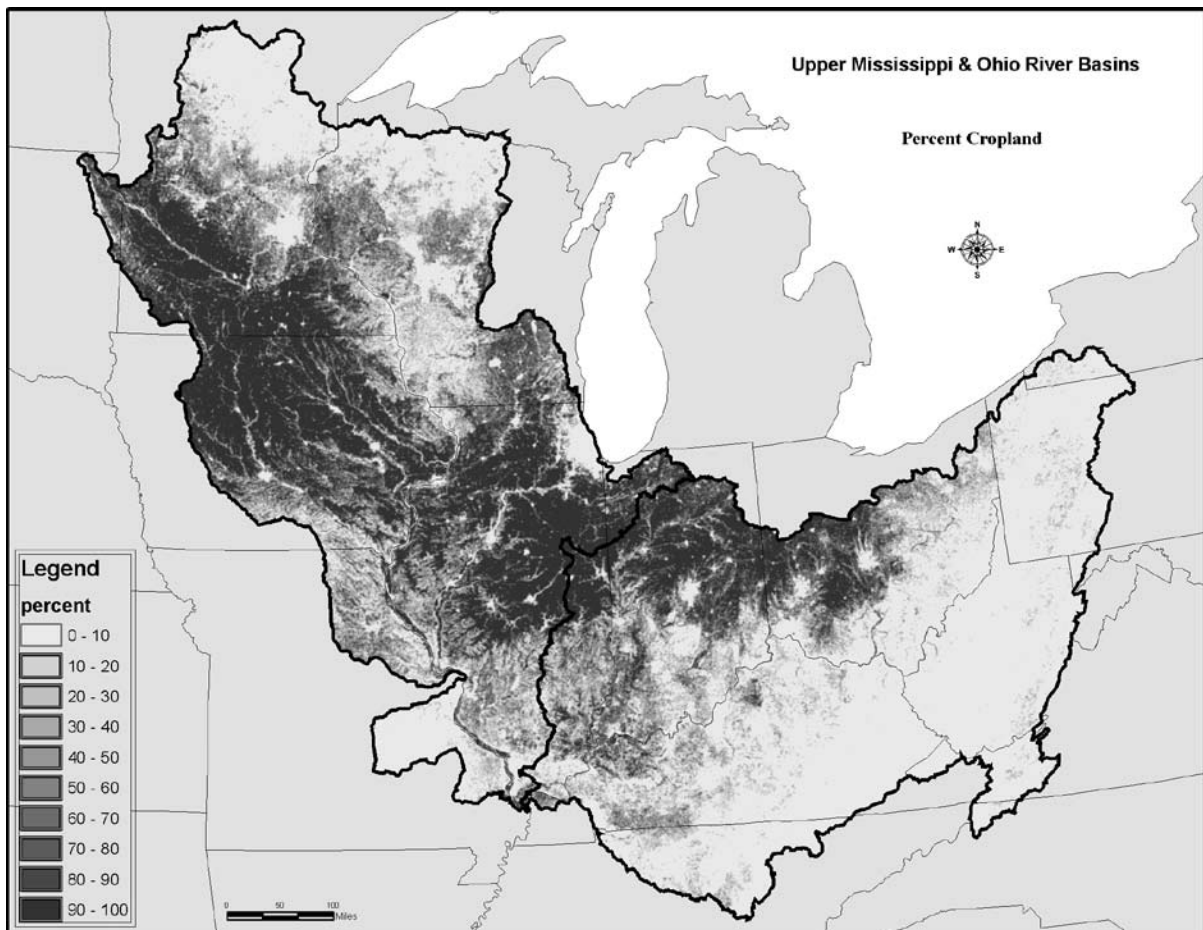


Figure 1. Land use in the Upper Mississippi and Ohio River drainages. Derived from 1992 Landsat data.

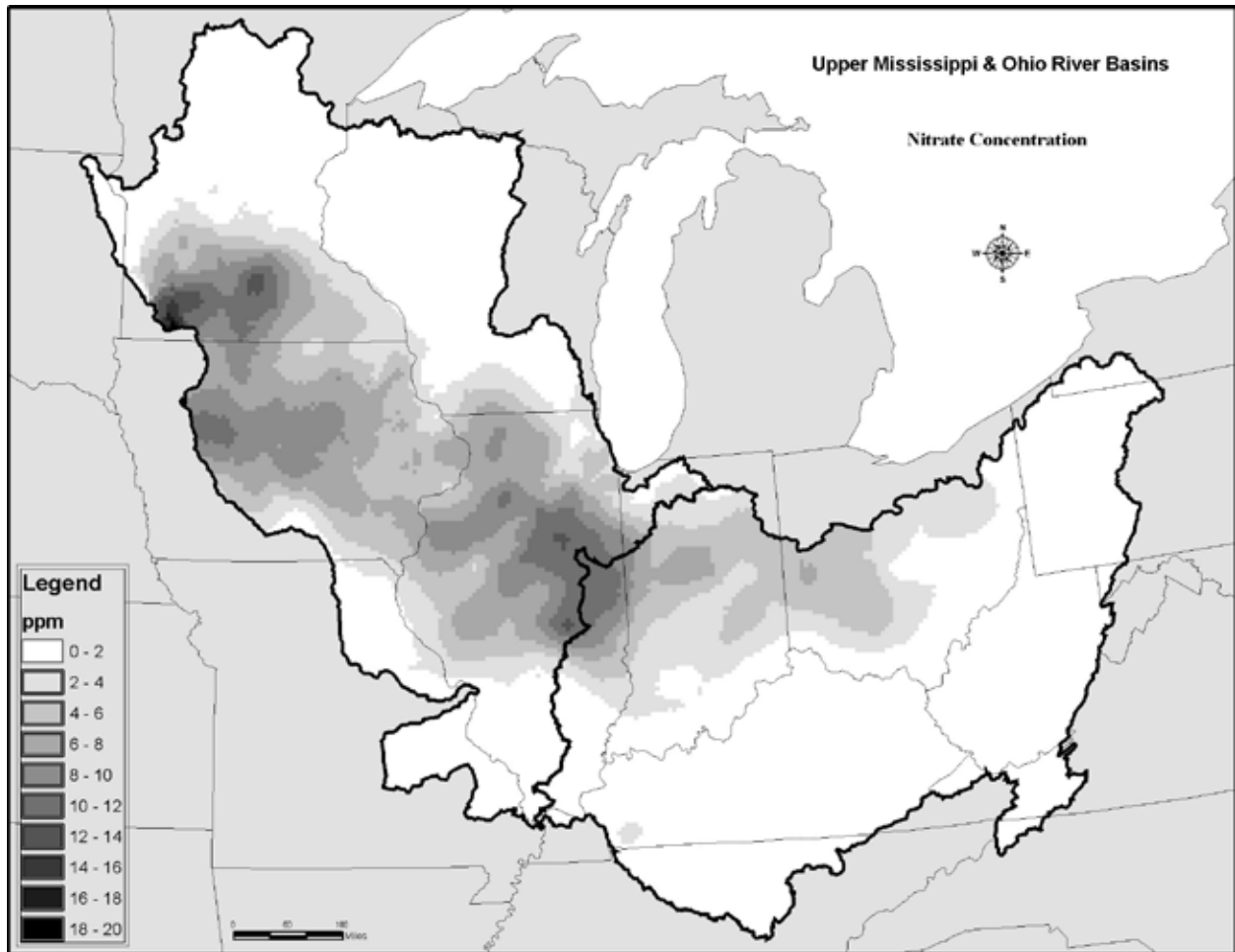


Figure 2. Estimated nitrate concentrations based on STORET and state data sets for 1990-2005.

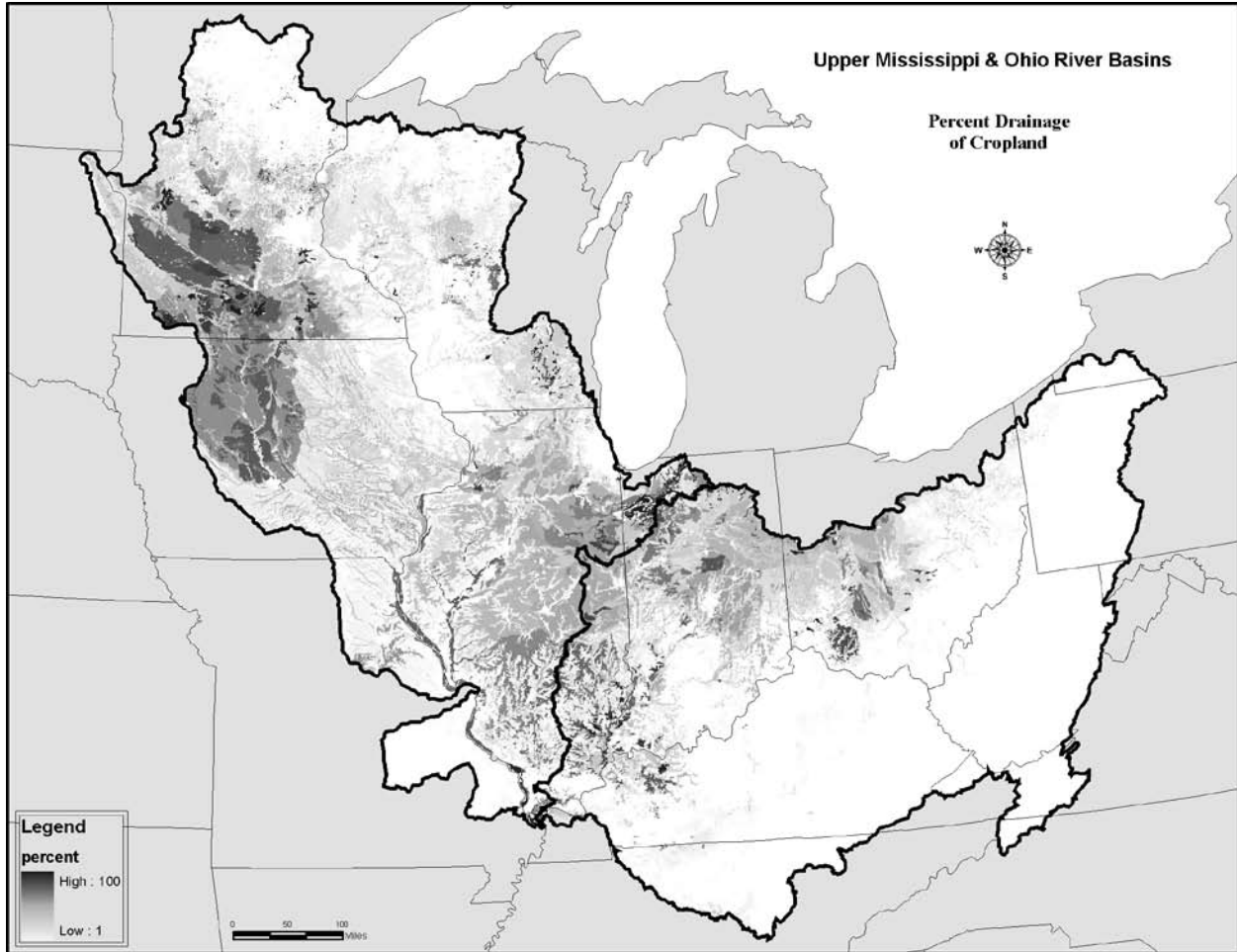


Figure 3. Land potentially benefiting from agricultural drainage in the Upper Mississippi and Ohio River drainages. Derived from NRCS STATSGO soil properties and Landsat data.

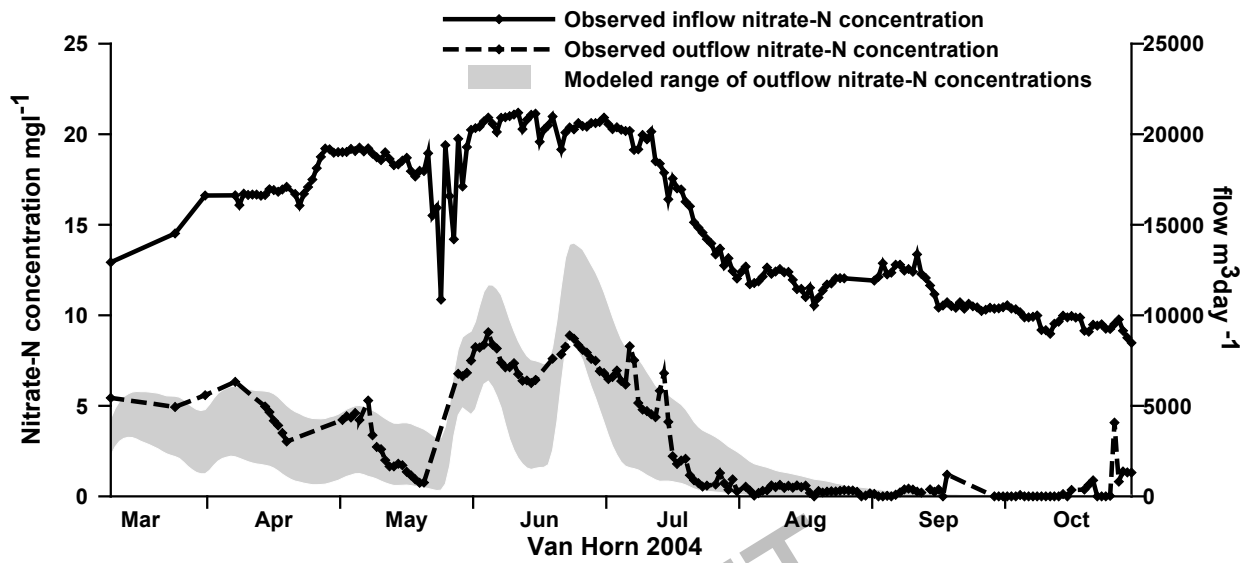


Figure 4. Observed nitrate concentrations and flow rates at inflow to Van Horn Wetland in 2004.

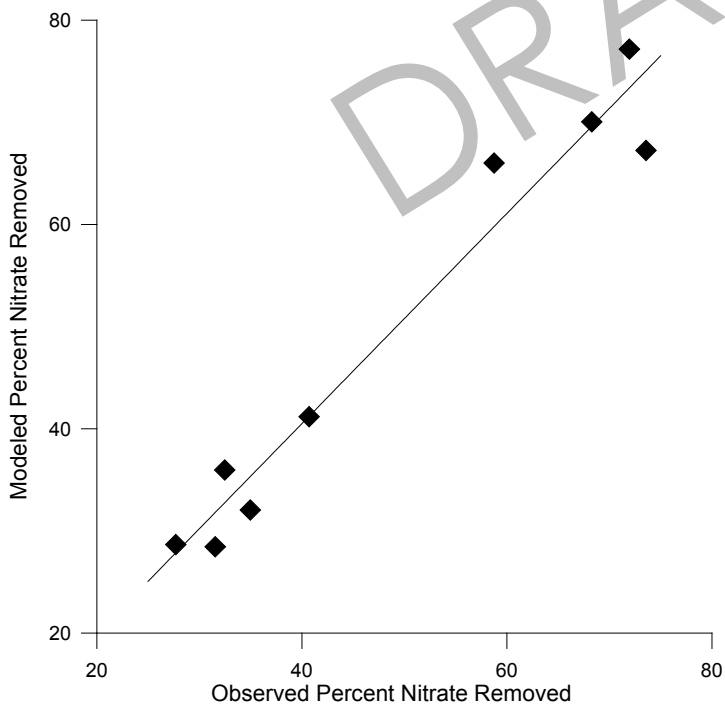


Figure 5. Comparison of measured and modeled wetland nitrate removal efficiencies ($r^2=0.95$).

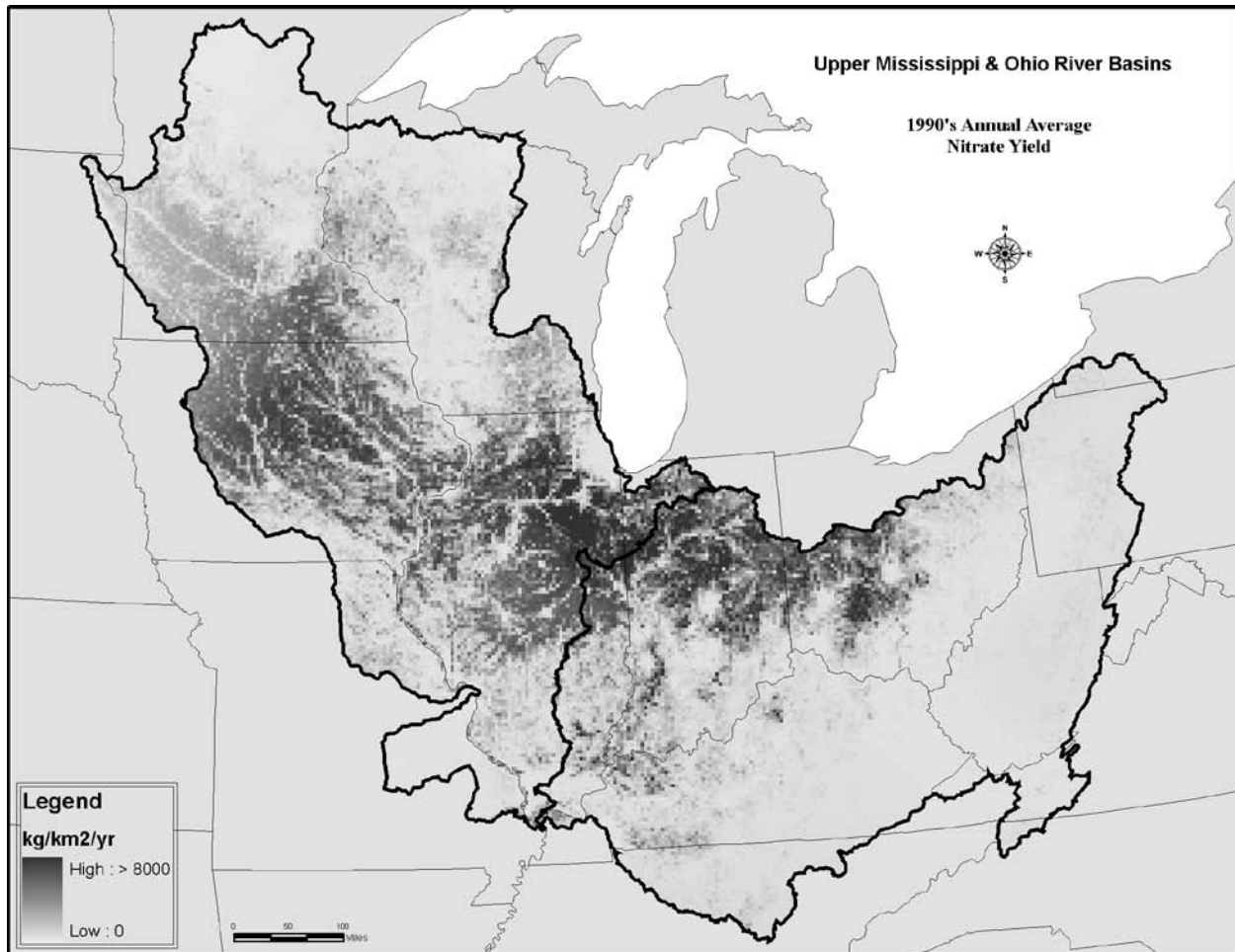


Figure 6. Estimated mass nitrate yield for specific areas within the Upper Mississippi and Ohio River drainages.

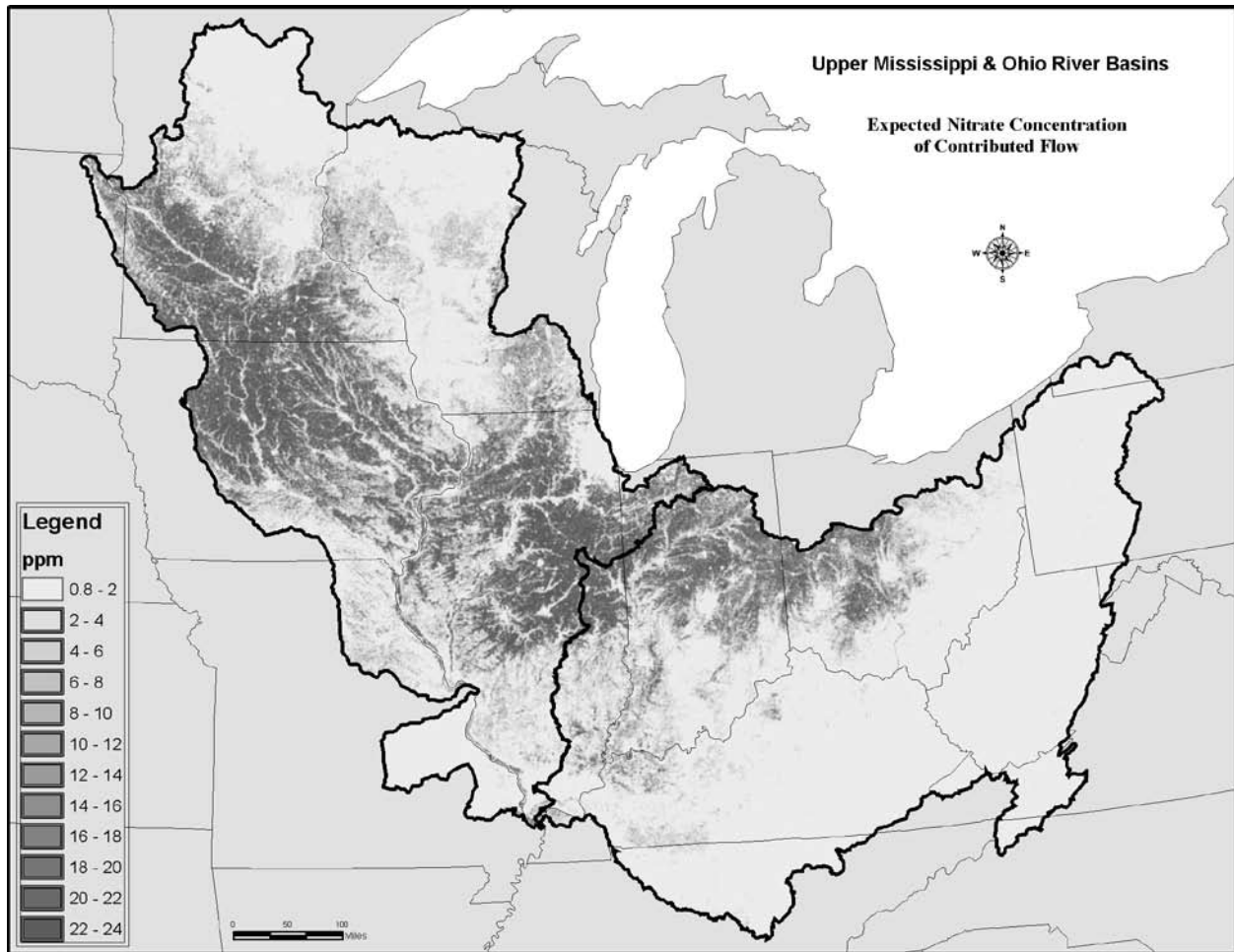


Figure 7. Estimated nitrate concentrations in flow contributed by specific areas within the Upper Mississippi and Ohio River drainages.

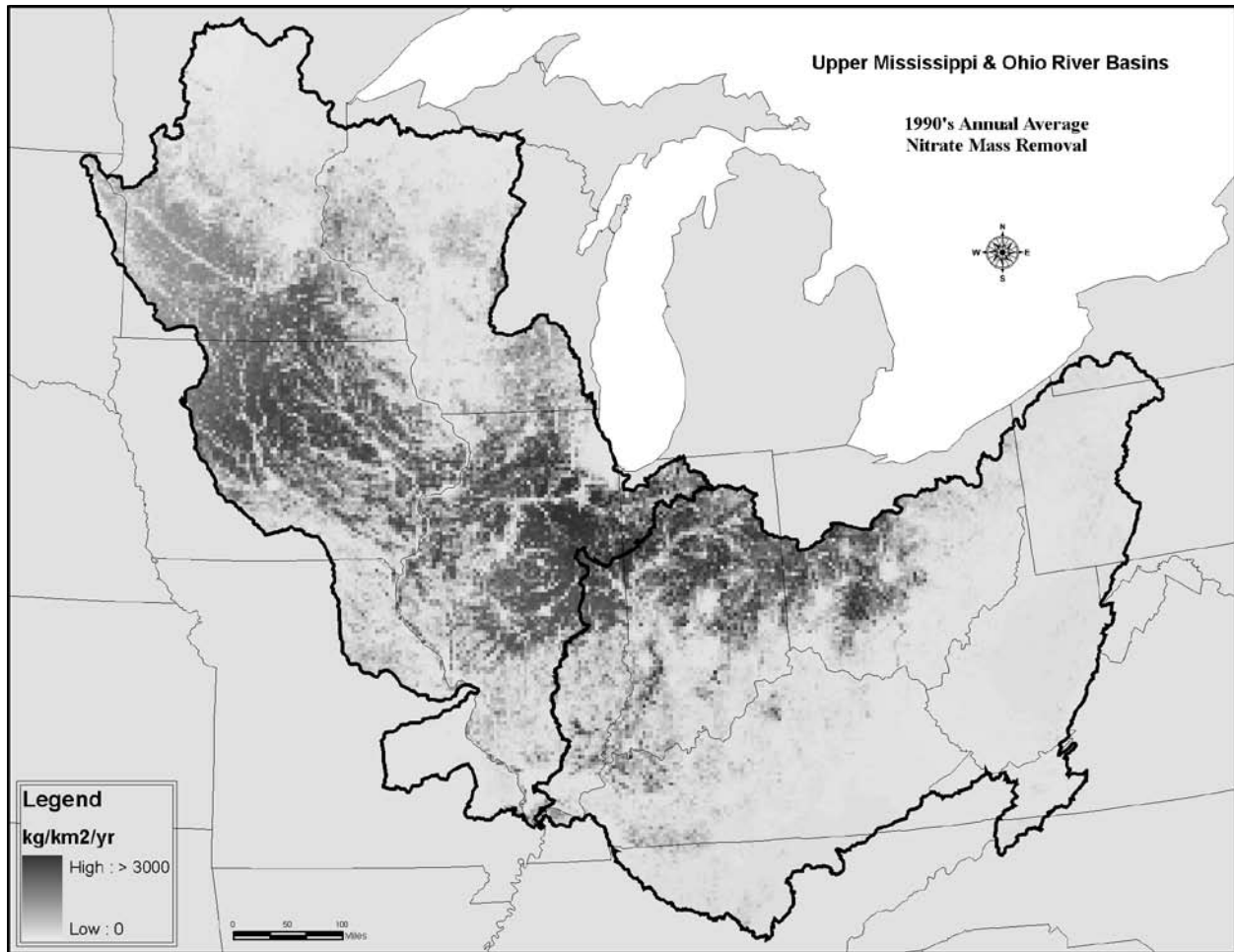


Figure 8. Estimated mass nitrate removal for restored wetlands occupying 1 % of the total acreage for specific areas within the Upper Mississippi and Ohio River drainages.

Buffers and Vegetative Filter Strips

Matthew J. Helmers, Assistant Professor Department of Agricultural and Biosystems Engineering, Iowa State University

Thomas Isenhardt, Associate Professor Department of Natural Resource Ecology and Management, Iowa State University

Jeffrey Strock, Associate Professor Department of Soil, Water, and Climate and Southwest Research and Outreach Center, University of Minnesota, St. Paul

Introduction

This paper describes the use of buffers/vegetative filter strips relative to water quality. In particular we are primarily discussing the herbaceous components of the following NRCS Conservation Practice Standards:

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

Common purposes of these herbaceous components would be 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;
- to reduce reduce sheet and rill erosion;
- to convey runoff from terraces, diversions, or other water concentrations without causing erosion or flooding; and (Grassed Waterway)

- to reduce gully erosion. (Grassed Waterway and Vegetative Barrier)

A primary mechanism by which buffers provide water-quality benefits is through reducing flow velocities because the vegetation provides greater resistance to water flow. The reduction in overland flow velocity then causes deposition of some of the suspended particulates. Increased resistance to overland 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 would also occur in the buffer reducing the 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 due to dense, perennial vegetation providing greater resistance to erosion.

Flow conditions will 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. As a result, the flow rate entering buffer systems 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 effect whether the vegetation is submerged or unsubmerged. Conditions under which vegetation becomes submerged will 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. From this, the specific flow rate range for filter strip type systems would be less than approximately $0.02 \text{ m}^2 \text{ s}^{-1}$ and the range for grassed waterways would be greater than this. Vegetative barriers have specific flow rates that span the range between filter strips and grassed waterways.

Potential

Researchers have conducted extensive studies on the pollutant-trapping capability of 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 efficiency from 9% to 100% (Arora et al. (1996), Arora et al. (1993), 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. (1994), Parsons et al. (1990), Patty et al. (1997), Schmitt et al. (1999), and Tingle et al. (1998)).

Numerous studies have also examined the nutrient-trapping effectiveness of buffers. Dosskey (2001) summarizes 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; and 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 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. (1996), Arora et al. (1993), 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. (1994), Parsons et al. (1990), 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; and 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 9-98% (Arora et al., 1996 and Schmitt et al., 1999).

For a source area with a field width (flow length) of 200 m, the buffer width (flow length) would be 40 meters for a drainage area to buffer ratio of 5:1. This size of buffer is likely unrealistic; a more typical width (flow length) of 20 m would result in a larger ratio of drainage area to buffer area. Based on guidelines from 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 flow pathways 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.

While the discussion of drainage area to buffer ratio captures one source of variability that can affect buffer performance since this ratio may affect the flow rate per unit width entering a filter, other factors that would be expected to influence flow reaching a buffer would be the 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; and Helmers et al., 2002). In addition, while this ratio may be important in many cases narrow buffers have been shown to provide significant benefits. Narrow buffers (<1 m) such as vegetative barriers have been shown to trap significant amounts of sediment (Van Dilk et al., 1996 and 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 with out grass hedges. These systems would be narrow grass hedges planted on the contour along a hillslope. These hedges would normally use stiff-stemmed grasses to reduce overland flow velocity and promote sediment deposition. These 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 would be applicable over a wider range of flow conditions than a buffer that is intended to intercept shallow overland flow since these are designed to control concentrated flow erosion. So, while drainage area into the buffer is important, a continuous well maintained buffer edge may just as important for maximizing 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, it would likely be expected that nutrients strongly bound to sediment such as phosphorus would have reductions lower but similar to sediment reductions, but dissolved nutrients would have lower reductions and their reduction would be closely tied to infiltration. Relative to nutrient trapping buffers would likely be less effective than for sediment. Daniels and Gilliam (1996) note that even though buffers are an accepted and highly promoted practice, little quantitative data exist on their effectiveness under unconfined flow-path conditions. Most of the reported experiments were performed on plot-scale systems and, in particular, bordered plots. A potential problem with bordered plots is that the natural flow paths are disrupted since the borders will confine flow to specific areas regardless of the natural pathways, which can reduce flow velocities or accentuate (in the case of Dillaha et al., 1989) flow concentration in comparison to natural conditions. In some cases, flow that would enter the plot is prevented from doing so, and thus the discharge rate is less than in a case where the flow entering the plot was not disrupted. This reduced discharge rate has the potential to affect the results of the transport capacity of water flowing through the buffer. In addition, controlled studies may have maintained the integrity of the buffer and maximized the density of vegetation. Many field situations may lack this integrity and the actual field performance may vary as a result.

While there is the potential that the performance of unbordered, field-scale buffers will be reduced compared to controlled studies, the few studies that have investigated the 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 60-90%, runoff loads by 50-80%, and total phosphorus loads by 50%. The retention of soluble phosphorus was about 20%. The retention of ammonium-nitrogen was 20-50%, 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 runoff reduction in the grass buffer averaged 56-72%, 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. (2005) found an average sediment trapping efficiency of 80%.

A significant unknown relative to the performance of buffers are 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 they 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 that 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 41-99% for the gross area. Helmers et al. (2005a) studied overland flow through a buffer that had been previously graded for surface irrigation and found there was convergence of overland flow even under fairly ideal conditions for overland flow. While they concluded this convergence of flow had little impact on the sediment trapping capability of the

buffer, their study highlighted that it is unlikely that uniformly distributed overland flow exists in the buffer.

While there are likely water-quality benefits of buffers even under concentrated flow conditions, Dosskey et al. (2002) demonstrated that buffer performance is significantly reduced where flow is concentrated. For the four farms they studied they estimated from modeling that buffers could potentially remove 99%, 67%, 59%, and 41% of sediment from field runoff if the flow was uniformly distributed over the entire buffer. However, because of non-uniform flow conditions resulting in less area of the buffer intercepting overland flow they predicted only 43%, 15%, 23%, and 34% of incoming sediment would be removed. Helmers et al. (2005b) found through modeling sediment trapping in a buffer that as the convergence of overland flow increases, sediment-trapping efficiency is reduced. 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 could be to use vegetative barriers on the upslope edge of buffers to distribute flow. This concentration of flow in addition to increasing the flow rate in portions of the buffer that receive runoff would be expected adversely affect the overall infiltration and soluble pollutant trapping of the system.

The discussion above has concentrated on conditions where buffers are designed to function without submergence of the vegetation. In contrast, grassed waterways are designed to function under submerged vegetative conditions. Although grassed waterways have been widely used as part of conservation systems, only a 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 600-meter-long grassed waterway with a width of 10 meters reduced the overall volume of runoff by 5%, peak runoff rates by 54%, and sediment yield by 72%. Fiener and Auerswald (2003) conducted a study in Germany, investigating the performance of a 650-meter-long grassed waterway with a width from 10 to 50 meters although the specific flow rates through their system were similar to expected flow rates in many systems where unsubmerged flow may occur. The grassed waterway was divided into two parts, one where mowing occurred and one unmanaged. The unmanaged waterway reduced runoff by 90% and sediment delivery by 97%, while the mowed waterway reduced runoff by 10% and sediment delivery by 77%. They attributed the difference in performance not entirely to the maintenance of the waterway but to the fact that the unmanaged waterway had longer side slopes and a flat bottom.

Another important contribution 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 the allowable velocity can be increased and vegetating the waterway is one form of protection (Haan et al., 1994). In many areas, reducing ephemeral gully erosion could have a significant impact on water quality. USDA (1996) reports that based on recent studies in 19 states, ephemeral gully erosion as a percentage of sheet and rill erosion ranged from 21% to 275%. So, being able to reduce this 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.

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 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 comes from naturally occurring riparian forests. Denitrification has been measured in riparian and swamp forests in at least 18 different studies mostly in temperate region. 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 30 to 100 kg N ha⁻¹ yr⁻¹ are not uncommon for these studies but very low rates in the 1-5 kg N ha⁻¹ yr⁻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. N removal through vegetation assimilation is clearly important (Lowrance et al., 1984) but requires active management of vegetation to maintain assimilation rates.

The capacity of vegetative 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 efficiencies was found to vary between 25 and 100 percent, with mean nitrate-removal efficiencies ranging from 48 to 85 percent in shallow groundwater under re-established riparian buffers. 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. 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 groundwater quality impacts of future buffers in the watershed.

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) uses terrain-analyses techniques for development of best-management-practice placement strategies. One practice they investigated was the placement of 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 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 timeline will be dependent 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 not only establish a vigorous and dense stand of vegetation but to maintain the vegetative stand. So integrity of the buffer system is likely more important than age in evaluating the effectiveness of the system. So, some of the benefits could be observed in what may be considered a relatively short time frame.

The National Conservation Buffer Initiative had a goal of two million miles installed on private land by 2002. Santhi et al. (2001) studied the economic and environmental benefits of this goal and doubling this goal. This analysis likely did not consider the overall impacts of concentrated flow on the performance of buffer systems. However, their national estimated reduction in sediment loss, total nitrogen loss, and total phosphorus loss was 15.6%, 10.8%, and 11.7%, respectively, when considering the 2 million mile goal. When the goal was doubled to 4 million miles the national estimated reduction in sediment loss, total nitrogen loss, and total phosphorus loss was 28.9%, 27.2%, and 25.3%, respectively. While there are significant assumptions in developing these values, this gives an order of magnitude impact buffer systems might have if 2 million miles or 4 million miles of buffers were installed.

Research has shown buffers to provide water quality benefits but that there is a significant range in the performance of the systems. This performance will depend 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 on the in-field performance of these systems. While it is expected that there would still be significant water quality benefits under these field conditions it is likely that the performance would be reduced compared to the results from the 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 a 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 would be to ensure buffer edges with dense vegetation that can tend to distribute flow.

From the review of the literature relative to grassed waterways it is apparent that there have been only a few studies that 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 does show 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, though. 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 mechanism. 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 there is a direct environmental benefit through the reduction in gully erosion with the use of grassed waterways provided the waterway is maintained so there is not short-circuiting of flow along the edge of the grassed waterway.

Limitations

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.

Another area that may be in need of future studies to quantify what percent of shallow groundwater moving to a particular stream interacts with the buffer zone. One specific landscape in which this might be important is where there is an extensive subsurface tile drainage system that would short-circuit 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 greatly reduced and this 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 prevents overland flow from uniformly entering the stream. Rather the overland flow may flow to a low area and then enters the drainage way through this pathway thereby reducing contact with the buffer and the effectiveness of the system. This should be considered when designing the buffer system.

The costs associated with this practice are through land being taken out of production. In some instances this could be productive farmland. As such, there is some negative attitude toward installation of these systems. However, there is not expected to be a yield reduction in the remainder

of the agricultural land. 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 10-yr evaluation horizon. They considered the private costs to be associated with land opportunity cost and buffer installation cost. From this, the annualized cost of the buffer was \$154.18/ha and the annualized benefit to be \$181.20 where the annualized benefit 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 vegetative filter strips reduced sediment yield from 10.1 t ha⁻¹ yr⁻¹ to 8.3 t ha⁻¹ yr⁻¹ (18% reduction). The approximate cost of sediment reduction for this tillage condition was \$10 t⁻¹. When no-till was considered the reduction in sediment yield due to vegetative filter strips was from 5.0 t ha⁻¹ yr⁻¹ to 3.7 t ha⁻¹ yr⁻¹ (26% reduction) and the cost of sediment reduction was \$13 t⁻¹. This type of work highlights the need for establishing what the environmental benefits of these systems are on a field-scale so that science may be able to help provide a basis for such “green” payments.

Santhi et al. (2001) evaluated the estimated annual economic impacts of implementing the National Conservation Buffer Initiative goal of 2 million miles as well as doubling this to 4 million miles. Their total net cost of the buffers considered the U.S. consumers loss from reduced supply, program payments to landowners, federal technical assistance cost, and the 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, they estimated that the annual net cost of the 2 million mile buffer goal was \$793 million and the value of water quality improvements was \$3288 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 \$1302 million and the return from water quality improvements was estimated to be \$5650 million for a benefit cost ratio of 4.3. They concluded their analyses showed the buffer programs to be cost-effective.

Important Factors

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. Munoz-Carpena et al. (1999) performed a sensitivity analysis using the model VFSSMOD (Munoz-Carpena and Parsons, 2000), which is a well-validated model for buffer performance. They found that the most sensitive parameters for the hydrologic processes in a buffer were initial soil water content and vertical saturated hydraulic conductivity. For the sediment component they found the most sensitive parameters were the sediment characteristics (particle size, fall velocity, and sediment density) as well as the grass spacing, which affects the resistance to overland flow. Their work thus highlights the importance of having a dense stand of vegetation to maximize the pollutant-trapping capacity of the buffer.

While VFSSMOD provides a useful tool for evaluating buffer performance under many conditions, it should be noted that it may be conservative in predicting the effect of narrow buffers including

vegetative barriers since these systems rely on backwater effects that are not modeled with VFSSMOD. Therefore, its predictions of trapping efficiency for narrow barriers can significantly underestimate the trapping and potential infiltration of these types of systems.

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 through direct 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, and 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 to the system will also impact the performance of the system. Some of the variables that would 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. Also, they found that as the storm size increased the performance of the buffer from a sediment trapping efficiency perspective decreased. Both of these factors (slope and storm size) influenced the loading, including the flow rate, to the buffer so as the loading 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, this system would need to be designed to effectively convey water off the landscape while minimizing channel instability so the hydrology of the site and the soils in particular 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 is also some runoff reduction and direct water quality benefits of the grassed waterway. The reduction in runoff will likely be greater under smaller storm and runoff conditions when the specific flow rate in the grassed waterway is the range commonly expected in other buffer systems. Under larger precipitation events the grassed waterway will likely function just in a water conveyance capacity.

Summary

1. Buffers and grassed waterways are broadly accepted practices for reducing nutrient runoff from agricultural fields.
2. 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.
3. Impact will be much lower if not properly located designed, or maintained. In addition, as flow rate of water entering the buffer is increased the percent effectiveness relative to the trapping of sediment and nutrients are expected to decrease so in-field management is important to maximize the effectiveness of the systems. In addition, the storm size would greatly affect the flow rate into the systems so the performance will vary depending on the storm size.

4. Costs are mainly land out of production with benefits from water quality and additional ecological benefits (e.g., wildlife)
5. There are research needs for better predictions (more accurate, less uncertainty) of overall field to watershed level effects of buffers and grassed waterways based on studies that examine channel/gully processes, gully control methods, field-scale performance of the various buffer systems, improved models for dissolved pollutant trapping and delivery to streams, and cost-effectiveness of these systems based on field-scale studies.

References

- Arora, K., S. K. Mickelson, J. L. Baker, and D. P. Tierney. 1993. Evaluating herbicide removal by buffer strips under natural rainfall. ASAE Meeting Paper No. 93-2593. 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. *Soil Science Society of America Journal* 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 Technology* 13(1): 157-164.
- Clausen, J. C., K. Guillard, C. M. Sigmund, and K. Martin Dors. 2000. Water quality changes from riparian buffer restoration in Connecticut. *Journal of Environmental Quality* 29: 1751-1761.
- Correll, D. L., T. E. Jordan, and D. E. Weller. 1997. Failure of agricultural riparian buffers to protect surface waters from groundwater contamination. In *Groundwater/Surface Water Ecotones: Biological and Hydrological Interactions and Management Options*, eds. J. Gibert and others. pp. 162-165. Cambridge University Press, Cambridge U.K.
- 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. *Journal of Soil and Water Conservation* 53(2): 140-145.
- 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. *Journal of Soil and Water Conservation* 50(4): 405-408.

- Dabney, S. M. 2003. Erosion Control, Vegetative. In: Encyclopedia of Soil Science, Rattan Lal (Editor) pp. 209-213. Marcel Dekker, New York, NY.
- Daniels, R. B., and J. W. Gilliam. 1996. Sediment and chemical load reduction by grass and riparian filters. *Soil Science Society of America Journal* 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. *Environmental Management* 28(5): 577-598.
- Dosskey, M. G. 2002. Setting priorities for research on pollution reduction function of agricultural buffers. *Environmental Management* 30(5): 641-650.
- Dosskey, M. G., M. J. Helmers, D. E. Eisenhauer, T. G. Franti, and K. D. Hoagland. 2002. Assessment of concentrated flow through riparian buffers. *Journal of 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. *Journal of 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. *Journal of Environmental Quality* 32(3): 927-936.
- Fiener, P., and K. Auerswald. 2005. Measurement and modeling of concentrated runoff in grassed waterways. *Journal of Hydrology* 301(1-4): 198-215.
- Gilley, J.E., B. Eghball, L.A. Kramer, and T.B. Moorman. 2000. Narrow grass hedge effects on runoff and soil loss. *Journal of 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. *Journal of Environmental Quality* 21: 666-671.
- Haan, C. T., B. J. Barfield, and J. C. Hayes. 1994. *Design Hydrology and Sedimentology for Small Catchments*, Academic Press, San Diego, CA.
- Hall, J. K., N. L. Hartwig, and L. D. Hoffman. 1983. Application mode and alternate cropping effects on atrazine losses from a hillside. *Journal of Environmental 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 Meeting Paper No. 83-2081. St. Joseph, Mich. ASAE.
- Helmers, M. J., D. E. Eisenhauer, M. G. Dosskey, and T. G. Franti. 2002. Modeling vegetative filter performance with VFSSMOD. ASAE Meeting Paper No. MC02-308.
- 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. of 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. of ASAE* 48(2): 541-555.
- Hill, A.R. 1996. Nitrate removal in stream riparian zones. *Journal of Environmental Quality* 25: 743-755.
- Hjelmfelt, A., and M. Wang. 1999. Modeling hydrologic and water quality responses to grass waterways. *Journal of Hydrologic Engineering* 4(3): 251-256.
- Jordan, T. E., D. L. Correll, and D. E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. *Journal of Environmental Quality* 14: 472-478.
- Jordan, T.E., D.L. Correll, and D.E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from cropland. *Journal of Environmental Quality* 22: 467-473.
- Lee, K. H., T. M. Isenhardt, R. C. Schultz, and S. K. Mickelson. 2000. Multispecies riparian buffers trap sediment and nutrients during rainfall simulations. *Journal of Environmental Quality* 29(4): 1200-1205.
- Lee, K.H., T.M. Isenhardt and R.C. Schultz. 2003. Sediment and nutrient removal in an established multi-species riparian buffer. *Journal of 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. and J. E. Parsons. 2000. VFSMOD, Vol. 1.04 User's manual. Raleigh, N.C.: North Carolina State University.
- Munoz-Carpena, R., J. E. Parsons, and J. W. Gilliam. 1999. Modeling hydrology and sediment transport in vegetative filter strips. *Journal of Hydrology* 214(1/4): 111-129.
- Natural Resources Conservation Service (NRCS). 1999. Filter strip, national standard No. 393. Washington, D.C.: U.S. Department of Agriculture.
- Natural Resources Conservation Service (NRCS). 2000. Grassed waterway, national standard No. 412. Washington, D.C.: U.S. Department of Agriculture.
- 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 *Environmentally Sound Agriculture, Proceedings of the 2nd Conference*, ed. K. L. Campbell, W. D. Graham, and A. B. Bottcher, pp. 147-154. St. Joseph, Mich: ASAE.
- 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 Meeting Paper No. 90-2501. 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. *Environmental 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. U.S. Department of Agriculture, Agric. Econ. Report No. 782.
- 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 Meeting Paper No. 01-2068. 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. *Journal of Environmental 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 Technology* 45(5): 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. *Journal of Soil and Water Conservation* 58(4): 198-206.
- U.S. Department of Agriculture. 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. *Journal of Environmental 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. *Journal of Soil and Water Conservation* 57(5): 259-267.

Nitrogen Rates

John E. Sawyer, Associate Professor and Soil Fertility Extension Specialist, Department of Agronomy, Iowa State University

Gyles W. Randall, Soil Scientist and Professor, Southern Research and Outreach Center, University of Minnesota

Introduction

In most crop rotations that include corn, nitrogen (N) is applied to the corn phase in order to increase yield and is a proven and profitable practice. Corn in some rotations requires little to no N application, with first-year corn following established alfalfa as an example. Corn in other rotations requires substantial application 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 yield will suffer. If N is not applied on an on-going basis, over time corn yield will often average around 50-60 bu/acre in CC and 100-110 bu/acre in SC, or less.

The soil system typically cannot supply the full corn plant N requirement. On average the yield with no N applied as a fraction of the yield at an economic optimum rate is around 70% in a SC rotation and 55% in CC. Therefore, supplemental N is needed to reach economic yield potential. Research has been on-going for over 50 years measuring corn response to N application. Guidelines for suggested N rates based on that research have been derived using economic principles to determine 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, in corn production systems nitrate losses occur even when no N is applied, and N application at optimum rates increases loss. These levels of nitrate loss may be unacceptable. 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 paper are to review the effect of N application rate to corn on economic return, nitrate in subsurface drainage (tile flow), and potential nitrate reduction.

Economic N Application Rates

Rates that producers should apply are not those to produce maximum yield, but instead profitable economic yield where the yield gain from N application will more than pay for the invested N. Nitrogen response trials are conducted where multiple rates of N are applied and grain yield measured at each rate. Analysis of that response data allows calculation of site EONR, the rate where the grain yield increase just pays for the cost of the last increment of applied N (example in Fig. 1). Economic net return is the 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. 2) and locations (Fig. 3) due to non-controllable factors and to improve determination of the point where expected maximum return to N (MRTN) occurs (MRTN approach as described by Nafziger et al., 2004). The MRTN approach incorporates the uncertainty in yield response to applied N, uses the diminishing yield increase as N rate increases for all sites, and provides the point where the economic net return is maximized across all sites (the closed symbols in Fig. 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 (the open symbols in Fig. 4). This range can provide flexibility in decisions regarding application rate and should provide adequate yield across changing production conditions. Also, 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 soil for potential loss than at the high end of the range. However, the risk of having inadequate N increases.

When N response trials are run with corn in different rotations, then the MRTN can be calculated for each rotation. Examples are given in Fig. 4 for CC (56 sites) and SC (121 sites) in Iowa for trials conducted approximately the past 10 years. In these Iowa trials, the MRTN rate for CC is approximately 175 lb N/acre and 125 lb N/acre for SC when the ratio of the N price to corn price is 0.10 (\$0.22/lb N:\$2.20/bu). 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. Fig. 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 by modifying rate.

Economic N rates also change with different relationships between N price and corn price (i.e. the N:corn price ratio, \$/lb:\$/bu). As shown in Fig. 4, as the N price becomes higher relative to the corn price (i.e. the ratio gets larger), the net return and MRTN rate decreases. Also, the economic penalty to high N rates increases (N rates above the MRTN) as evidenced by the steeper decline in net return as 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. 4), as evidenced by the rapid decline in net return as N rate falls below the maximum profit range. This is likely the greatest concern for producers, increased production risk and associated severe yield and economic loss with deficient N. Incentives for producers to take on increased risk as rates are used at the lower end of the MRTN range could be provided by documentation of N adequacy or deficiency with tools such as soil N tests or plant sensors. Another approach would be to insure producers against this risk.

For sound N management, crop producers should apply the rate of N that provides maximum return to the N investment. This application, however, does result in increased soil nitrate with potential then for higher 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 which 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 soybean. This concentration or load varies depending upon climate, soil properties and tile system characteristics, but often spans the range of 3 to 10 mg/L or 8 to 20 lb/acre. As N is applied at increasing rate the concentration and load of nitrate-N in tile flow increases; examples shown in Tables 1 and 2 and in Figs. 6 and 7, with further examples in Baker et al. (1975); Baker and Johnson (1981); Davis et al. (2000); Jaynes et al. (2001); Kladvivko et al. (2004); Jaynes et al. (2004); Clover (2005); 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, 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 for rivers and streams or 0.78 mg Total-N/L 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. 6 and 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 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 by Baker et al. (1975), N applied only to corn at a rate of 100 lb N/acre in a Oat-Corn-Oat-Corn-Soybean sequence resulted in an average 21 mg nitrate-N/L in tile flow (site located at Boone, IA). Continuing research at the site (Baker and Johnson, 1981) with two N rates of approximately 90 lb N/acre and 240 lb N/acre applied only to corn in a Corn-Soybean-Corn-Oat-Soybean sequence resulted in an average 20 mg nitrate-N/L (24 lb nitrate-N/acre) with the low N rate and 40 mg nitrate-N/L (43 lb nitrate-N/acre) 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 inches) was 18 mg/L at the EONR, was <10 mg/L when N rates were more than 45 lb N/acre below the EONR, and >20 mg/L when N rates were more than 45 lb N/acre above the EONR. Work reported by Randall and Mulla (2001) with depleted ¹⁵N 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/acre compared to 120 lb N/acre. Davis et al. (2000) reported that increasing N rates from 90 to 200 lb N/acre in CC (Waseca, Minnesota) resulted in a linear increase in nitrate-N loss (0.8 to 22.8 lb nitrate-N/acre). Jaynes et al. (2004) achieved a 30% reduction in nitrate-N concentration in water leaving a central Iowa subbasin 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 drinking water MCL.

If achieving the drinking water standard is a goal for nitrate in subsurface drainage, it will be difficult solely with application rate. However, if N is being applied well above rates that produce maximum economic return, reduction in nitrate loss can be accomplished by reducing rates to those levels (examples in Table 1 and Figs. 6 and 7). The gain will be dependent upon the specific location, rate change, and production situation. Continued research on development and refinement of tools such as soil N testing, plant testing, and plant N stress sensing can help adjust N application toward economic rates, and help producers manage production risk from inadequate N when rates are adjusted toward the lower end of profitable N rate ranges.

Nitrate-N concentrations in subsurface drainage are generally greater for CC compared with 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/acre/year in a CC system provides greater potential for much higher nitrate losses than an over-application of 50 lb N/acre 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. Also, nitrate does move in drainage water after soybean harvest, thus influencing nitrate loss. Data from the Nashua, IA water quality site for 1990-1992 provides an excellent example. The average loss (across all tillage systems) was 30 mg nitrate-N/L (52 lb nitrate-N/acre) with CC and 18 mg nitrate-N/L (25 lb nitrate-N/acre) with SC at N rates of 180 lb N/acre applied each year to corn in CC and 150 lb N/acre 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/acre in CC and 100 lb N/acre in SC, the average loss was 11 mg nitrate-N/L (15 lb nitrate-N/acre) with CC and 10 mg nitrate-N/L (12 lb nitrate-N/acre) with SC. Another example is tile-flow data collected by Randall et al. (1997) where N (based on spring soil sampling) applied in CC compared to SC increased average annual nitrate-N concentrations approximately 8 mg/L (from 24 to 32 mg/L) and increased flux 7%.

While not directly comparing N rates, at a site in southeastern Indiana Kladvko 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 lb N/acre to 160 lb N/acre), 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 mg/L to 8 mg/L. Important characteristics that influenced nitrate-N concentrations and changes over time at this 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 a N rate of 300 lb N/acre to SC with a N rate of 200 lb N/acre and a winter cover crop, annual flow-weighted nitrate-N concentrations were reduced from about 22 mg/L to 12 mg/L.

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

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 MRTN optimized.

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. Also, 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 1 and Fig. 6). Also, 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. 6 and 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 1 and 2 and Figs. 6 and 7). As an example (Fig. 6), let's say the goal is to reduce tile-flow nitrate-N to 10 mg/L and the starting N rate is at the MRTN. At the MRTN rate for Iowa SC (125 lb N/acre) the associated tile-flow nitrate-N is approximately 12 mg/L (Lawlor et al., 2005). The N rate associated with 10 mg nitrate-N/L is 85 lb N/acre. The net economic loss due to an N rate reduction from 125 to 85 lb N/acre is $-\$5.85/\text{acre}$. With another example where corn yield and tile-flow nitrate is more responsive to N application (Fig. 7), moving from the site EONR of 190 lb N/acre to 120 lb N/acre (an associated 30% reduction in tile-flow nitrate load from 61 to 42 lb nitrate-N/acre) the net economic loss is $-\$27.15/\text{acre}$.

Since yield response to increasing N rate is curvilinear (follows diminishing return), 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 rates decline below the MRTN and approach zero (Table 2 and Fig. 7). For the Filson, IL site, the first 70-lb N rate increment below the site EONR costs \$0.52 per unit of nitrate-N load reduction, but the last 70-lb N rate increment at zero N costs \$29.70 per unit of nitrate-N load reduction (Table 2).

These examples illustrate the significant risk and economic constraints that face producers if asked to reduce N application to rates below maximum net return. If N rates in both examples given above were reduced to zero, the economic losses would be $-\$81.75$ and $-\$200.10/\text{acre}$. 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 to corn results in greater nitrate in subsurface drainage water. While rates that produce maximum net economic gain through yield return to N will moderate nitrate-N, 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 through change in N rate 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 greater reduction in corn grain yield than offset by N input reduction. If such rate restrictions are placed on N application 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 cover lost producer income. It is also important to recognize that N need, 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 changes in N inputs.

Key Points

- In corn production systems nitrate is lost in tile-flow drainage even if no fertilizer N is applied, often in the 3-10 mg nitrate-N/L range. These concentrations exceed the currently proposed EPA nutrient ecoregion VI surface water quality criteria for total N.
- Nitrate-N concentration in subsurface drainage generally increases in a continuous relationship with increasing N rate. More N rate research where nitrate loss is concurrently measured is needed to better quantify that relationship.
- Application of N near rates that provide maximum economic return to N usually results in tile flow having nitrate-N concentrations above the EPA drinking water MCL, often in the range of 10-20 mg nitrate-N/L for soybean-corn and 15-30 mg nitrate-N/L for continuous corn.
- Application of N above optimal rates reduces economic return and further increases nitrate losses.
- 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.
- Nitrogen rate reduction will only directly benefit producers when current application rates are above optimum. Reduction to optimal rates will also improve nitrate losses.
- Because of uncertainty in actual N fertilization need for any given year and location,

producer risk increases with rates at the lower end of profitable N rate ranges.

- Nitrogen rate reductions below optimum result in economic losses to the producer because the value of yield losses are greater than offset by reduced N costs.
- In Iowa studies, to lower the nitrate concentration to 10 mg nitrate-N/L in tile drainage with a soybean-corn rotation, the N rate applied to corn had to be reduced 40 lb N/acre below the rate providing maximum economic return; this reduction would have an associated net loss of \$5.85/acre.
- In an Illinois study with a soybean-corn 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 70 lb N/acre below the economic optimum rate with an associated net loss of \$27.15/acre.
- As N rate reductions increase below the optimum, the cost in yield loss per unit of nitrate-N reduction becomes much larger.
- Cropping rotations that include fewer years with corn consequently reduce N rate and frequency of application, and result in lower nitrate in subsurface drainage.
- Optimal N rates for corn, associated nitrate levels in subsurface drainage, and thus potential to effect improvement in nitrate losses through N rate change varies across the Upper Mississippi River Sub-basin and needs to be accounted for in water quality programs addressing N application.
- 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 attempting to improve determination of needed N fertilization and thus reduce risk of N deficiency.
- To achieve desired water quality goals, other in-field or out-of-field practices will need to be implemented in addition to N rate management for corn production.

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.
- Clover, M. 2005. M.S. thesis, Univ. of Illinois, Urbana, IL.
- 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, MI, ASAE.
- Nafziger, E.D., J.E. Sawyer, and R.G. Hoelt. 2004. Formulating N recommendations for corn in the corn belt using recent data. In *Proc. N. C. Ext.-Ind. Soil Fertility Conf.*, Des Moines, IA. 17-18 Nov., 2004. Potash and Phosphate Inst., Brookings SD.
- Owens, L.B., W.M. Edwards, and M.J. Shipitalo. 1995. Nitrate leaching through lysimeters in a corn-soybean rotation. *Soil Sci. Soc. Am. 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.
- United States Environmental Protection Agency. 2002. Nutrient water quality criteria, summary table for the nutrient criteria documents [Online]. Available at <http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/> (verified 12 August 2005).
- 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.

Table 1. Corn production and nitrate loss to tile drainage as affected by rate and time of N application at Waseca, 2000-2003 (Gyles Randall, Univ. of Minnesota, personal communication).

Time	N Treatment		Grain Yield bu/acre	4-Yr Average	
	Rate lb N/acre	N-Serve		Net Return to N \$/acre	Flow-Weighted NO ₃ -N Conc. ^{1/} mg/L
--	0	--	111	--	--
Fall	80	Yes	144	38	11.5
"	120	"	166	72	13.2
"	160	"	172	74	18.1
Spr.	120	No	180	105	13.7

^{1/} Across four SC rotation cycles.

Table 2. Corn production and nitrate loss to tile drainage as affected by spring-applied anhydrous ammonia N rate at Filson, IL, 2002-2004 (M. Clover, M.S. Thesis, Univ. of Illinois).

N Rate lb N/acre	Grain Yield bu/acre	Tile-Flow NO ₃ -N ^{1/} lb/acre	Change Per 70-lb N Rate Increment			
			Yield bu/acre	NO ₃ -N lb/acre	Net Loss ^{2/} \$/acre	Net Loss Per Unit NO ₃ -N \$/lb
210	180	61.1	---	---	---	---
140	169	41.2	11.7	19.9	10.34	0.52
70	130	30.9	38.1	10.3	68.42	6.64
0	69	26.9	61.0	4.0	118.80	29.70

^{1/} Rotation total from the average across three years of each crop in a SC rotation, i.e. the total amount for the 2-yr rotation.

^{2/} Nitrogen at \$0.22/lb N and corn grain at \$2.20/bu.

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

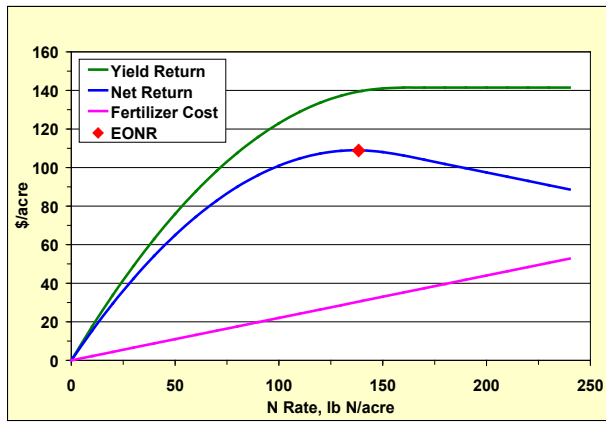


Figure 2. Variation in EONR (0.10 price ratio) and yield across years for SC and CC at the same site location, Ames, Iowa.

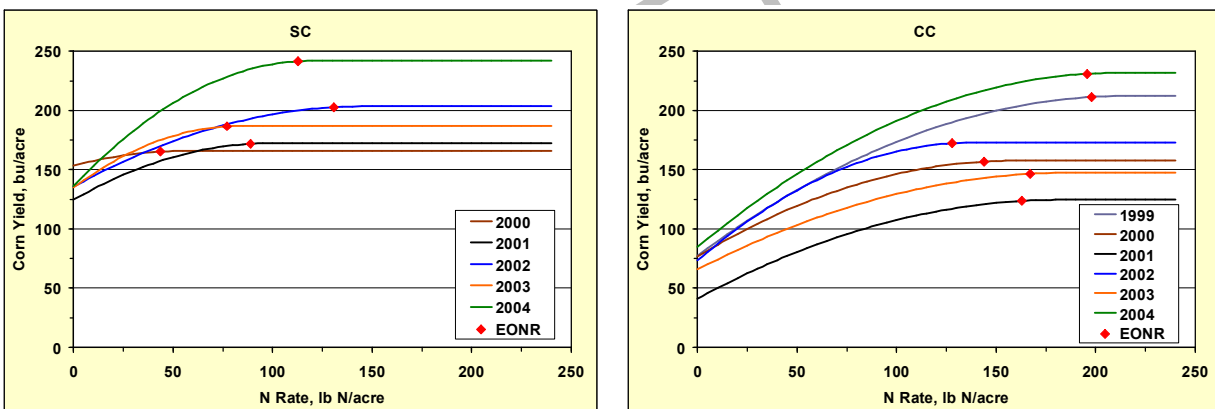


Figure 3. Example of variation in response to N and EONR (0.10 price ratio) at different locations in Iowa.

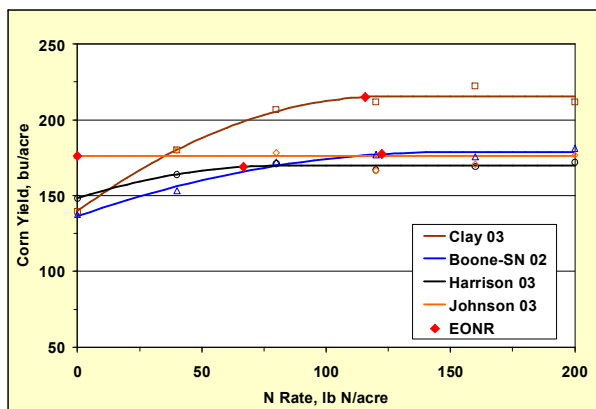


Figure 4. Effect of fertilizer N:corn grain price ratio on net return to N (SC and CC rotations in Iowa). The closed symbols correspond to the maximum return to N (MRTN) and open symbols the range around the MRTN with similar return.

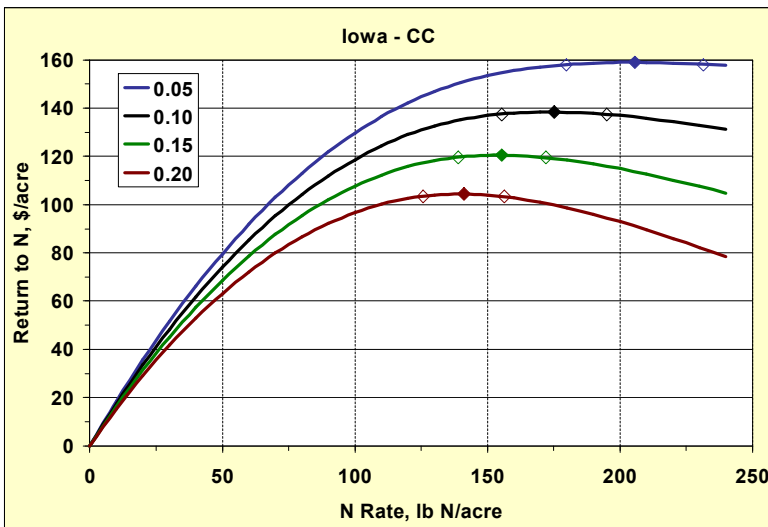
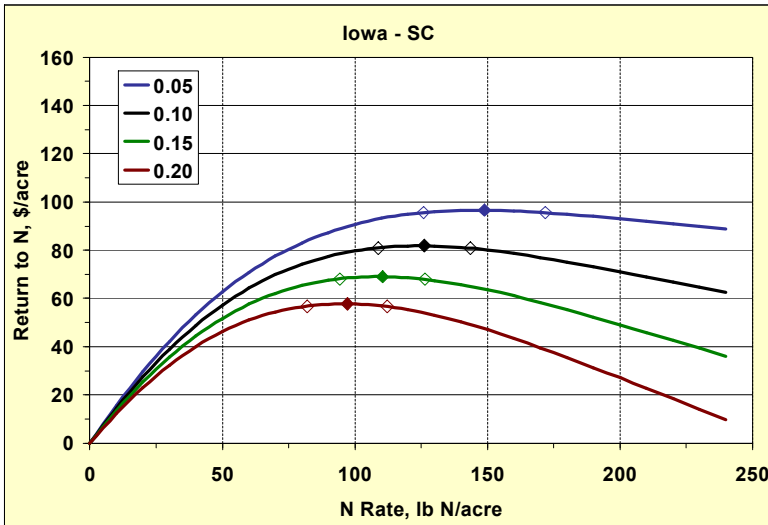


Figure 5. Differences between net return to N for SC and CC for various states at a 0.10 N:corn price ratio (\$0.22/lb N:\$2.20/bu corn). The closed symbols correspond to the maximum return to N (MRTN) and open symbols the range around the MRTN with similar return. Response data courtesy of Emerson Nafziger, Univ. of Illinois and Gyles Randall, Univ. of Minnesota.

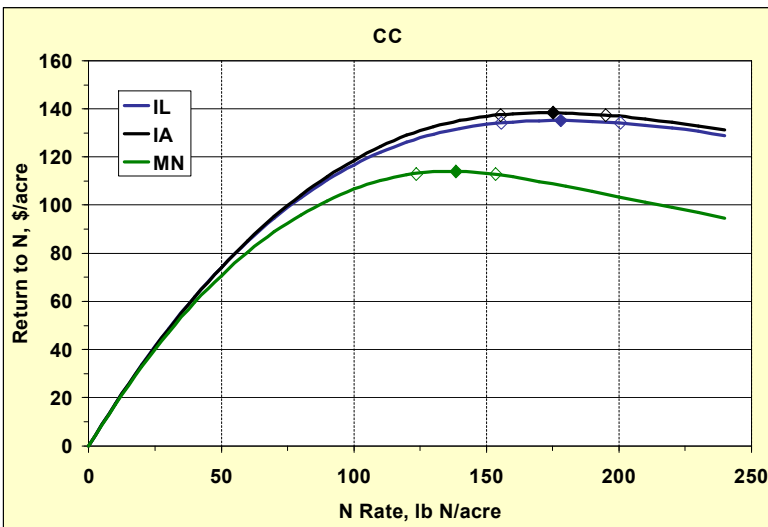
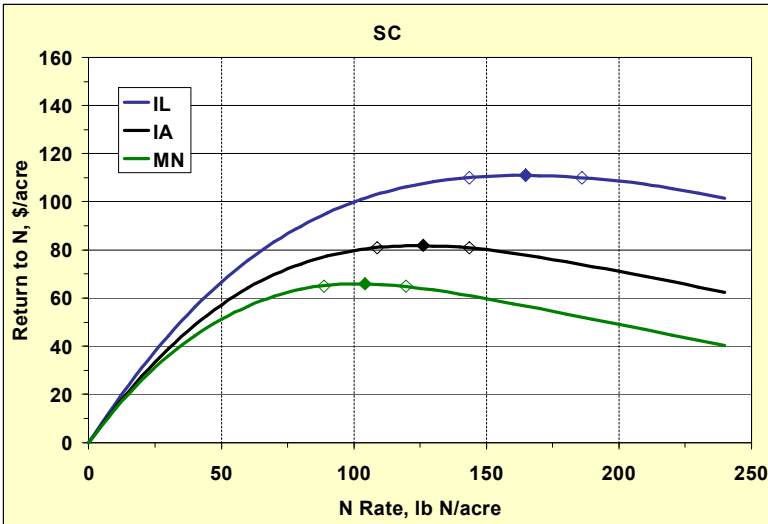


Figure 6. Tile - flow nitrate - N concentration average in a SC rotation from N rates applied in various years from 1990-2004 at the Gilmore City, IA site (Lawlor et al., 2005) and the net economic gain or loss (\$0.22/lb N:\$2.20/bu corn) across N rates for SC in Iowa (Nafziger et al., 2004). The solid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

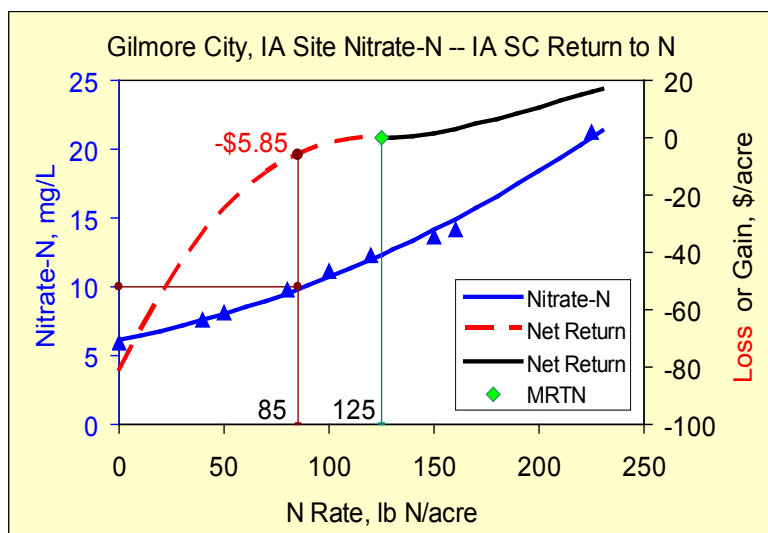
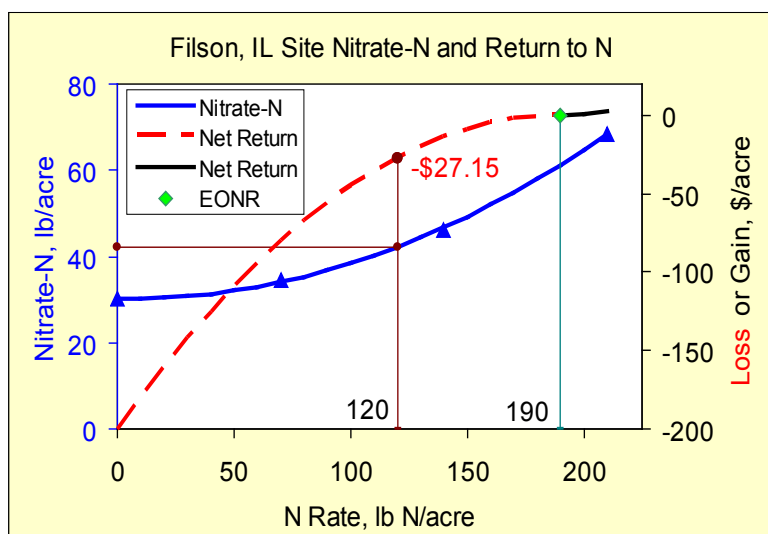


Figure 7. Rotation total tile-flow nitrate - N mass load and net economic gain or loss (\$0.22/lb N:\$2.20/bu corn) across spring-applied N rates in a SC rotation, average of 2002-2004 at the Filson, IL site (M. Clover, M.S. Thesis, Univ. of Illinois). The solid black section of the net return line represents the gain if N rates are reduced to the site economic optimum N rate (EONR), and the red dashed section represents the loss if N rates are reduced below the EONR. The indicated economic loss 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.



DRAFT

Nitrogen Application Timing, Forms and Additives

Gyles Randall, Soil Scientist and Professor, Southern Research and Outreach Center, University of Minnesota

John Sawyer, Associate Professor and Soil Fertility Extension Specialist, Dept. of Agronomy, Iowa State University

Introduction

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. Crop production is improved due to better physical conditions for field operations and a deeper unrestricted root zone for greater crop rooting and yields. However, subsurface, tile drainage in agricultural production systems has been identified as a major source of nitrate entering surface waters.

Factors influencing the nitrate content in subsurface waters draining from agricultural production landscapes can be divided into two categories - - noncontrollable and controllable. Precipitation, including variations in annual amount, temporal distribution within a year, or extreme daily events, is a noncontrollable factor having the greatest impact on nitrate loss.

Controllable factors are those management practices that crop producers use to improve yield and profitability of their enterprise. Time of N application, N form/source, 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 speaking, 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 time and field conditions for application are better. Early planting of corn as soon as the soils are fit in the spring is desirable for highest yields and profit. Consequently, the window of opportunity for spring N application becomes very narrow (Randall and Schmitt, 1998). Soil compaction can also be a deterrent 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 amounts of nitrate lost by denitrification or leaching during spring rainfall. They suggested that economic and environmental benefits of delaying application of fertilizer N maybe 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). Corn yields from the late fall application averaged 8% lower (146 vs. 159 bu/A/yr) than with spring application (Table 1). Moreover, annual losses of nitrate-N in the tile drainage water averaged 36% higher (30 vs. 22 lb/A/yr) 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/A) and net return (\$54/A) were obtained for the spring treatment.

A long-term corn-soybean rotation study comparing late-October application of ammonia with and without N-Serve with a spring preplant application without N-Serve showed distinct yield, economic, and environmental advantages for spring application, but not in all years (Table 2). Across the 15-yr period, corn yields averaged about 10 bu/A greater for the fall N + N-Serve and spring N treatments compared with fall N without N-Serve (Randall et al., 2003b; Randall and Vetsch, 2005b). Also, compared with fall application of N without N-Serve, $\text{NO}_3\text{-N}$ losses in the drainage water were reduced by 14 and 15% (Randall et al., 2003a; Randall and Vetsch, 2005b), economic return to N was increased by \$9 and \$19/A, 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 7 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/A and average economic return was increased by \$22.50 and \$51.00/A for the fall N + N-Serve and spring N treatments, respectively. In summary, the 15-yr data suggest that applications of ammonia in the late fall + N-Serve or in the spring preplant were BMPs. 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 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 1997-99 (Vetsch and Randall, 2004). Yields were not different between fall and spring-applied N in 1997 or 1998 (Table 3). The effect of wet spring conditions was evident in 1999 when corn yields were 36 bu/A lower for fall-applied N. An interaction between tillage system and time/placement of N was not found, indicating 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 $\text{NO}_3\text{-N}$ concentrations in subsurface drainage from a corn-soybean rotation to not be different between fall and spring applications of aqua ammonia with and without N-Serve under slightly dry to normal precipitation conditions (Lawlor et al., 2004) (Table 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. Evidence in the literature to support this concept is mixed, however. Baker and Melvin (1994) reported losses of

nitrate-N to be higher for split application compared to a preplant application for 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 following year when soybeans 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.

A split application of ammonia with 40% applied preplant and 60% applied sidedress at the V8 stage was compared with late October and spring preplant applications of ammonia (Table 5). In this 7-yr period, grain yields were significantly greater (6 bu/A) 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 $\text{NO}_3\text{-N}$ concentration in the tile drainage across the 4-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).

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. Nitrapyrin (N-Serve) has been the most commonly used nitrification inhibitor in the U.S. and has been a component in many N research studies. 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 (Hoelt, 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 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 4-yr corn-soybean rotation flow period (1990-93) averaged 16.8, 13.7, 13.7, and 14.6 mg L^{-1} for the four treatments, respectively (Table 5). Yields were increased significantly in the very wet years by the addition of N-Serve to the fall application.

A 6-yr study comparing fall vs. spring application of N-Serve with ammonia showed a statistically and economically significant 10 bu/A yield response to N-Serve applied in the fall (Table 6). The 4 bu/A yield increase to spring-applied N-Serve was not statistically significant and is 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 can not be predicted at the time of spring ammonia application, adding N-Serve to spring-applied ammonia is not considered to be a BMP.

The interaction between time of N application and N-Serve in the above study was significant for $\text{NO}_3\text{-N}$ concentration in the drainage water in three of six years during the corn phase and two of six years during the soybean phase. Annual $\text{NO}_3\text{-N}$ concentrations were reduced 2 to 4 mg/L when

N-Serve was added to fall-applied N but were increased 1 to 3 mg/L when N-Serve was added to spring-applied N. These increased concentrations of $\text{NO}_3\text{-N}$ in the drainage water with spring-applied N-Serve are similar to the results with split-applied N (spring + sidedress) shown in Table 4.

Smicklas and Moore (1999) reported that $\text{NO}_3\text{-N}$ concentrations in subsurface drainage from six 5-acre drainage parcels were 58% greater for fall-applied N than for the same N rate applied in the spring in Illinois. Nitrate-N concentrations were also decreased by 9% with N-Serve compared to without N-Serve. N-Serve added to spring-applied urea for continuous corn in Ohio reduced nitrate losses in drainage water from lysimeters (Owens, 1987). A 3-yr drainage study in Illinois showed significant differences among fall, spring, and sidedress application of N to corn on the $\text{NO}_3\text{-N}$ concentrations and losses in corn and soybean the following years (R.G. Hoelt, personal communication, 2005). However, the addition of N-Serve to fall-applied N did not affect either $\text{NO}_3\text{-N}$ concentration or loss in the drainage water or corn yields.

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 10-yr 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 2 and 4 suggests 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 9 bu/A and economic return was increased \$9.30/A. Moreover, $\text{NO}_3\text{-N}$ losses in tile drainage water were reduced 14%. These data further suggest that N-Serve would not likely be beneficial in reducing nitrate losses to tile drainage or in boosting yields and profitability when applied with ammonia in the spring.

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, with and without N-Serve, to spring application of the same. Two-year average corn yields shown in Table 7 indicate: (a) broadcast and incorporated urea was inferior to anhydrous ammonia when fall-applied, and (b) 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 1.

A subsequent study on Nicollet and Webster glacial till soils in southern Minnesota compared late October application of urea (4" deepband) and anhydrous ammonia with and without N-Serve to spring preplant urea and anhydrous ammonia. Three-year average yields show a 33 bu/A advantage for urea and a 14 bu/A for ammonia when applied in the spring (Table 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, 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 for the 240-lb N rate were 13 bu/A greater when applied in the spring compared with the fall. In the last four years, the yield advantage for spring-applied urea was 16 bu/A. Moreover, the 160-lb spring rate yielded 10 bu/A more than the 240-lb fall rate.

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 0% under dry conditions to more than 50% under very wet conditions.
- Research findings:
 - (1) 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 seen on medium to fine-textured soils.
 - (2) 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. If using UAN, split application (preplant and sidedress) is desirable as it reduces the risk of loss when conditions are wet prior to the V10 stage. Data showing this are limited, however.
- Water quality improvement:
 - (1) Minnesota data suggest an average reduction of leaching loss up to 15% in drainage water with spring application compared to a late October, fall application when soil temperatures are at or below 50°F.
 - (2) Nitrate losses from fall-applied N throughout the Corn Belt could range between 0 to 25% depending on time of fall application (early vs. late), fall and winter soil temperatures, and spring rainfall.
- Cost:

Spring-applied N may cost up to \$5-10/A/yr more than fall N.
- Extent of area:

Percent of corn acres in the Corn Belt that could benefit from spring or split applications of N is 100% of all acres presently receiving fall N.

- Limitations for adoption of spring N
 - the current mindset or tradition of fall anhydrous ammonia among growers and suppliers will be slow to change.
 - supplier infrastructure, although this is currently changing, will cause spring supply and storage issues and will require equipment changes and \$.
- Impact on other resources
 - Incorporation of urea and UAN to limit volatilization or surface runoff losses could enhance soil erosion. (Negative)
 - Crop yields will likely become less variable and risk will be reduced. (Positive)

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:
 - (1) Minnesota data obtained on calcareous, poorly drained, glacial till soils suggest an average reduction of 14% in nitrate leaching losses 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.
 - (2) 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/A for a reduction of 3.5 lb NO₃-N N/A (range is 0 to 9 lb NO₃-N/A).
- 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.
- Limitation for adoption:
 - old chemistry and inconsistent, weather-related results
 - extra cost, \$
 - new inhibitors and controlled release forms of urea that perform satisfactorily and are inexpensive are needed
- 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.

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 highly acceptable sources of N for optimum crop production when spring-applied. Fall-applied urea has performed poorly.

- Water quality improvement

No effect on water quality would be expected among the fertilizer forms of N as long as they are applied using best management practices.

- Cost:

Costs among the fertilizer N sources will vary depending upon 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.

- Extent of area:

No limitation other than suppliers' source inventory.

- Limitations for adoptions

Two primary limitations exist. From the suppliers 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 growers and suppliers perspective, 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

None

References

- Baker, J. L., and S. W. Melvin. 1994. Chemical management, status and findings. pp. 27-60. In Agricultural Drainage Well Research and Demonstration Project – Annual Report and Project Summary. Iowa Dept. of Agric. and Land Stewardship and Iowa State Univ.

- 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. *Appl. Eng. 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. of Soil Sci.* 44:35-48.
- Hoelt, R.G. 1984. Current status of nitrification inhibitor use in U.S. agriculture. p. 561-570. In R.D. Hauck (ed.) *Nitrogen in crop production*. ASA, CSSA, and SSSA, Madison, WI.
- Kyveryga, P.M., A.M. Blackmer, J.W. Ellsworth, and R. Isla. 2004. Soil pH effects on nitrification of fall-applied anhydrous ammonia. *Soil Sci. Soc. Am. 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, MI, ASAE.
- 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. p. 274. In *Agronomy abstracts*. ASA. Madison, WI.
- 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 M. A. Schmitt. 1998. Advisability of fall-applying nitrogen. pp. 90-96. In *Proc. of the 1998 Wisconsin Fertilizer, Agrilime, and Pest Management Conf.* Middleton, WI. 20 Jan. 1998.
- 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., 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.
- Smicklas, K.D., and A.S. Moore. 1999. Fertilizer nitrogen management to optimize water quality. p. 117-124. In R.G. Hoelt (ed.) *Illinois Fertilizer Conf. Proc.*, Peoria, IL. 25-27 Jan. 1999.
- 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. of 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.

Table 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 ^u		Annual Loss of Nitrate-N in Drainage lb N/A/yr	5-Yr Yield Avg.	
Rate lb/A	Time		Yield bu/A	Net return \$/A
0	0	7	66	--
120	Fall	27	131	100
120	Spring	19	150	135
180	Fall	34	160	143
180	Spring	26	168	154

^u Ammonium sulfate applied to continuous corn about 1 Nov or 1 May.

Table 2. Corn yield and economic return to nitrogen program as affected by time of application and N-Serve at Waseca, 1987-2001 (adapted from Randall and Vetsch, 2005a, b and Randall et al. 2003a, b)^{1/}.

Parameter	Time of Application		
	Fall	Fall + N-Serve	Spring
15-Yr Avg. Yield (bu/A)	144	153	156
15-Yr Avg. Economic return over fall N (\$/A/yr) ^{2/}	--	\$9.30	\$18.80
7-Yr Avg. Yield (bu/A) ^{3/}	131	146	158
7-Yr Avg. Economic return over fall N (\$/A/yr) ^{2/}	--	\$22.50	\$51.00
15-Yr Flow-weighted NO ₃ -N concentration in tile drainage from the corn-soybean rotation (mg/L)	14.1	12.2	12.0
15-Yr Nitrogen recovery in the corn grain (%) ^{4/}	38	46	47

^{1/} Rate of N was 135 lb/A/yr for 1987-93 and 120 lb N/A/yr for 1994-2001.

^{2/} Based on corn = \$2.00/bu, fall N = \$0.25/lb N, spring N = \$0.275/lb N, and N-Serve = \$7.50/A.

^{3/} Only those seven years when a statistically significant yield difference occurred among treatments.

^{4/} Nitrogen recovery in the corn grain as a percent of the amount of fertilizer N applied.

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

Time/Placement	Years		
	1997-98	1999	3-Yr. Avg.
	----- Yield (bu/A) -----		
Fall, near row	188	145	174
Spr., between rows	188	181	186
LSD (0.10):	NS	5	3

Table 4. Average annual flow-weighted NO₃-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).

Time	N Treatment		N-Serve	4-Yr Avg.
	Rate			Flow-weighted NO ₃ -N
	lb N/A			mg/L
Fall	150		No	14.2c
“	150		Yes	16.2bc
“	225		No	18.1b
Spring	150		No	15.4bc
“	150		Yes	17.7b
“	225		No	24.4a
LSD (0.05):				3.0

Table 5. Corn production and nitrate loss as affected by time of N application and N-Serve at Waseca, 1987-93 (adapted from Randall et al., 2003a, 2003b).

N Treatment		7-Yr Average			Flow-weighted
Time	N-Serve	Corn yield	N recovery	Economic return to N ^{1L}	NO ₃ -N conc. in tile drainage ^{2L}
		bu/A	%	\$/A	mg/L
Fall	No	131	31	34	16.8
“	Yes	139	37	43	13.7
Spring	No	139	40	47	13.7
Split	No	145	44	56	14.6
LSD (0.10):		4			

^{1L} Based on corn = \$2.00/bu, fall N = \$0.25/lb, spring N = \$0.275/lb, N-Serve = \$7.50/A, and application cost = \$4.00/A/time.

^{2L} Across the 4-cycle corn (1990-93) – soybean (1991-94) rotation.

Table 6. Corn grain yield as affected by fall and spring application of N-Serve with anhydrous ammonia at Waseca, 1994-99 (adapted from Randall and Vetsch, 2005b).

Time of application	N-Serve	
	No	Yes
	----- 6-Yr Avg. Yield (bu/A) -----	
Fall	161	171
Spring	172	176

Table 7. Corn yield as influenced by N source, time of application, and N-Serve at Waseca, 1981-82 (unpublished data).

Nitrogen treatment		Time of Application	
Source	N-Serve	Fall	Spring
		----- Yield (bu/A) -----	
None	--	104	
Urea	No	157	164
"	Yes	155	167
An. Ammonia	No	162	168
"	Yes	170	173

Table 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).

Time	Nitrogen Management		3-Yr Average	
	Source	N-Serve	Yield bu/A	N Recovery %
Fall	Urea	No	152	43
"	"	Yes	158	47
"	An. Ammonia	No	168	60
"	"	Yes	170	63
Spr. Preplant	Urea	No	185	76
"	An. Ammonia	No	182	84
--	None	--	112	--

LSD (0.10):			8	

Agronomic and Environmental Implication of Phosphorus Management Practices

Antonio P. Mallarino, Department of Agronomy, Iowa State University, Ames

Larry G. Bundy, Department of Soil Science, University of Wisconsin-Madison

Introduction

Phosphorus (P) is an essential nutrient for growth of plants and aquatic organisms. Fertilizer or manure P application to land often is 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, including P, 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 region, 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 (site and management) factors and transport factors. This presentation 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

Soils of the North Central Region typically contain 300 to 1000 ppm of total P. Soils and aquatic systems are similar in that 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 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 availability to plants in soils is near to 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., 2004. Unpublished. NCR-13 Regional Soil Testing and Plant Analysis Committee). While water-soluble manure P may be a good indicator of short-term P loss potential when manure is applied to the soil surface, it is not a good indicator of P available to a crop or long-term loss. The more labile inorganic and organic P forms can become readily available for crops or algae shortly after being in contact with soil or surface runoff 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, a group of enzymes used 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 about 25 to 35% when mineral P supplementation is reduced accordingly. Recent research does not confirm early reports suggesting that its use significantly increases the proportion of soluble P in manure.

Application of 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 effect 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 effect 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.

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 the

soil and empirical work to assess how they correlate with crop growth and P uptake. 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 in 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 (Mallarino, 1997). 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.

Interpreting a soil-test value requires an understanding of the impacts of the extractant used on test results, method of soil sampling, sample handling, and 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. Although with the exception of calcareous soils the tests commonly used in the region are highly correlated, the actual quantities measured can differ greatly. For example, 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 tests in acid or near neutral soils (Mallarino, 1997). The inductively coupled (ICP) method of determining extracted P results in more measured P than the traditional colorimetric method. The ICP method is commonly used only for Mehlich-3 extractant. Mallarino (2002) showed that different interpretations are needed for the Mehlich-3 soil P test when colorimetric or ICP determination methods are used. As an example, Fig. 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 categories very low, low, medium or optimum, high, and very high or excessive. Specific interpretations for soils, crops, and growing conditions of each state have been published. In general, there are only small differences across states 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 among others) 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 often is very important for nutrients that are highly mobile in the soil (such as nitrate-N, B, Cl, and S) but less important for nutrients such as P with stronger retention by the soil.

Soil Sampling for Phosphorus

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 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.

Soil-test variation with depth results from a combination of soil-forming and management factors and from the differential mobility of nutrients in soils. Nutrients such as P with relatively low mobility tend to accumulate near the application point. 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, 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 (Fig. 2). Therefore, the proper depth for soil sampling and its consistency are important considerations. 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 P and other nutrients with relatively low mobility in soil often are collected from the top 6 to 8 inches of soil. Although a 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 provide sufficient information for establishing differential sampling depth recommendations.

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 using band methods. The challenge in these situations is to determine 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 appropriately.

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. Landscape position, soil color, soil mapping unit, and crop growth differences are examples of factors often used to help 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 many regions. 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 to have reasonable confidence on soil-test results. Much uncertainty still exists regarding how to best perform site-specific soil sampling and generate accurate soil fertility maps.

Phosphorus Application and Placement Methods

Phosphorus applications can be tailored to match crop needs and minimize excessive soil P accumulation. Use of soil testing and estimates of P removal with harvest are useful tools that allow P management over time. Phosphorus removal by crops varies greatly among species and with plant part harvested, and Extension services of most states provide tables with average values. Soils with naturally high STP levels are scarce 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, 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 /acre/year 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 with STP four to five times higher than optimal levels for crops, except for small starter fertilizer rates in some conditions.

The time of P application before 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 of planting without a significant loss of efficiency. Also, 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 efficiency. However, similar efficiency of broadcast and band fertilizer P for no-till crops in Iowa has been partly explained by broadcast P application several months (in the fall) before planting (Bordoli and Mallarino, 1998; Borges and Mallarino, 2000). Also, 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 et al. (1985) and Randall and Hoef, (1988) provide excellent summaries of early work. 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 the only practical approach (such as for forages) for applying P unless fertilizer is incorporated into the soil before crop establishment. These reviews 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 late 1980s has continued, although more emphasis has been placed 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., Randall et al, 1985; Robbins and Voss, 1991; Rehm et al., 1995), 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. Recent work in Iowa also 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 (2 by 2 inches relative to 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. 2) and similar results have been observed for injected liquid swine manure.

Starter fertilization, which involves low rates of nutrient mixtures placed near or in the seed furrow with the planters, is commonly used in corn production. Starter fertilization very often enhances early season plant growth and sometimes also increases grain yield and reduces grain moisture content at harvest. Most starter fertilizers contain N, P, and K or N and P and mechanisms of crop response to starter fertilizer are not always clear. Starter fertilization increases yield in low-testing soils because crops respond to the nutrients regardless of other management practices. At high soil fertility levels, however, the response to starter, when it occurs, is probably due to a placement effect that enhances early plant growth or helps overcome limitations to early nutrient uptake imposed by the management system or climate.

Several studies have attributed the corn yield response to starter fertilizer to the P in the starter while others indicated that starter N and/or K was needed. Geographic location and the associated climate appear to influence these findings, since experiments showing responses mainly to starter N (Touchton and Karim, 1986; Howard and Tyler, 1987; Howard and Mullen, 1991; Scharf, 1999; Niehues et al., 2004) generally are located in southern or central USA corn-producing areas while results showing response to P and/or K in starters (Bundy and Widen, 1992; Rehm et al., 1995) usually are from northern production regions. The research in central regions has shown frequent response to starter P and (or) N (Ritchie et al., 1996; Lamond et al., 2001; Bermudez and Mallarino, 2002; Mallarino, 2003; Niehues et al., 2004). Vetsch and Randall (2002) concluded that corn responses to N-P-K starter fertilizer in their 4-yr study with several tillage systems were probably not due to a consistent response to a single nutrient. Iowa work with no-till corn after soybean in high-testing soils (Mallarino, 2003) showed that starter 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/acre) 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 partly explain response to S containing starter mixtures in the region.

The rate and placement of starter fertilizers can influence their performance. Higher starter rates may be needed to optimize production in low-testing soils than in soils that test higher. Early and recent work (Niehues et al., 2004) with seed-placed starter indicate 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/acre of these nutrients, although the safe application rate is highly affected by soil moisture content and the source of N and K. Because of this limitation, 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 (Kaiser et al., 2005), and it cannot substitute for large nutrient application rates using different methods.

Although the response to starter fertilizer is consistent and often large in low-testing soils, responses are less consistent when soils test optimum or high in P mainly when it is applied in addition to broadcast fertilizer rates recommended for corn or the corn-soybean rotation. For example, studies by Scharf (1999), Bundy and Widen (1992), Ritchie et al. (1996), Lamond, et al. (2001), Niehues et al. (2004), reported responses to starter mixtures either in high-testing soils. However, Bermudez and Mallarino (2002), Mallarino (2003), and Kaiser et al. (2005) found no significant response to starter P in soils testing high in P or when the starter was applied in addition to recommended broadcast P-K rates for corn-soybean rotations. Table 1 shows a summary of experimental results. While the reasons for positive response to starter have not always been conclusively identified, Bundy et al. (2005) suggested that possible causes include response to N and/or K in the starter; stimulation of early season plant growth, and enhanced uptake caused by interaction of the starter fertilizer nutrients. Furthermore, research indicate a greater likelihood of response to starter for continuous corn than for corn after soybean, in environments with short growing period where an acceleration of plant growth can translate into higher yield even at high STP levels (Farber and Fixen, 1986; Bundy and Widen, 1992; Bundy and Andraski, 1999), and for some corn hybrids than for others.

Studies in northern corn production areas 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. (1993) evaluated starter fertilization for corn silage on a soil with excessively high STP in northern Wisconsin. The starter increased yield in 1 of 3 years, but there was no interaction between manure and starter fertilization. Jokela (1992) determined corn silage yield response to N-P-K starter fertilizer at 12 sites in Vermont with medium or higher soil P and K levels that had received manure. Silage yield was increased by the starter fertilizer at most of the sites with medium P and K soil tests and poor soil drainage. 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.

Dense grid soil sampling from many fields of the Midwest has shown very large within-field spatial variability of STP. Available precision agriculture technologies to producers or custom fertilizer and manure applicators facilitate application of P at rates adequate for different parts of a field. 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. 3).

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. We briefly discuss here 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 often also are considered as source factors.

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 (Attia and Mallarino, 2002; Andraski and Bundy, 2003). Many studies have found that concentrations of dissolved, bioavailable, and particulate P in runoff increase linearly as STP increases. In some cases, P concentration in runoff may increase more rapidly at very high STP levels compared with lower levels. Another consideration is that the total P concentration in sediment is higher than in the eroded soil; this P enrichment occurs due to removal 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 and Mallarino, 2002) indicated a change point of approximately 60 ppm by the Olsen test or 100 ppm by the Bray P-1 test, which is four to five times larger than optimal 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 near the soil surface is well known, and unless the P is incorporated, it results in high P levels in the mixing zone of soil and runoff especially for no-till and forage fields. This affects STP results and has implications for P loss in runoff. Tillage and deep P banding reduces STP stratification, but significant stratification

exists with use of implements such as chisel plows and field cultivators (Fig. 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 is showing that reasonable predictions of STP stratification are possible for the purpose of assessing risk of P loss.

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 (2003) showed that variable-rate P application could be practically implemented based on P index ratings for field zones, not just based on STP.

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 tend to reduce P concentrations (Figs. 4 and 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 shows that a runoff event 10 to 15 days after fertilizer or manure application can reduce total and dissolved P concentrations in runoff by over 50% as compared to rain within 24 hours when manure was applied to soil having corn or soybean harvest residues and was not incorporated (Fig. 4). Data in process indicates this effect varies for different manures, and is higher for liquid swine manure. We believe that added P that reacts with the soil is less prone to losses in runoff. Therefore, when the P is not injected or incorporated, applying P when runoff events are less likely can substantially reduce the risk of P loss with runoff. 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. For example, Daverede et al. (2004) showed slightly larger runoff P concentrations for liquid swine manure than fertilizer (Fig. 5) while

Iowa research (M.U. Haq and A.P. Mallarino, unpublished) is finding slightly higher runoff P for fertilizer. However, P losses after applying other manure types tend to be lower than for fertilizer at similar total P application rates (M.U. Haq and A.P. Mallarino, Iowa State University, unpublished). Several factors not well identified at this time can explain this result. Manure typically has less soluble P than fertilizer P, adds organic matter (sometimes it includes bedding), and often increases water infiltration all of which can result in less runoff P immediately after applying manure than fertilizer. 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. Also, 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.

We mentioned above that annual or biannual applications are similarly effective for most crops of the region. 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, more spaced P application strategy may increase P loss. However, there is little evidence that applying the same amount of P in infrequent applications at higher rates results in more long-term potential for P runoff loss than annual applications with proportionally lower rates of application. Infrequent application may allow better timing of application and more careful application so risk of runoff may be less with infrequent applications at higher P rates. Also, infrequent N-based applications of manure benefit farmers as it allows them to meet the full N need of the crop in that year and reduce the need for supplemental N fertilization.

Incorporation of applied P, deep banding of P fertilizer, or injection of liquid manure generally reduces 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 of a comprehensive tool, such as the P index, that considers both source and transports factors to assess risk of P loss from fields.

References

- 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.* 2003 32:310-316.
- Atia, A.M., and A.P. Mallarino. 2002. Agronomic and environmental phosphorus testing for soils receiving swine manure. *Soil Sci. Soc. Am. J.* 66:1696-1705.
- 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. *Soil Sci. Soc. Am. 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. *Soil Sci. Soc. Am. 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., 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., and P.C. Widen. 1992. Corn response to starter fertilizer: Planting date and tillage effects. *Better Crops Plant Food* 76:20-23.
- Bundy, L.G., H. Tunney, and A.D. Halvorson. 2005. Agronomic aspects of phosphorus management. p. 685-727. In A. Sharpley and T. Sims (eds.). *Phosphorus: Agriculture and the Environment*. ASA Monograph Series No. 46. ASA-CSSA-SSSA, Madison, WI.
- Daverede, I.C., A.N. Kravchenko, R.G. Hoelt, 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.* 2004 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. *Soil Sci. Soc. Am. 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.
- Hoelt, R.G., and T.R. Peck. 2001. Soil testing and fertility. p. 78-116. In *Illinois Agronomy Handbook 2001-2002*. Circ. 1360. Coop. Ext. Serv., Univ. of Illinois. Urbana.
- Howard, D.D., and M.D. Mullen. 1991. Evaluation of in-furrow and banded starter N P K nutrient combinations for no-tillage corn production. *J. Fert. Issues* 8:34-39.
- Howard, D.D., and D.D. Tyler. 1987. Comparison of surface applied phosphorus and potassium rates and in-furrow fertilizer combinations for no-till corn. *J. Fert. Issues* 4:48-52.

- Jokela, W.E. 1992. Effect of starter fertilizer on corn silage yields on medium and high fertility soils. *J. Prod. Agric.* 5:233-237.
- 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. University of Wisconsin Extension Publ. A2809. Madison, WI.
- 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. p. 183-189. In North-Central Extension-Industry Soil Fertility Conf. Proceedings. Vol. 18. Des Moines, IA.
- 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. p. 137-143. In Proc. North Central Ext. Industry Soil Fert. Conf. Vol. 17. 14-15 November. Des Moines, IA. Potash and Phosphate Inst., Brookings, SD.
- Mallarino, A.P., and R. Borges. 2005. Phosphorus and potassium distribution in soil following long-term deep-band fertilization in different tillage systems. *Soil Sci. Soc. Am. J.* In press.
- 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, AP, 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.
- 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. *Soil Sci. Soc. Am. J.* 67:1928-1934.
- Mallarino, A.P. 2003b. Foliar and starter fertilization. Are they needed to supplement primary fertilization? p. 113-120. In *The Integrated Crop Management Conf. Proceedings*. Dec. 3-4, Ames, IA. Iowa State Univ. Extension.
- Mallarino, A.P. 2003c. Using the Iowa phosphorus index for agronomic and environmental management of fertilizer and manure phosphorus. p. 121-126. In *The Integrated Crop Management Conf. Proc.* Dec. 3-4, Ames, IA. Iowa State Univ. Extension.
- Mallarino, A.P., and J.S. Schepers. 2005. Role of precision agriculture in phosphorus management practices. In A. Sharpley and T. Sims (eds.). *Phosphorus: Agriculture and the Environment*. American Society of Agronomy Monograph Series No. 46. Madison, WI.
- 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.

- Potash and Phosphate Institute (PPI). 2001. Soil test levels in North America. Summary update. Tech. Bull. 2001-1. PPI/PPIC/FAR. Norcross, GA.
- Randall, G.W., and E.G. Hoelt. 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 Plant Food* 85:12-15.
- Randall, G.W., K.L. Wells., and J.J. Hanway. 1985. Modern techniques in fertilizer application. p. 521-560. In O.P. Engelstad (ed.) *Fertilizer technology and use*. 3rd ed. SSSA, Madison, WI.
- 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. *Soil Sci. Soc. Am. 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. Univ. Minn. Ext., St. Paul.
- Ritchie, K.B., E.G. Hoelt, E.D. Nafziger, L.C. Gonzini, J.I. Warren, and W.L. Banwart. 1996. Starter fertilizers for no-till corn. *Better Crops Plant Food* 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.). Iowa State Univ. 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
- Touchton, I.T. and F. Karim. 1986. Corn growth and yield responses to starter fertilizers in conservation-tillage systems. *Soil Tillage Res.* 7:135-144.
- 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. SERA-17 Committee Position Paper. I. press.
- Vetsch, J.A., and G.W. Randall. 2002. Corn production as affected by tillage system and starter fertilizer. *Agron.* 1. 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.

Table 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).[†]

Location	Reference	Response frequency	Response
Illinois	Ritchie et al. (1996)	8 of 9 trials	14 bu/acre average
Iowa	Buah et al. (1999)	7 of 9 trials	4 to 18 bu/acre
Iowa	Bermudez & Mallarino (2002)	5 of 7 trials [‡]	2 to 8 bu/acre [‡]
Iowa	Mallarino (2003)	3 of 8 trials	5 bu/acre average
Iowa	Kaiser et al. (2005)	1 of 2 [‡]	15 bu/acre [‡]
Missouri	Scharf (1999)	6 of 6 trials	13 bu/acre average
Wisconsin	Bundy & Widen (1992)	8 of 12 trials	15 bu/acre average

[†] Soils tested medium, optimum, or higher in P and K according to local interpretations.
[‡] Excludes data for other no-till sites testing low and others managed with tillage.

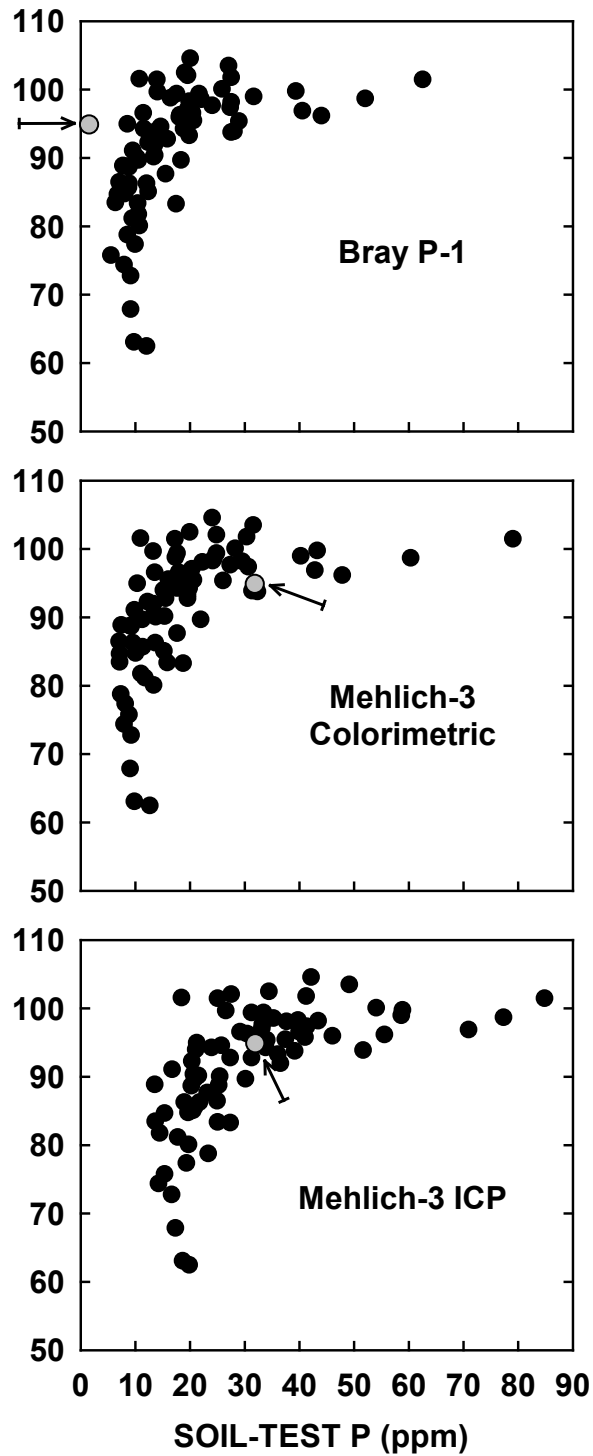


Fig. 1. Example of the relationship between corn yield and soil-test P measured by three P tests commonly used in the Upper Mississippi River region (adapted from Mallarino, 2003a). The gray point and arrow indicate results for a highly calcareous soil.

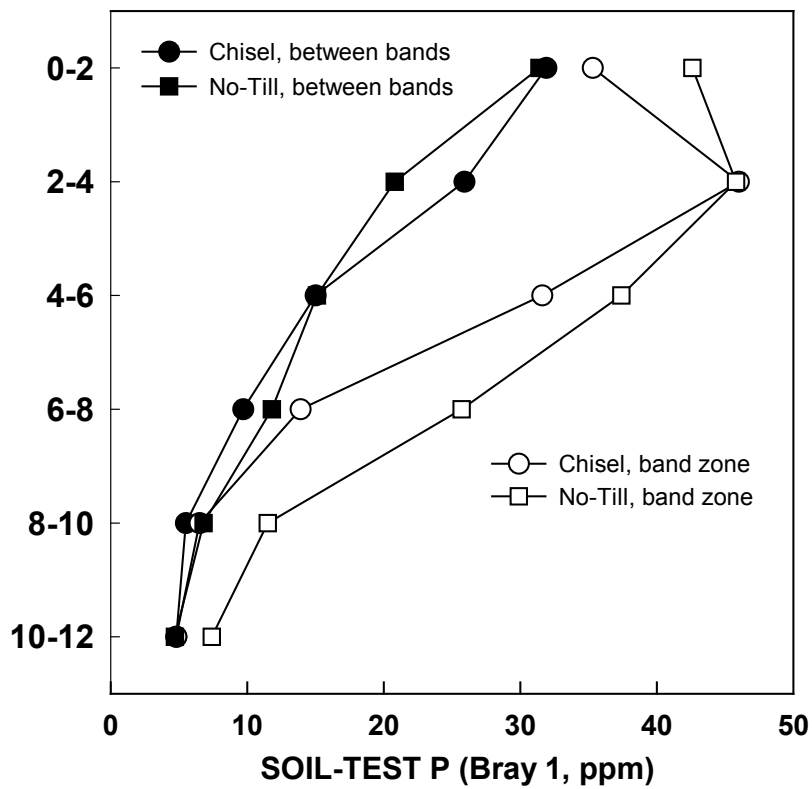


Figure 2. Mean soil-test P across five sites after several years of no-till or chisel-plow tillage and deep-band P fertilization for band and inter-band zones (adapted from Mallarino and Borges, 2005).

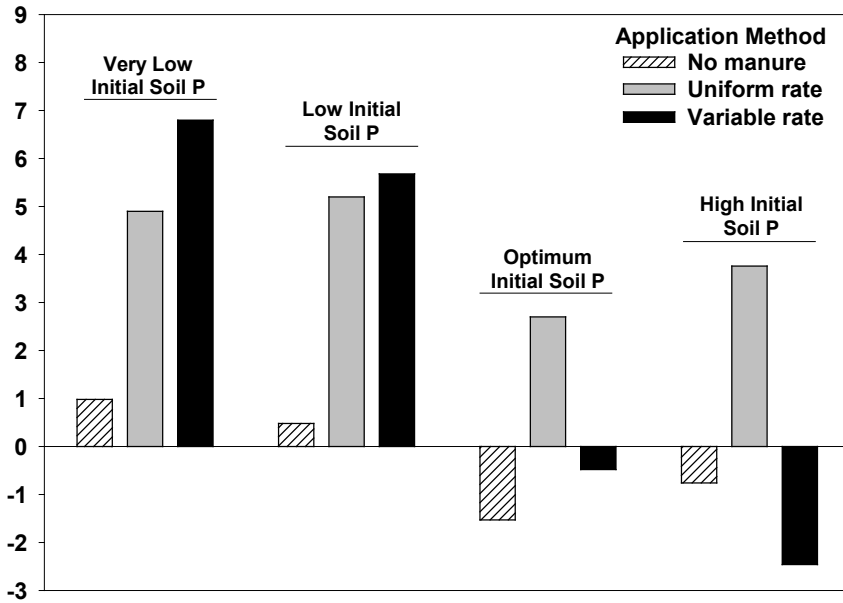


Figure 3. Effect of uniform application and soil-test P based variable-rate application of liquid swine manure on soil test P change within a field for various initial soil-test P interpretation classes. Adapted from Mallarino and Schepers (2005).

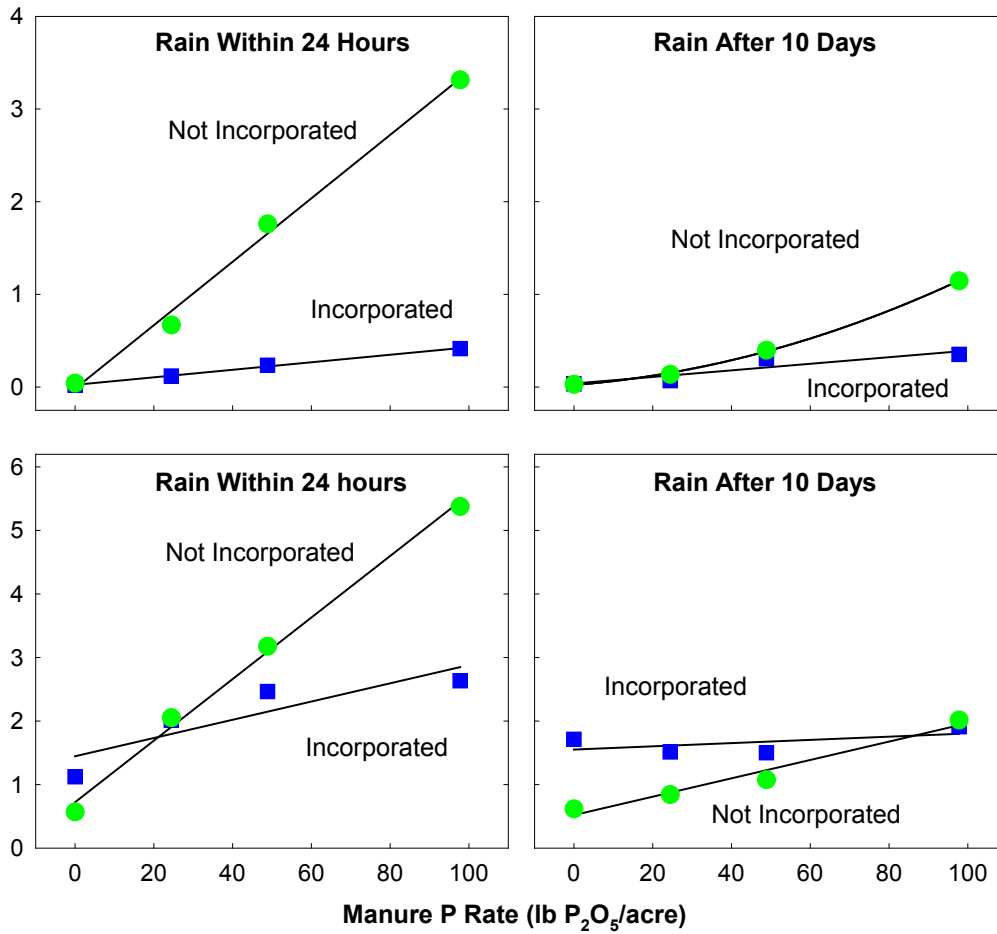


Fig. 4. Effect of liquid swine manure incorporation into the soil and time of simulated rainfall on dissolved and total P concentrations in runoff (unpublished, B.L. Allen, A.P. Mallarino, and J.L. Baker, Iowa State University).

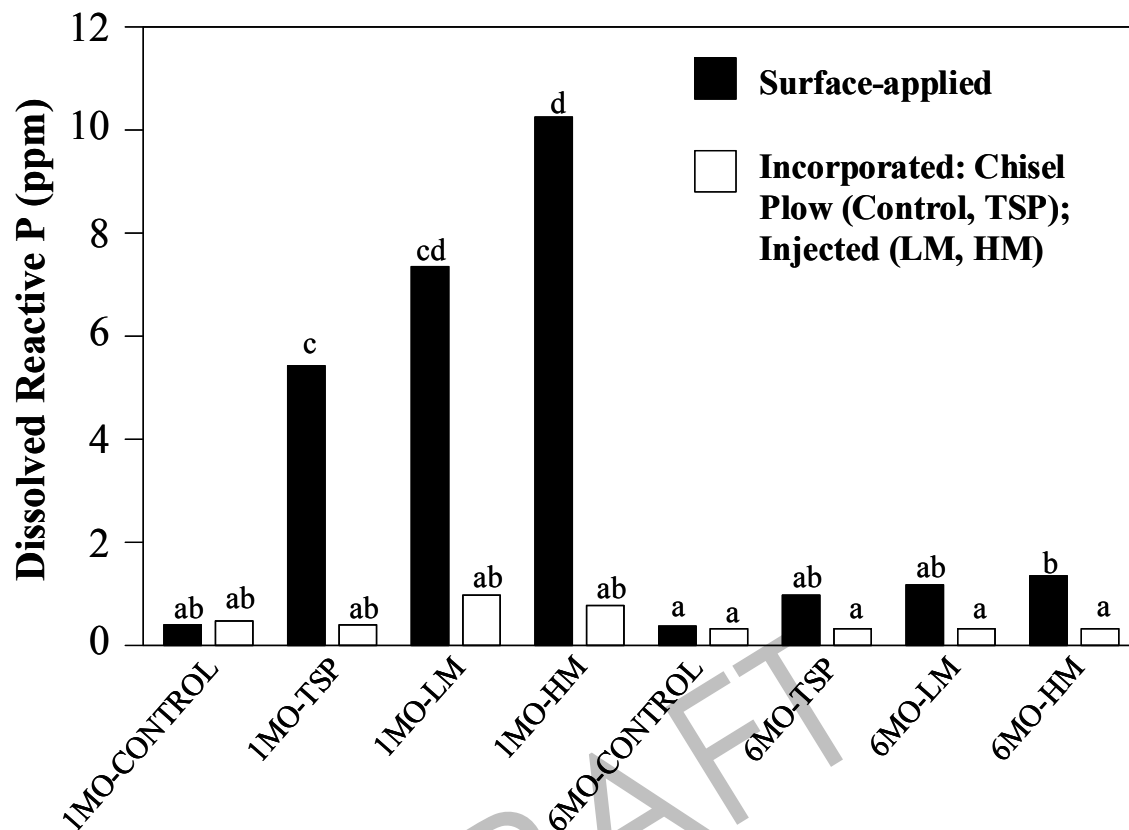


Fig. 5. Mean dissolved reactive phosphorus (DRP) concentration in runoff as affected by time of rainfall simulation one (1MO) and six months (6MO) after P application; P source (control, TSP = triple superphosphate, LM = low swine manure rate, and HM = high swine manure rate); and application method (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 /acre for TSP, 68 to 80 lb P_2O_5 /acre for LM, and 135 to 161 lb P_2O_5 /acre for HM. Values that have the same letters are not significantly different ($P < 0.1$). Adapted from Davedere et al. (2004).

DRAFT

Using Manure as a Fertilizer for Crop Production

John A. Lory and Ray Massey, Associate Professors of Extension, University of Missouri

Brad Joern, Professor, Purdue University

Introduction

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.

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 controlled by many factors. Nutrients can leach through the soil profile into ground water or reemerge as seeps, springs or from tile drains to enter surface waters. Runoff can carry nutrients as dissolved ions and in particulate matter. 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. Frequently, additions of phosphorus to surface freshwater resources increases algal growth, increases the cost of water treatment and reduces aesthetics and some recreational uses. Excess nitrate nitrogen in drinking water can pose health risks to babies 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 being used in the US 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 the past decade are:

- The NRCS 590 nutrient management policy released in 2001.
- The revised Concentrated Animal Feeding Operation (CAFO) rules released by USEPA in 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, improve 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 under utilized 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 source:

- **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 the blending of fertilizer constituents providing custom fertilizers that meet the specific nutrient requirements of a crop and field. Manure nutrient ratios are a product of animal nutritional considerations and manure storage and frequently do not match the crop requirements. For example applying poultry litter to meet the nitrogen needs of a corn crop 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.
- **Nutrient availability:** Most commercial fertilizers are designed to be rapidly available to crops when applied to the soil. The organic nitrogen fraction of manure reduces the availability and predictability of the manure as a nitrogen source because the availability of organic nutrients are dependent on microbial activity in the soil. The chemistry of manure makes inorganic nitrogen in manure 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 insures portions of the field will have nutrient deficits, an economic liability to the farmer; the second strategy maximizes yield but also insures 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. This is especially true for manure storages with an inflexible cleanout schedule such as liquid manure from storages with less than one-year capacity. 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.

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 over estimating its value to the farmer. The objective of this paper is to help the reader to better understand the characteristics of manure and the value of manure as a fertilizer source.

Potential

Use of manure does not require that nutrient losses from agricultural systems increase compared to commercial fertilizer systems. There is extensive research showing that when equivalent rates of nutrients are applied as manure or commercial fertilizers that nutrient losses from manure applications are similar or below 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. 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.

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 conservatively high rates to insure sufficient available nutrients or failure to properly account for the fertilizer value of manure (e.g. waste applications) all lead to over-application of nutrients. There is extensive research demonstrating 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 linear 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. Numerous studies have shown that soil test values increase linearly 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.

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

Important factors

Manure nutrient characteristics

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 Table1). Publications such as *Manure Characteristics* (MWPS-18, 2004) provide book values for many animal types and specialized manure handling systems. Despite this specificity, book values should be judged as rough estimates. 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.

Table 1. Estimated nutrient concentration in manure for selected animal types and manure storage and handling systems. Data is adapted from *Manure Characteristics* (MWPS-18, 2004).

Livestock system	Units	Total nitrogen (N)	Ammonium N	Phosphate (P ₂ O ₅)	Potash (K ₂ O)
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
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

Sampling manure storages is an essential part of using manure as a fertilizer. Unfortunately it can be challenging to obtain representative samples of manure storages at the time of manure application. Slurry tanks are best sampled 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. Lagoons can be easily sampled a week or so before pumping to provide results representative of lagoon water if no agitation is planned. Current methods to sample lagoon sludge in the bottom of the lagoon are inadequate and the resulting estimates of lagoon sludge nutrient concentrations are unreliable.

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 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.

When test results do not exist for a manure storage, results from a similarly managed storage will typically be superior to book values. In poultry operations litter test results from other buildings affiliated with the same integrator 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 grand mean of all the buildings sampled within the same integrator complex.

An emerging approach is to estimate manure nutrient content based on animal feed and engineering design criteria of the storage facility. This approach has the most potential for covered slurry storages 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 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 storages 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 water logged soil conditions all limit nutrient release from manure.

Inorganic nitrogen in manure is dominantly in the ammonium form ($\text{NH}_4\text{-N}$) because there is little oxygen in most manure storages preventing formation of nitrate ($\text{NO}_3\text{-N}$). Manure also typically has a pH of at least 7. This combination of ammonium nitrogen and alkaline pH makes inorganic nitrogen prone to volatilization. Significant amounts of nitrogen are lost from manure storages 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 significantly in their 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.
- 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.

There are significant differences among states in estimated available nutrients, particularly in estimates of nitrogen availability (Table 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.

Table 2. First-year plant available nutrients in 1000 gallons of 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, 2004).

State	Available nutrients		
	Nitrogen	Phosphate	Potash
lbs per 1000 gallons			
Illinois	30	42	30
Indiana	27	42	30
Iowa	38	42	30
Kansas	9	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

State-to-state differences have dramatic impacts on the amount of manure that can be applied to a field. An Iowa farmer seeking to apply 150 lbs/acre of available nitrogen can apply 3,950 gallons/acre of slurry, a Minnesota farmer 8,350 gallons/acre and a Michigan farmer 16,650 gallons/

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.

Predicting nitrogen availability in manure is difficult because it is so 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 capabilities of the state of the science.

Manure value

The value of manure nutrients is a topic fraught with misconceptions and over-simplifications. 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 and importance of manure to the operation's bottom line varies significantly among farms.

An earlier section of this paper 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 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 gallons). Low nutrient concentration increases the time needed for nutrient application and limits the distance manure can be economically hauled.

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 36 hog operations (17 lagoon operations and 18 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. Extracting manure value was a more important element of profitability on slurry operations. Manure value represented 2% of net income on lagoon operations compared to 16% on slurry operations. Manure value exceeded application costs on nearly 60% of slurry operations compared to 15% of lagoon operations. Why are lagoons favored by some farmers over slurry tanks if they depress manure value? Farmers with lagoons need less land for manure application and were less dependent on land not owned by the operation. It also took more time per animal unit to apply manure on slurry operations. Most importantly, investment in slurry storage and handling systems

did not increase return on assets on these operations; it was more profitable to invest money in raising more hogs than 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. Animal feeding operations are unlikely to need to immediately convert to a phosphorus-based application rate because of the revised rules. An estimate of the phosphorus land base requirement does provide 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:

Land increase (%) =

$$((\text{crop need N:P}_2\text{O}_5 \text{ ratio} \div \text{manure available N:P}_2\text{O}_5 \text{ ratio}) - 1) \times 100\% \text{ Eq. 1}$$

This equation emphasizes that both the nutrient ratio of crop receiving manure and the nutrient ratio of manure affect the conversion from nitrogen-based to a phosphorus-based application strategy. There will be less impact on fields with crops that have low nitrogen to phosphate removal ratio such as wheat (1.9) or corn (2.2) than on a crop with a higher ratio 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% $((2.4)/(0.7)-1) \times 100$ increase in land needs; if the operation used 100 acres for nitrogen-based application rates 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 $((2.4)/(3.4)-1) \times 100 < 0$. Phosphorus rules also will have a greater impact on farms with less productive land because increased land needs are proportional to current land needs. 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 the phosphorus rotation allows application to a fraction of the land base but rotates which land receives manure each year.

An analysis of 39 US swine operations (19 slurry and 20 lagoon operations) indicated that annual phosphorus limit approach posed significant feasibility issues for farmers spreading slurry manure. 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 only through 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 not a feasibility problem on operations applying unagitated 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 more similar to 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 ever be applied in a single phosphorus-based application rate.

Another question is whether 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 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.
- Global economic concerns such as does it pay to invest in upgraded manure equipment compared to adding to other aspects of animal production.
- Manure value concerns such as does it pay to haul manure to a particular field and will the manure provide the needed nutrients for crop production.

Manure has a positive impact on the bottom line 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.

References

- ASAE. 2005. Manure Production and Characteristics. Publication D384.3, March 2005.
- Bundy, L.G., T.W. Andraski and J.M. Powell. 2001. Management practices effects on phosphorus losses in runoff in corn production systems. *J. of 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. Env. Qual.* 23:579-584.
- Joern, B.C. and P. Hess. 2004. Manure Nutrient Availability Calculator. Available at <http://www.agry.purdue.edu/mmp/>.
- Lory, J.A., R.E. Massey, J.M. Zoluvich, J.A. Hohne, A.M. Schmidt, M.S. Carlson and C.D. Fulhage. 2004. Feasibility of phosphorus-based application rates on 39 swine operations. *J. of Environ. Qual.* 33:115-1123.
- MWPS-18. 2004. Manure Characteristics. Section 1, 2nd edition. Midwest Plan Service. Ames IA.
- USEPA. 2002. National Water Quality Inventory: 2000 report. EPA-841-R-02-001. August 2002. Washington DC.

Effects of Erosion Control Practices on Nutrients Losses

George F. Czapar, University of Illinois

John M. Laflen, Iowa State University

Gregory F. McIsaac, University of Illinois

Dennis P. McKenna, Illinois Department of Agriculture

Elements of Soil Erosion

Soil erosion by water is 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 flowing water to small channels or rills. Detachment of soil particles also occurs in these 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 be grassed waterways, or designed channels that limit flow conditions so that channel erosion does not occur.

Gross erosion includes sheet, rill, gully and channel erosion, and is the first step in the process of sediment delivery. Because much eroded sediment is 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, 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, the lower the ratio of sediment yield at the watershed outlet 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 percent (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 are dominant factors affecting soil erosion, but cannot be easily modified. In this discussion, erosion control practices are grouped as conservation tillage, which reduces sheet and rill erosion, and other practices which 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 diverts 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 1 and 2 show current tillage practices for soybean and corn from the seven corn belt states (CTIC, 2004). Illinois, Indiana, Iowa, Ohio, Minnesota, Missouri, and Wisconsin planted 17.6 million acres of no-till soybeans in 2004, while only 6.3 millions acres of corn were planted using no-till practices in those seven states.

Table 1. Tillage practices in seven corn belt states for soybean production (CTIC, 2004).

State	Soybean Acres	No-Till	Mulch-Till (30% residue)	Reduced-Till (15-30% residue)	Conventional-Till (0-15% 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 2. Tillage practices in seven corn belt states for corn production (CTIC, 2004).

State	Corn Acres	No-Till	Mulch-Till (30% residue)	Reduced-Till (15-30% residue)	Conventional-Till (0-15% 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%

Contouring is the practice of performing field operations on the contour. Usually, there are ridges developed when the land is tilled or at planting, 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 no impact on soil erosion.

Stripcropping is the practice of growing alternate strips of different crops along the contour. Alternating strips are 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 or permanent grass. These strips reduce water erosion by being on the contour, and with runoff passing from highly erodible row crops into small grains or grass where considerable deposition may take place.

Grassed waterways and grade control structures are designed to keep erosive forces in channels

carrying 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 follow these broad channels to an outlet. Terraces reduce slope length and deliver surface runoff through terrace channels that are designed to be non erodible and to prevent deposition of sediment. 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

Although significant gains in erosion control have been made over the last 20 years, soil erosion continues to be a important environmental concern. It is estimated that over 460,000 tons of topsoil eroded from the seven corn belt states in 1997, while in 1982 the estimated loss was approximately 790,000 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/acre/year, 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. However, both have limits of application. As slope increases, the maximum slope length decreases, and when erosion is most severe, such as slopes exceeding 9%, much of the effectiveness is lost, and the length of slope to which it can be applied becomes quite low.

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 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. Laffin 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. 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 3 summarizes simulation results comparing the effects of various erosion control practices on soil and nutrient losses relative to a tillage system typically used in the Corn Belt. WEPP (Laflen et al., 1997) 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, 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 sediment delivery ratio may range between 0 and 0.95 depending primarily on distance to a stream.

For reference, the base soil loss of 7.8 tons/acre/year is about twice the 1997 average annual soil loss in the Corn Belt. (2004 estimates for Illinois indicate less than 10 percent of fields have erosion rates > 7.5 t/a/y). For many watersheds in the region, total phosphorus yields to streams in intensively cropped watersheds are about 1-2 lb/a/y. Total nitrogen yields vary greatly, but are typically less than 10 lb/a/y in non-tiled drained watersheds and greater than 20 lb/a/y in tile-drained watersheds. 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., 2000).

All erosion control practices considered show greater losses of dissolved nutrients. The effect of erosion control practices in increasing runoff losses of nitrate is probably not significant because the dominant path for nitrate loss is leaching and nitrate concentrations in runoff are usually low. In contrast, the long term impacts of increased losses of dissolved phosphorus and decreased losses of particulate phosphorus due to the widespread adoption of conservation tillage systems is unclear.

Table 3. Estimated soil and nutrient losses under various erosion control practices.

Practice	Runoff	Soil erosion/ Sediment yield	Nutrient enrichment ratio*		Losses in surface runoff water (lb/ac)		Losses in eroded soil (lb/ac)		Total water and soil losses (lb/ac)	
			Sediment	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

*Nutrient enrichment ratios, calculated relative to the typical tillage practice, were based on concentrations taken from Baker and Laffen (1983), and on soil erosion and sediment yields.

Important 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 for a variety of reasons, such as high clay content, surface crusting, high water table, or shallow bedrock. Phosphorus transport in runoff tends to increase with increasing P concentration at the soil surface and increasing runoff (Sharpley et al. 2003). Thus, practices that reduce P concentrations in the soil surface and/or reduce surface runoff are most effective in controlling P transport. When tillage is reduced or eliminated, particulate P losses in surface runoff usually decline, but dissolved P losses may increase because P 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 in reduced tillage systems.

Conservation tillage practices that leave crop residue on the soil surface protect fine textured soils from forming surface crusts, and thereby 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 maintain high porosity at the soil surface (Shipitalo et al. 2000). But in some settings, no-till has not had much influence on runoff (Gihdey 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. When infiltration is limited by a claypan, shallow bedrock, saturation of the soil from a high water table, or seasonal precipitation patterns which leave most soils saturated,

residue cover may have little influence on runoff or dissolved P transport. Conservation tillage is probably most effective in reducing runoff, and losses of soil and phosphorus in well drained fine textured soils, and where P fertilizer or manure is injected.

The interaction of tillage systems and nutrients in tile drainage is unclear. Tile drainage reduces surface runoff and thereby 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.

In contrast to P, nitrate is highly soluble and generally does not adsorb to soils. Rather, when water infiltrates into the soil, nitrate tends to move with the 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 ground water, 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).

Most of the soil and nutrients losses in surface runoff tend to occur in a few rare events that involve large quantities of runoff. Most conservation measures work best in 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 a 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 are also somewhat variable and therefore often difficult to 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 flows are likely to 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 P concentrations are available. This will lead to a different set of water quality problems. This possibility should not discourage conservation efforts, but should inform expectations and strategies of conservation programs.

Limitations of Erosion Control Practices

Conservation tillage systems that leave a great deal of crop residue 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 for the corn belt that profits from conservation tillage systems for a corn-soybean rotation 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. In some cases, 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 American agriculture. The moldboard plow has been replaced with the chisel plow, or other full width tillage tools, that can leave considerable residue on the soil surface—even though when it is combined with secondary tillage system on a number of crops, it may not leave 30% of the surface covered with plant residue, the residue cover level used as the minimum level for conservation tillage. These tillage tools have become the “conventional” tillage tools of modern agriculture and have few limitations. There is almost always one of these systems that every farmer can use, and they can be used in such a way as to have a major impact on reducing soil erosion. Even small amounts of residue may reduce soil erosion considerably on many lands in the Corn Belt.

While 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 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, generally it is practiced as cross slope farming with machines traveling parallel to field boundaries.

Contouring is effective in small and medium sized storms, and has little effectiveness for large storms. It diminishes in effectiveness as annual rainfall increases, and as slopes increase. At its maximum effect, contouring will reduce erosion about 50%. However, on long slopes, or very steep slopes, this practice is not very effective. Contouring has no impact on crop yields, unless ridges are high and it is used in areas where yields are reduced because of moisture stress. In these cases, yields may be increased

because of moisture conservation.

Terraces that drain via underground drain lines trap so much sediment that pondage volume may 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 such terraces. An additional benefit of such terraces is that much runoff is stored in the impoundments, and released at very low rates, reducing down stream channel erosion and off-site damages due to flooding. However, such terraces are usually designed to store a limited amount of runoff, and storms that are larger than the usual 10-year design period may lead to overtopping, causing damage not only to the terrace, but to channels and structures downstream. Terraces are expensive to construct, remove land from production, and 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 very 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 life of the basin.

Cost Effectiveness of Erosion Control Practices

The annual per ton and per pound cost estimates shown in Table 4 should be considered as order of magnitude estimates of the cost-effectiveness of various erosion control practices. As expected, the costs of land management changes by a producer are less than those for structural practices, even though some structural practices may be more effective in reducing sediment and nutrient losses. If a producer adopts a practice as a standard 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 cost per acre of incentive payments for changes in management practices, such as zero till or contouring, is constant. Therefore, the cost per ton of reductions in soil loss and associated nutrients is dependent on the current erosion rate on the field. The costs of structural practices vary more widely based on site conditions. Forster and Rausch (2002) reported per-ton costs in two Ohio watersheds of about \$2.50 for no till and more than \$40 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 will also vary 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 per ton of soil or per pound of nutrient may be significantly different, depending on location.

Table 4. Estimated annual costs for reductions in soil and nutrient losses for various erosion control practices. Practice effectiveness from Table 3 used for estimates of cost effectiveness.

Practice	Incentive payment/ construction cost (\$/ac)	Practice life-span (years)	Annual cost erosion reduction (\$/t/yr)	Annual cost nitrogen reduction (\$/lb/yr)	Annual cost phosphorus reduction (\$/lb/yr)
No-till	\$20	2	\$1.46	\$0.38	\$1.00
Contouring	\$10	5	\$0.51	\$0.10	\$0.26
Stripcropping	\$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

Summary

The maximum annual amount of soil that can be removed before the 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. Nutrient and sediment loads delivered to surface water are 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 phosphorus levels adjacent to a stream and eroding at half the T value may have greater impacts on water quality than a field with low phosphorus eroding at $>3T$, 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. Continuous no-till systems may sequester five times more carbon than conventional tillage.

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. Laffan. 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.
- Conservation Technology Information Center. 2004. National Crop Residue Management Survey. Online at: <http://www.ctic.purdue.edu>
- Forster, D.L. and J.N. Rausch, 2002, Evaluating Agricultural Nonpoint-Source Pollution Programs in Two Lake Erie Watersheds, *JEQ*, 31:24-31.
- Ghidey, F. and E.E. Alberts. 1996. Comparison of Measured and WEPP Predicted Runoff and Soil Loss for Midwest Claypan Soil. *T. Am. Soc. Agr. Engr.* 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. NOAA Coastal Ocean Program Decision Analysis Series No. 17. NOAA Coastal Ocean Program, Silver Spring, MD.
- Lafren, J. M., H. P. Johnson, and R. C. Reeve. 1972. Soil loss from tile-outlet terraces. *J. Soil Water Conserv.* 27(7):74-77.
- Lafren, 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*. Van Nostrand Reinhold, New York, NY.
- Sharpley, A.N., T. Daniel, T. Sims, J. Lemunyon, R. Stevens, and R. Parry. 2003. *Agricultural Phosphorus and Eutrophication 2nd Ed.* US Department of Agriculture, Agricultural Research Service. ARS-149, 44pp.
- Shipitalo, M.J., W.A. Dick, W.M. Edwards. 2000. Conservation tillage and macropore factors that affect water movement and the fate of chemicals. *Soil & 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.
- U.S. Department of Agriculture. 2000. Summary Report: 1997 National Resources Inventory (revised December 2000), Natural Resources Conservation Service, Washington, DC, and Statistical Laboratory, Iowa State University, Ames, Iowa, 89 pages.
- Zucker, L.A. and L.C. Brown (editors). 1998. *Agricultural drainage: Water quality impacts and subsurface drainage studies in the Midwest*. Ohio State University Extension Bulletin 871.

Potential and Limitations of Cover Crops, Living Mulches and Perennials to Reduce Nutrient Losses to Water Sources from Agricultural Fields

T.C. Kaspar, Plant Physiologist, USDA-ARS, National Soil Tilth Laboratory

E.J. Kladivko, Professor of Soil Physics, Agronomy Dept., Purdue University

J.W. Singer, Research Agronomist, USDA-ARS, National Soil Tilth Laboratory

I) Introduction

A) What is the definition of the specific practice that would be recommended?

- 1) Cover crops, living mulches, and perennial crops can extend the season of active nutrient uptake and living soil cover and thereby reduce nutrient losses in water and sediment. The conversion of the prairies or other native vegetation ecosystems to summer annual grain crops resulted in a shortening of the season of living plant cover and nutrient uptake. Summer annual grain crops, like corn and soybean, accumulate water and nutrients and provide living cover for only about four months (mid-May to mid-Sept), whereas in natural systems some living plants are actively accumulating nutrients and water whenever the ground is not frozen (at least 7 months; April-Oct.). As a result, soil nutrients in summer annual cropping systems are susceptible to losses in part because there are extended periods during each year when active plant uptake and soil cover are absent.
- 2) 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. 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 or can be produced on site. Many varieties of winter rye, triticale, and winter wheat can overwinter in the upper Midwest and continue growing in the spring. These winter-hardy cover crops must be killed with herbicides or tillage prior to planting corn or soybean. Oat, barley, spring wheat, and some rye, winter wheat, and triticale varieties are not winter-hardy in the upper Midwest. Because the non-winter-hardy small grains don't survive the winter, they don't need to be killed prior to planting the main crop, but they also don't produce as much shoot or root growth as winter-hardy small grains planted after full season grain crops. When the non-winter-hardy small grains are seeded in August after short-season crops or by overseeding, they can produce substantial biomass. Legumes are also excellent cover crops and they fix nitrogen as an added benefit. However, if nitrogen is available in the soil, legumes will take up N rather than fix it. Legumes, however, usually don't 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.

- 3) 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. Often the living mulches are perennial species and are maintained from year-to-year. 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. The main problem with living mulch systems is that the living mulch competes with the main crop for water and nutrients, which can reduce main crop growth and yield. Living mulch management to reduce stress on the grain crop during the growing season needs to be improved before they will be widely adopted.
- 4) Perennials are crops growing for multiple years from a single planting that would replace annual grain crops as the cash crop. Currently, the most common perennials found in agricultural systems in the Mississippi basin are forages (grasses and legumes) planted for hay, grazing, or pasture. Perennials also include tress 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.

B) What is the logic/process behind the practice?

- 1) Soluble nutrients in the soil, like nitrate and dissolved phosphorus, are susceptible to leaching losses. When plants are present and actively growing they remove soluble nutrients from the soil and reduce the amount of these nutrients susceptible to leaching losses. Because cover crops, living mulches, and perennial crops grow earlier in the spring and later in the fall than annual grain crops, they extend the season of active nutrient uptake beyond that of annual grain crops. For all three of these 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 and 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 the more established root systems scavenge nutrients from a larger soil volume.

- 2) 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, potassium, and pesticide losses and movement associated with soil erosion. Additionally, because these practices increase infiltration we would assume that they would reduce losses of dissolved phosphorus and pesticides in surface runoff.
- 3) Cover crops, living mulches, and perennial crops normally begin growth and water use earlier in the spring and then continue later into the fall than most annual grain crops. This extended period of water use normally reduces the volume of drainage water and thus, leaching losses of nitrate and dissolved phosphorus.

II) Potential

A) What are the estimated range and mid-range values for concentrations and losses of relevant nutrient forms for “current/conventional conditions?”

B) What are the estimated range and mid-range values for percent loss reductions given a defined “extent of practice application?”

- 1) Winter cover crops and rye cover crops specifically can reduce nitrate load and nitrate concentrations 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 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.
- 2) 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.

- 3) 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 is present for N uptake. Thus, perennials can reduce nitrate losses substantially, especially if no nitrogen fertilizer is applied, but nitrate losses depend to some extent on management. However, we assume that perennials will usually lose substantially less N than annual grain crops if they receive similar amounts of N fertilizer.
 - 4) 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. We estimate that losses of P and K to surface waters, which are linked to sediment losses, might be reduced by a similar amount relative to no-till. 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 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, 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.
- C) What is the predicted timing/delay and “seasonality” of reductions?
- 1) Most nitrate losses in drainage water occur during the periods of the year when the ground is not frozen, the soil is near saturation, and the main 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 (Ohio, Indiana, Illinois, Missouri), nitrate losses would occur in late fall, winter, and early spring, when most of the drainage occurs.
 - 2) Reductions of nitrate leaching are most likely to occur when the cover crop, living mulch, or perennial crop is actively growing and taking up nutrients. 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 growth in late fall. For cool season species that do not overwinter, such as oats, oilseed radish, oriental mustard, substantial fall growth can occur if they are planted early enough. Even the 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. Additionally, after cover crop death in winter or spring there is a carryover effect due to the nutrients taken up by the cover crop or immobilized during early stages of decomposition.
 - 3) Reductions of nutrient losses associated with erosion would most likely occur during

the winter and early spring, when runoff and erosion potential are greatest.

D) What is the degree of confidence of estimations (what are we more sure of, what do we know is only “directionally correct,” what do we need more information on)?

- 1) 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/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 or watershed scale; cultivar or species selection; time of planting and termination; seeding rate; and effect of multispecies mixtures.
- 2) 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. Magnitude of reductions depend on the 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 lessen the beneficial effects.
- 3) Confidence is extremely high that perennial crops would reduce nitrate losses compared with an annual grain crop, if the perennial crop is not fertilized with N fertilizers or manures. If a perennial crop receives N fertilizer or manures at rates comparable to annual grain crops, it is likely that the losses would still be less than that from the annual grain crop system.
- 4) Confidence is extremely high that all three of these practices would substantially reduce nutrient losses associated with erosion and overland water flow. Amount of reduction is dependent on amount of growth, surface cover, and anchorage of residues

III) Important Factors

A) How do site conditions (soils, topography, hydrology, climate) affect effectiveness in reducing nutrient losses?

- 1) Unknown based on research, but 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). 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 and K from upper and steeply sloped landscape positions or from areas of the field where overland water flow occurs.
- 2) Climate differences across the Midwest 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 and winter season, rather than in a short drainage period of April-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 Midwest region, which is warmer and wetter from Sept. to May, than in the northwestern part of the region, which is colder and drier from Sept. 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. Fig. 1 shows that predicted oat cover crop fall growth varies from 900 to 400 kg ha⁻¹ from southeastern Iowa to northcentral Iowa. Certainly, other factors would also limit growth, but in general we would expect that this general climate trend for fall growth would hold and that trends for spring growth of rye would be similar. 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 Midwest because these grasses are not actively growing during the periods of annual drainage.

- 3) Length of the cover crop growing season will also determine their 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 the next crop. This can result in relatively short cover crop growing seasons in the northern parts of the Midwest. Feyereisen et al. (2003) used modeling to predict that in most years a rye cover crop planted in southern Minnesota on Oct 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 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 Midwest.

B) How does a range of weather conditions affect effectiveness in reducing nutrient losses?

- 1) Weather affects cover crops, living mulches, and perennial crops effectiveness in reducing nitrate losses in two ways. First, if weather conditions are such that N mineralization and nitrate leaching are not favored then cover crops will not show much of an advantage over no cover crops. For example, in Iowa in some years little if any nitrate leaching occurs either because the soil profile was not recharged with water between harvest and planting of the grain crops, because the soil surface layers froze before recharge occurred, or because the soil remained so cold that N mineralization was greatly reduced. Second, if weather conditions limit cover crop, living mulches, or perennials establishment or growth, then their effectiveness will be reduced. Dry conditions, very cold conditions, or freezing of the soil surface in the fall or spring will limit cover crop growth, but will also limit N mineralization and nitrate leaching.

IV) Limitations

A) In terms of physical constraints, what percent of crop acres could benefit from this practice?

- 1) 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 also would 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 corn-soybean rotation may have even better opportunities for cover crops.
- 2) Living mulch systems are not ready for widespread adoption at this time in the upper Midwest in annual grain crop rotations, but have significant potential to be an important management option in the future.
- 3) In general, perennial crops are limited by demand, processing facilities, infrastructure, and markets. 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 20 to 30% of corn and soybean acres could be converted to perennial crops, if infrastructure, processing facilities, and markets were encouraged and supported.

B) In terms of cost constraints, what are the annualized costs expressed as \$ per pound reduction?

- 1) Cost estimates for living mulch - alfalfa establishment year (approx. every 3rd yr):
 - (a) Alfalfa seed 10 lb/ac at \$3.00/lb
 - (b) Custom rate for planting alfalfa with grain drill \$9.65/ac
 - (c) Generic Glyphosate \$20.00/gal, 2 appl./yr at rate of 0.33 qt/ac = \$3.32/ac/yr
 - (d) Custom rate for spraying herbicide 2 appl./yr at \$4.75/ac/appl = \$9.50/ac/yr
 - (e) Chopping/mowing living mulch 2 operations/yr at \$7.15/ac/op = \$14.30/ac/yr
 - (f) Total = \$121.05/ac/3 yrs = \$40.35/ac/yr
 - (g) Estimates are based on custom rates for field operations, which include fuel, labor, and machinery costs. Assuming that alfalfa would need to be reestablished every 3rd year, which may be too conservative. Assume 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.

- 2) Cost estimates for a perennial alfalfa crop establishment year (approx. every 3rd yr; may last longer):
 - (a) Alfalfa seed 15 lb/ac at \$3.00/lb
 - (b) Custom rate for planting alfalfa with no-till grain drill \$9.65/ac
 - (c) Generic Glyphosate \$20.00/gal, rate applied 1qt/ac = \$5.00/ac
 - (d) Custom rate for spraying herbicide \$4.75/ac
 - (e) Total = \$64.60/3 yrs = \$21.53/yr
 - (f) 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 as the cost of practice. Also, costs presented are for no-till and alfalfa is commonly established following tillage.
- 3) Cost estimates for a rye cover crop:
 - (a) Rye seed (bagged) \$6.00/bu plant at 1 bu/ac
 - (b) Custom rate for planting rye with no-till grain drill \$9.65/ac
 - (c) Generic Glyphosate \$20.00/gal, rate applied 1qt/ac = \$5.00/ac
 - (d) Custom rate for spraying herbicide \$4.75/ac
 - (e) Total = \$25.40/yr
 - (f) Estimates do not include any potential yield decreases of corn crop following a rye cover crop. Costs can be reduced by using bulk or bin seed and operator-owned equipment, and by assuming that glyphosate application is part of no-till program.
- 4) Costs per pound reduction of nitrate losses.
 - (a) 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, \$1.21 to \$0.48 per pound for perennial alfalfa, and \$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. Also, 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.

- C) In terms of production risks, will crop yields be reduced and/or be more variable?
- 1) Corn yields may be reduced following winter-hardy small grain cover crops that are killed immediately before corn planting. We believe that most of this yield reduction is caused by a “rotation effect” similar to that of continuous corn. Yield reduction can be minimized by killing small grain cover crop more than 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 are not reduced. Soybean yields are not reduced following any small grain cover crop unless low soil water content limits soybean germination and emergence. There is also a risk that the cover crop will not be completely killed the first time it is sprayed in spring. This would then require additional cost and time for additional applications as well as a risk of a yield reduction in the cash crop.
 - 2) 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.
 - 3) 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. For example, 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.
- D) Are there other limitations such as negative attitudes, lack of knowledge, additional equipment needed?
- 1) Many producers are not familiar with cover crops, living mulches, or perennials.
 - 2) 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.
 - 3) A grain drill is needed for planting many of cover crop, living mulch, and perennial species.
 - 4) Perennial forage or hay crops require equipment (mowers, rakes, balers) not required for corn and soybean production.
 - 5) Timeliness of cover crop and living mulch field operations in spring and fall will be limited by machinery and labor required for field operations associated with planting and harvesting of annual grain crops.
 - 6) 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.

- 7) Most producers in the upper Midwest don't see an immediate monetary benefit or reduction in costs from including cover crops and living mulches in their farming systems and have increased cost and labor to implement the practice.
- 8) There are limited markets for perennials such as harvested forages, such as alfalfa, orchardgrass, red clover, and smooth brome grass.
- 9) There is a great need for development of new perennial crops or new uses for well-known perennials.
- 10) There are limited seed sources and cultivars available for cover crops, living mulches, and some perennials. To our knowledge, there are no presently available cultivars that have been bred specifically for use as cover crops or living mulches.
- 11) There is a great need to quantify the nutrient loss reductions of these systems under a range of locations and growing conditions.
- 12) Cover crops, living mulches, and perennial crops do not enjoy the government risk protections that are provided program crops under federal farm policy.
- 13) 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.

V) Summary

- A) Are there any common misconceptions about this practice that need to be corrected?
- 1) 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, intensive grazing, or application of high rates of fertilizer or manure may result in substantial nutrient losses.
 - 2) There may be a misconception that pasture- or forage-based systems are always less profitable than annual grain crop systems.
 - 3) There may be a misconception that cover crops, living mulches, and perennial crops will always result in substantial reductions in nutrient losses when in some years nutrients losses are very low because weather, management, and main crop growth. Additionally, in some years the cover crop, living mulches, and perennial crops may fail to establish or grow poorly.
 - 4) 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.
 - 5) 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.

- B) Are there any potential positive or negative effects on other resources (e.g., soil, air, wildlife)?
- 1) Soil organic matter increases and soil quality improves from use of most cover crops, living mulches, forages, or perennial crops.
 - 2) These systems may also reduce some pest and disease pressures (nematodes, disease, insects, weeds) but may also increase others (rodents, insects, weeds).
 - 3) These systems increase plant diversity and provide food and cover for wildlife.
 - 4) These systems reduce wind and water erosion and sediment load to surface waters.
 - 5) These systems may improve water infiltration and help to remove excess water from the soil.
 - 6) These systems may reduce surface runoff, concentrated or channel water flow, residue transport, accumulation of water in low areas of fields, and flooding potential.
 - 7) These systems increase carbon sequestration in soils.
 - 8) Perennial crops provide new cropping and market options for producers.
 - 9) These systems could be used as a mechanism to make federal agricultural payments to producers in exchange for ecological benefits that would not conflict with World Trade Organization guidelines.
- C) What new information/research is needed to enhance the practice and/or accurately assess its benefits?
- 1) 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.
 - 2) Information is needed on the geographic range of these practices.
 - 3) Adaptation of water quality models to include cover crops, living mulches, and perennials is needed to estimate environmental benefits of these practices.
 - 4) Information is needed on when to kill the cover crop to optimize N uptake and N release for the cash crop, given that weather is variable and unpredictable.
 - 5) Information is needed on long term cycling and balance of N and C in these systems and whether N rates can be reduced in future due to improvements in SOM and N cycling.
 - 6) 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.

- 7) Screening, selection, and breeding programs are needed for new cultivars and species for use as cover crops, living mulches, or perennials.
- 8) Discovery and development of new oil, fiber, starch, or chemical products derived from perennial plants is needed.
- 9) Development of new production, harvesting, transporting, and processing technologies are needed for perennial crops.
- 10) Develop, strengthen, and support existing markets for products of perennial crops.
- 11) The economic viability of these systems need to be reevaluated in response to dramatic increases in fuel and energy costs.
- 12) Guidelines are needed for site/soil/landscape specific application of these practices to target areas in fields susceptible to nutrient loss.
- 13) Research is needed on minimizing the impacts on growth and yield of main cash crops by cover crops and living mulches.
- 14) Research is needed on the nutrient losses from intensively managed pastures and hay fields.
- 15) Research is needed on determining the effect of cover crops, living mulches, and perennial crops on P losses.
- 16) New management practices are needed to reduce the costs of implementing cover crops and living mulches.
- 17) Research is needed to evaluate cover crops and living mulches for biosuppression of nematodes, insects, and diseases.
- 18) Research is needed to evaluate cover crops, living mulches, and perennials for low input systems.
- 19) 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.
- 20) Quantification of the direct and indirect ecological benefits of these systems in diverse locations over a number of years is needed.

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.
- 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.

- Campbell, C.A., G.P. Lafand, R.P. Zentner, and Y.W. Jame. 1994. Nitrate leaching in a udic haploboroll as influenced by fertilization and legumes. *J. Environ. Qual.* 23:195-201
- 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.
- 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. American Society of Agricultural Engineers. St. Joseph, MI.
- Glover, J. 2003. Characteristics and impacts of annual vs. perennial systems. Proc. Sod Based Cropping System Conf. 20-21 Feb., 2003, Quincy, FL pp 1-6.
- Hall, J.K., N.L. Hartwig, and L.D. Hoffman. 1984. Cyanazine losses in runoff from no-tillage corn in "living mulch" and dead mulches vs. unmulched, conventional tillage. *J. Environ. Qual.* 13:105-110.
- Hartwig, N.L., and H.U. Ammon. 2002. Cover crops and living mulches. *Weed Sci.* 50:688-699.
- Jaynes, D.B., T.C. Kaspar, T.B. Moorman, and T.B. Parkin. 2004. Potential methods for reducing nitrate losses in artificially drained fields. p. 59-69. In R. Cooke. (ed.) Proceedings of the Eighth International Drainage Symposium, Sacramento, CA. 21-24 March 2004. ASAE, St. Joseph, MI.
- 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. *Bull.* 557. Ky. Agr. Exp. Sta. Lexington, KY.
- Kaspar, Z.C., J.K. Radke, and J.M. Lafflen. 2001. Small grain cover crops and wheel traffic effects on infiltration, runoff, and erosion. *J. Soil Water Conserv.* 56:160-164.
- Kladivko, E. J., Frankenberger, J. R., Jaynes, D. B., Meek, D. W., Jenkinson, B. J., Fausey, N. R. 2004. Nitrate leaching to subsurface drains as affected by drain spacing and changes in crop production system. *J. Environ. Qual.* 33:1803-1813.
- Langdale, G.W., R.L. Blevins, D.L. Karlen, D.K. McCool, M.A. Nearing, E.L. Skidmore, A.W. Thomas, D.D. Tyler, and J.R. Williams. 1991. Cover crop effects on soil erosion by wind and water. p. 15-22. In W.L. Hargrove (ed.) Cover crops for clean water. Proceedings of International Conference, Jackson, TN. 9-11 Apr. 1991. Soil and Water Conservation Society of America, Ankeny, IA.
- 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. *Soil Sci. Soc. Am. 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. p. 57-68. In W.L. Hargrove (ed.) *Cover crops for clean water*. Proc. of an international conf. April 9-11, 1991. Jackson, TN. Soil and Water Conservation Society, Ankeny, IA.
- 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. *Bull.* 466. Conn. Agr. Exp. Sta., New Haven, CT.
- 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 Group for the Assessment of Nitrogen Fertilizer Requirement. Netherlands Fert. Inst., Haren, The Netherlands.
- Owens, L.B., and J.V. Bonta. 2004. Reduction of nitrate leaching with haying or grazing and omission of nitrogen fertilizer. *J. Environ. Qual.* 33:1230-1237
- Pesant, A.R., J.L. Dionne, and J. Genest. 1987. Soil and nutrient losses in surface runoff from conventional and no-till corn systems. *Can. 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.
- Ryden, J.C., P.R. Ball, and E.A. Garwood. 1984. Nitrate leaching from grassland. *Nature (London)* 311:50-53.
- Sharpley, A.N. and S.J. Smith. 1991. Effects of cover crops on surface water quality. p. 41-49. In W.L. Hargrove (ed.) *Cover crops for clean water*. Proc. of an international conf. April 9-11, 1991. Jackson, TN. Soil and Water Conservation Society, Ankeny, IA.
- 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 US. *Agric. Ecosyst. Environ.* 77:203-210.
- Strock, J. S., Porter, P. M., Russelle, M. P. 2004. Cover cropping to reduce nitrate loss through subsurface drainage in the northern U.S. Corn Belt. *J. Environ. Qual.* 33:1010-1016.
- 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.

Web Sites

Wilson, R., J. Smith, and R. Moomaw. 1993. Cover Crop use in Crop Production Systems [Online]. NebGuide. G93-1146-A Cooperative Extension, Institute of Agriculture and Natural Resources, University of Nebraska-Lincoln. Available at <http://www.ianr.unl.edu/pubs/fieldcrops/g1146.htm> (Verified 8-16-05).

Sullivan, P. and S. Diver. 2001. Overview of Cover Crops and Green Manures [Online]. Appropriate Technology Transfer for Rural Areas (ATTRA), University of Arkansas, Fayetteville, AR. Available at <http://www.attra.org/attra-pub/covercrop.html> (Verified 8-16-05).

Sustainable Agriculture Network. 1998. Managing Cover Crops Profitably [Online]. Sustainable Agriculture Network. USDA-ARS, National Agricultural Library, Beltsville, MD. Available at <http://www.sare.org/publications/covercrops/covercrops.pdf> (Verified 8-16-05).

UC SAREP. 2002. Cover Crop Resource Page [Online]. University of California, Sustainable Agriculture Research and Education Program, Davis CA. Available at <http://www.sarep.ucdavis.edu/ccrop/> (Verified 8-16-05).

Table 1. Literature summary of percent reduction in 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 & Jensen, 1985	Denmark	Ryegrass	62
Martinez & Guirard, 1990	France	Ryegrass	63
Staver & Brinsfield, 1990	Maryland, U.S.	Rye	77
McCracken et al., 1994	Kentucky, U.S.	Rye	94
Wyland et al., 1996	California, U.S.	Rye	65-70
Brandi-Dohrn et al., 1997	Oregon, U.S.	Rye	32-42
Ritter et al., 1998	Delaware, U.S.	Rye	30
Kladivko et al., 2004	Indiana, U.S.	Winter wheat + less fert.	61
Jaynes et al., 2004	Iowa, U.S.	Rye	62
Strock et al., 2004	Minnesota, U.S.	Rye	13

Table 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 et al., 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

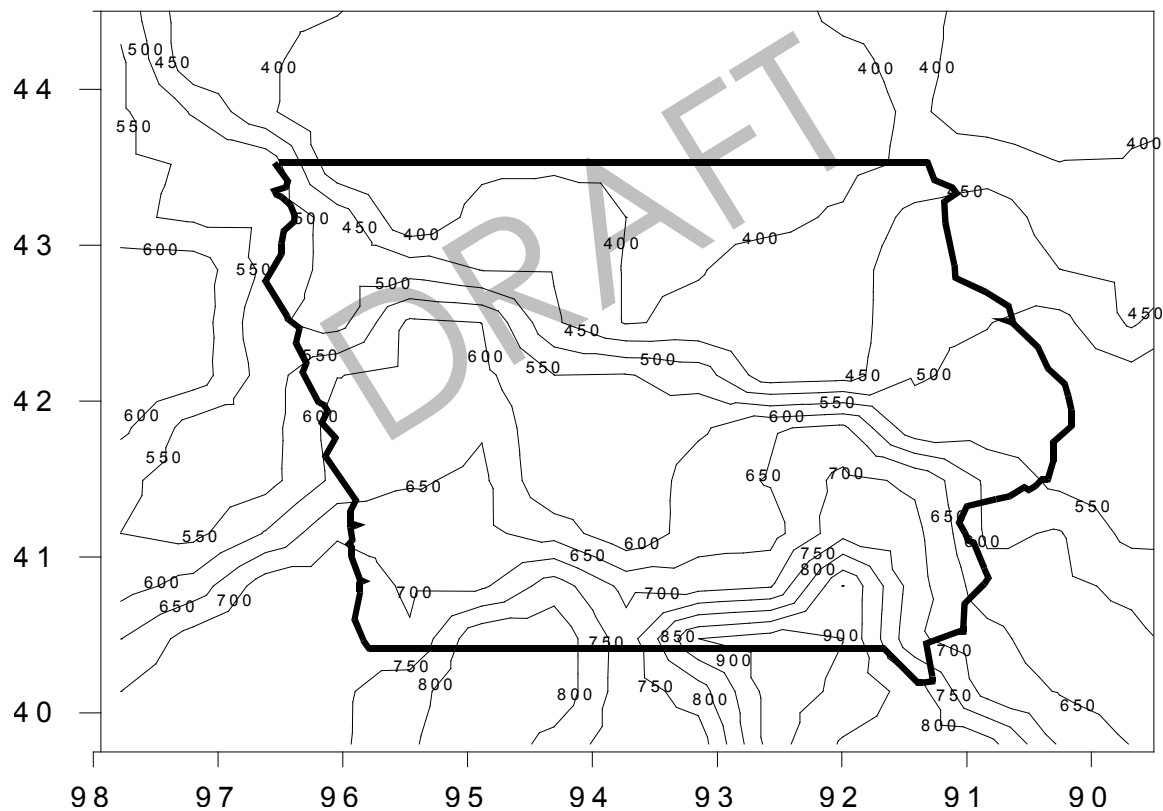


Figure. 1. Contour map for predicted fall production of oat shoot dry matter (kg/ha) vs latitude and longitude in Iowa.

Sustaining Soil Resources While Managing Nutrients

Dan B. Jaynes, National Soil Tilth Laboratory, Ames, IA

Douglas L. Karlen, National Soil Tilth Laboratory, Ames, IA

Introduction

The focus of most nutrient management studies has logically been on economic viability and water quality. In this paper, 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, as it is SOM that 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 it is the topsoil that erodes, these sediments are 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 defined 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 the Workshop, 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 CO₂ and soluble compounds that can be leached from the soil. On average, soil humus contains about 5.6% N and 56% C or 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 (Karlen et al., 1998; Jaynes et al., 2001; Webb et al., 2004) and watershed scales (David et al., 1997; Burkart and James, 1999; Goolsby et al., 1999). The estimates are made by assuming the conservation of mass, i.e.

$$\Delta \text{ inputs} - \Delta \text{ outputs} - \Delta \text{ soil residual mineral N} = \text{residual}$$

A residual > 0 indicates that inputs of N exceed losses from the field and 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 include the application of fertilizer and manure, N contained in rain and dry deposition, and N fixed by soybean. Outputs include N removed with the grain harvest and NO_3 in deep drainage and runoff (Table 1). 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 $\text{ha}^{-1} \text{yr}^{-1}$. For dry deposition, the approximation used by Goolsby et al. (1999) can be used where dry deposition equals 0.7X of 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

$$N_{\text{fixed}} = 81.1 * \text{yield} - 98.5$$

This compares to a more conservative estimate when soybean yields are $> 2.1 \text{ Mg ha}^{-1}$ used by McIsaac et al. (2002)

$$N_{\text{fixed}} = 33.4 * \text{yield}$$

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 averages 395 g kg⁻¹. 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 thus account for much 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. 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 was not accounted for in this partial N balance, but this is generally considered to be trivial. Denitrification is not explicitly accounted for 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 account for these latter losses, but it is important to note that all are important processes that can occur in the field and thus lower the residual mass balance.

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 this is 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 (1X, 2X, and 3X) 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 4-yr study with the economic optimum N rate being equal to about the 2X 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 NO₃ concentrations ranged from 11.4 mg L⁻¹ for the 1X treatment to 18.8 mg L⁻¹ for the 3X treatment. Using the MCL for NO₃ in drinking water, 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 NO₃ and NH₄ concentrations in precipitation for central Iowa. Dry deposition was estimated to be 0.7X of 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 of Barry et al. (1993). Grain removal of N was determined using measured grain yield and protein content. Drainage losses of NO₃ were computed using measured drainage volume and NO₃ 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 and runoff was observed only twice over 4 yr. Changes in soil mineral N were measured every fall within the top 1.2 m by collecting cores, extracting and analyzing for NO₃ and NH₄.

The 4-yr average partial N balance for each N rate in this study is shown in Table 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 NO₃ 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 1X to the 3X. 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 1X and 2X treatments means that more N was being lost from those systems than was being applied. This missing N had to come from some source unaccounted for in Table 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 3X rate do we see a residual N balance > 0, indicating that more N was being applied than was being removed. Thus, only for the 3X 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 to be responsible for a SOM increase, even with moldboard plowing, after 15 yr 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 fact, accounted for approximately 42% of the N budget for that period. However, it is important to remember that Table 1 shows only a partial N balance and we are not considering additional N loss pathways such as denitrification or volatilization, which would result in a smaller N balance than is shown, and the water quality implications of the N lost through the drainage water (13% for the Till Plain Study) must still be acknowledged and rectified.

In summary, whereas the economic optimum N fertilizer rate was approximately 134 kg N ha⁻¹ for the 4-yr 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 doesn't meet the definition because of high NO₃ concentrations leaving the field in tile drainage. Based on this assessment, the practice of deficit fertilization, although suggested as a viable alternative for solving NO₃ 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.

Table 1. Partial N mass balance for 4-yr rate study by Jaynes et al. (2001).

Fertilizer rate	----- N inputs -----			----- N outputs -----			Change in residual mineral N	N balance residual
	Total fertilizer applied	Total wet and dry deposition	Total fixed	Total grain removal	Total drainage loss	Total runoff [†]		
----- kg ha ⁻¹ -----								
1X	144	43	395	522	119	0	6	-55
2X	289	43	397	590	142	0	13	-26
3X	414	43	394	606	195	0	-7	47

[†]Not measured, but little runoff observed during 4-yr period.

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. 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 N₂ 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 NO₃. 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 NO₃ concentration for the conventional treatment was 22.4 mg N L⁻¹, well above the MCL for drinking water. The flow-weighted average NO₃ 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 good cover crop was well established. The flow-weighted average NO₃ 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 NO₃ losses in year where good cover crops could be established. The bioreactor treatment was sustainable from a water quality perspective as the NO₃ 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 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 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 $\gg 0$, 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 biomass 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 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 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. Installing 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.

Table 2. Partial N mass balance from 5-yr rate study by Jaynes et al. (2004).

Treatment	-----N Inputs-----			-----N Outputs-----			Change in residual mineral N	N balance residual
	Total N fertilizer applied	Total wet and dry deposition	Total fixed	Total grain removal	Total drainage loss	Total runoff [†]		
	----- kg ha ⁻¹ -----							
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

[†]Not measured, but little runoff observed during 4-yr period.

Liquid manure Traditionally on diversified farms, animal manure was 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 became 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 6-yr study conducted on Iowa Till Plain soils near Nashua, IA 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 $\text{ha}^{-1} \text{yr}^{-1}$ during the 6-yr study (Karlen et al., 2004). When averaged for continuous corn, drainage loss accounted for 16% of the applied N, 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 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 2-yr corn-soybean rotation, respectively. 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. Summing inputs and outputs for this manure study shows a substantial residual N balance for continuous corn (Table 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 6 yr of the study. Perhaps measuring one of the more active C/N pool such as particulate organic matter (Cambardella and Elliot, 1993) would be more sensitive to changes in SOM content.

Table 3. Partial N mass balance for a 6-yr swine manure study by Karlen et al. (2004).

Treatment	-----N Inputs -----			-----N Outputs -----			Change [‡] in residual mineral N	N balance residual
	Total N [§] applied	Total wet and dry deposition	Total fixed	Total grain removal	Total drainage loss	Total runoff [†]		
	----- kg ha^{-1} -----							
Continuous corn	958	63	0	510	156	0	82	273
Corn phase [‡]	794	63	0	600	84	0	116	57
Soybean phase [‡]	0	63	1058	1092	150	0	2	-123

[†]Not measured, but very little runoff was observed during the 6-yr study.

[‡]Both phases of a corn-soybean rotation were present each year

[§]A 2% loss from volatilization was assumed for liquid injection.

[¶] Estimated 6-yr values based on 1996, 1997 and 1998 measurements (Bakhsh et al., 2001)

Limitations

As verified by our inability to detect SOM differences in soil samples from these three sites, assessing the impact of nutrient practices on SOM content and long-term productivity of soil is difficult because we are trying to measure small changes in a large quantity. 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). Whereas 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. In the examples used here, the negative N mass balances for the 1X treatment in Table 1 and the corn/soybean rotation under liquid manure application in Table 3, are equivalent to about a 0.2% yr⁻¹ loss in SOM from the top 20 cm of the profile. Conversely, the positive N balances for the cover crop in Table 2 and the continuous corn with liquid manure application in Table 3, represent increases in SOM of about 0.2% and 0.9 % yr⁻¹, respectively. While small in terms of our current ability to directly measure, these changes represent about a 5% loss in SOM over 30 yr for the first two, and a gain of 5% and 26% in 30 yr for the latter two. The losses may be even more important for soil function, as these losses are primarily from the labile pool, which is the most chemically, biologically, and physically active SOM pool.

Thus, 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.

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. *Soil Sci. Soc. Am. 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.
- Francis, D.D., J.S. Schepers, and A.L. Sims. 1997. Ammonia exchange from corn foliage during reproductive growth. *Agron. J.* 89:941-946.
- 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. NOAA Coastal Ocean Program Decision Analysis Series No. 17. NOAA Coastal Ocean Program, Silver Springs, MD.
- Hatfield, J.L., and B.A. Stewart (eds.) 1998. *Animal Waste Utilization: Effective use of manure as a soil resource.* Sleeping Bear Press, Chelsea, MI. 320 p.
- Heichel, G.H. 1987. Legume nitrogen: Symbiotic fixation and recovery by subsequent crops. p. 63-80. In: Z.R. Helsel (ed.) *Energy in Plant Nutrition and Pest Control.* Elsevier, Amsterdam.
- 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., Kaspar, T. C., Moorman, T. B., and Parkin, T. B. 2004. Potential methods for reducing nitrate losses in artificially drained fields. p. 59-67. *Internat. Drainage Symp.* Sacramento, CA. Mar. 21-24, 2004.
- Karlen, D.L., Ajay 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 & Till. Res.* 48:155-165.
- Karlen, D.L., C.A. Cambardella, and R.S. Kanwar. 2004. Challenges of managing liquid swine manure. *Appl. Eng. 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. Iowa State Univ. Extension. Ames, IA.
- 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. NOAA Coastal Ocean Program Decision Analysis Series No. 19. NOAA Coastal

Ocean Program, Silver Springs, MD.

Parkin, T.B., and J. Meisinger. 1989. Denitrification below the crop rooting zone as influenced by surface tillage. *J. Environ. Qual.* 18:12-16.

Waksman, S.A. 1938. *Humus*. Williams and Wilkins Co., New York.

Webb, J. S. Ellis, R. Harrison, and R. Thorman. 2004. Measurement of N fluxes and soil N in two arable soils in the UK. *Plant and Soil* 260:253-270.

DRAFT

Field-Scale Tools for Reducing Nutrient Losses to Water Resources

Larry G. Bundy, Department of Soil Science, University of Wisconsin-Madison

Antonio P. Mallarino, Department of Agronomy, Iowa State University, Ames

Introduction

Phosphorus (P) loss in runoff from cropland is a water quality concern because this P often promotes algae and other vegetative growth in 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 which 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).

Phosphorus entry into natural waters from point sources such as industrial discharges and municipal sewage treatment facilities are currently regulated under the provisions of water quality protection legislation. Nonpoint or diffuse sources of 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 water resources.

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 paper will be on addressing local surface water quality concerns.

Specifically, this paper will focus on 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 USA, 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, for use as planning tools for agronomic P management. The high level of activity in development of P indices in the USA is largely in response to USDA and/or EPA proposals that all animal feeding operations (AFO's) 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 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 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.

Of the alternative strategies proposed in the 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, and fertilizer or manure P application rates and methods.

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.

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. Multiplying the P source loss potential value by the corresponding P source 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.

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 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 1) and transport (Table 2) factors typically included in P indices along with the weighting factors assigned to various components.

Several states in the North Central Region of the USA 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 USA, North Carolina has developed a P index using a generally similar modeling approach (NC PLAT Committee, 2005). These indices are sensitive to the need to utilize input data that is available or easily obtainable by users and are much less data intensive than more complex process-based research P loss models. The P indices developed in Iowa (Mallarino et al., 2002) (<http://www.ia.nrcs.usda.gov/technical/Phosphorus/phosphorusstandard.html>), Minnesota (Minnesota Phosphorus Site Risk Index, 2005), Missouri (<http://www.nmplanner.missouri.edu/>), and Wisconsin (<http://wpindex.soils.wisc.edu/>) (<http://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 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 soluble P (SP) separately and sum these values. The separate estimates of PP and SP 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, modification in cropping systems and tillage to control sediment loss would likely reduce overall P loss. Alternatively, a high SP contribution to the PI 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 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 and Minnesota while a process to account for buffer influences is under development in the Wisconsin index. 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 applications 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 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 resources 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.

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 sub-watershed scale, and these measurements were compared with the Wisconsin P index values for the same areas. The 21 sites represented 18 fields on 7 farms in 4 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 (4 incorporated, 7 surface) in the monitoring year in 11 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.

Data in Figure 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 (Figure 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 3 shows the relationship between index values calculated using the Pennsylvania P index and measured annual P runoff loads from the same 21 locations as used in Figures 1 and 2. Comparison of Figures 1 and 3 indicate 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.

Summary and Conclusions

Field-scale tools for predicting 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. Currently used field-scale

tools for assessing the risk of P losses to water resources include mainly soil test P and P indices. The extent of loss identified by these tools is expressed as a categorized risk level (eg. 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. Additional validation of field-scale tools against measured annual P losses is needed.

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.
- Heathwaite, A.L., A. Sharpley, M. Bechmann, and S. Rekolainen. 2005. Assessing the risk and

- magnitude of agricultural nonpoint source phosphorus pollution. p. 981-1020. In J.T. Sims and A.N. Sharpley (eds.) Phosphorus: Agriculture and the environment. Agronomy Monograph no.46, ASA-CSSA-SSSA, Madison, WI.
- 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. 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, Department of Soil, Water and Climate, University of Minnesota, St. Paul, MN, May, 2005.
- NC PLAT Committee. 2005. North Carolina phosphorus loss assessment: I. Model description and II. Scientific basis and supporting literature, North Carolina Agricultural Research Service Technical Bulletin 323, North Carolina State Univ., Raleigh, NC.
- Natural Resource Conservation Service (NRCS). 1999. General Manual, 190-GM, Issue 9, Part 402-Nutrient Management. April, 1999.
- 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.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. *Journal of Soil and Water Conservation* 58:137-152.
- 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.

- 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. University Park, Pa.: Pennsylvania State University, College of Agriculture. Publication Cat. US 180 5M3/03PS4591. Available at: http://panutrientmgmt.cas.psu.edu/main_related_programs.htm.

Table 1. The Pennsylvania P Index: Source factors (Weld et al., 2003).

Contributing Factors	Risk Levels				
	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 more than 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 ₂ O 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 ₂ O 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 2. The Pennsylvania P Index: Transport factors (Weld et al., 2003).

Characteristics	Risk Levels				
Soil Erosion	Risk value = Annual soil loss = _____ tons acre ⁻¹ year ⁻¹				
Runoff Potential	Very Low	Low	Medium	High	Very High
	0	1	2	4	8
Subsurface Drainage	None		Random		Patterned ^[a]
	0		1		2
Contributing Distance	>500 ft.	500 to 350 ft.	350 to 250 ft.	250 to 150 ft.	<150 ft.
	0	1	2	4	8
Transport sum = Erosion + Runoff potential + Subsurface drainage + Contributing distance					
Modified Connectivity	Riparian buffer		Grassed waterway		Direct connection
	Applies to distances <150 ft.		or none		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					

DRAFT

Table 3. General structure of P-indices in Iowa, Minnesota, and Wisconsin.

State	P-index formulae
Iowa	PI = PP + SP + Subsurface P
Minnesota	PI = PP + rainfall SP + snowmelt SP
Wisconsin	PI = (PP + SP + event losses) × TP delivery ratio
PI = P index value; PP = particulate P; SP = soluble P; TP = total P	

Table 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	Dis. field to stream	Distance & slope on TP
Buffer factor	Buffer width	Sediment trap factor	Under development
Sediment P content	Calc. from soil test P	Calc. from soil test P & organic matter	Calc. from soil test P & organic matter
Adjust. of PP for recent P additions	None	Optional based on soil P buffer cap.	Soil test P adjusted based on buff. cap.
PP enrichment factor	1.1-1.3 depending on mgmt. practices	None	Under development
Dissolved/soluble P:			
Runoff volume	From runoff curve nos. & % of precip.	From runoff curve nos. & % of precip.	Ave. precip., runoff curve nos. + winter runoff
Dissolved P in runoff	From soil test P	From soil test P	Soil soluble P from soil test P x extraction efficiency
Adjust. of DP for recent P additions	From buffer cap. & method & time factor	Optional based on soil P buffer cap.	Soil test P adjusted based on buff. cap.
Dissolved P from surface P applications	Included in adjust. of DP for recent P additions (above)	From amount of P applied, timing, and tillage	Included in soil test P adjust. (above) + 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

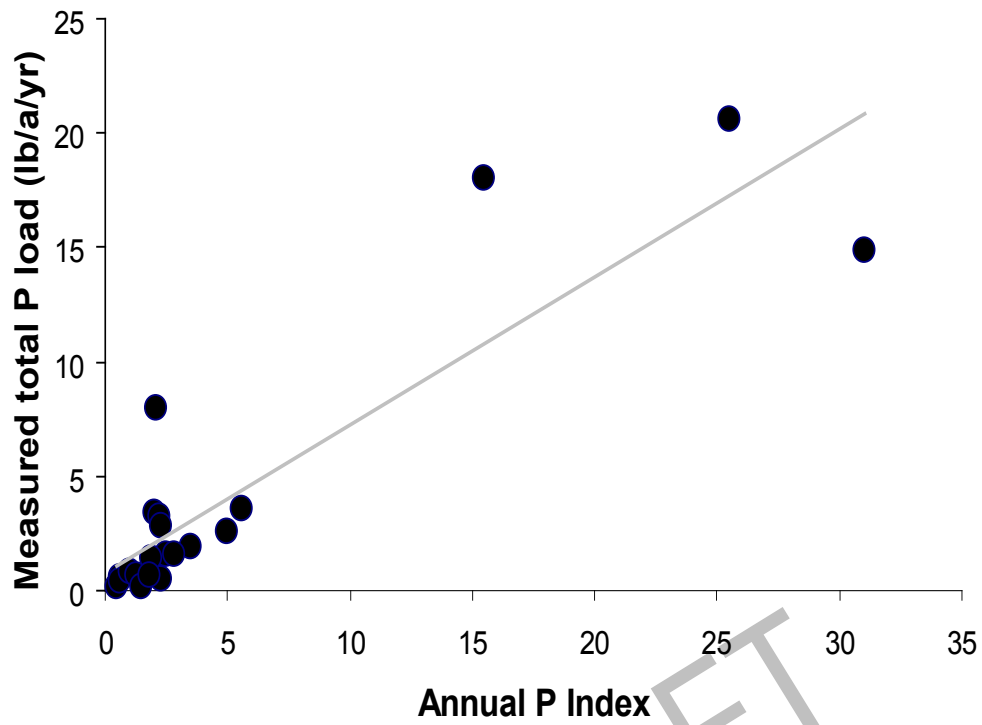


Figure 1. Relationship between measured annual runoff P loads and Wisconsin P index values for 21 field locations in Wisconsin.

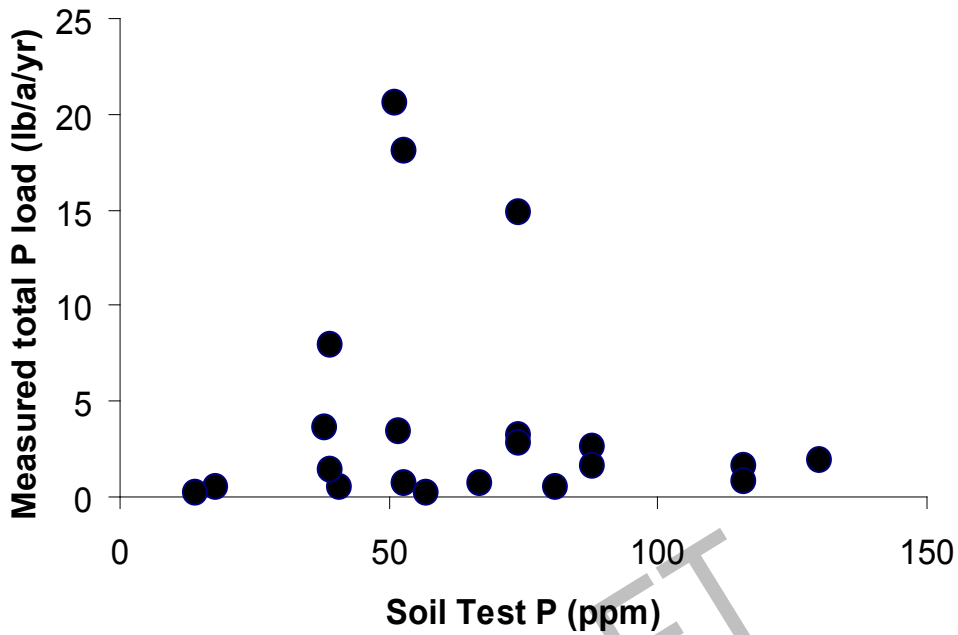


Figure 2. Relationship between measured annual runoff P loads and Bray P-1 soil test values for 21 field locations in Wisconsin.

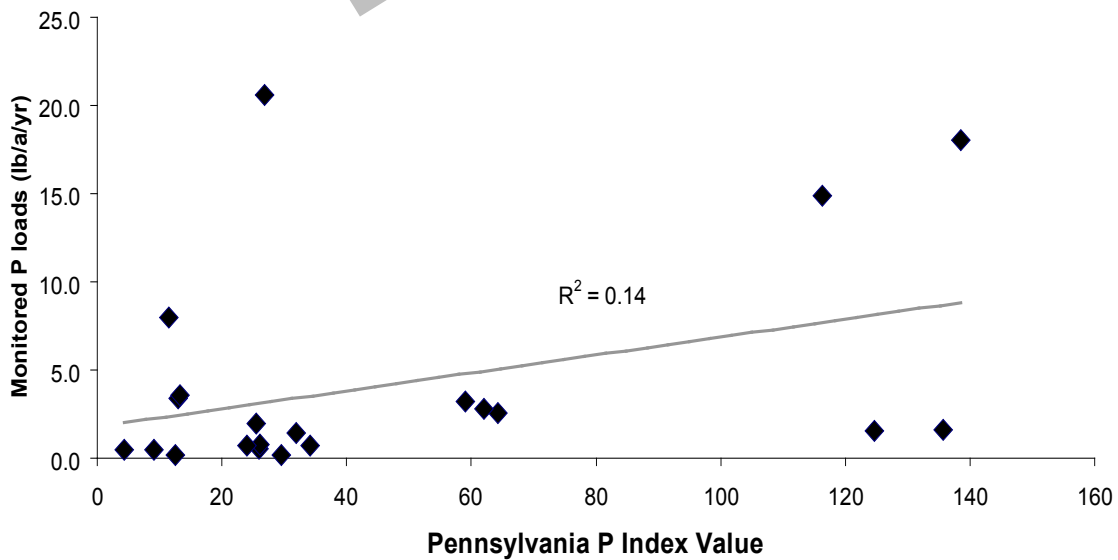


Figure 3. Relationship between measured annual runoff P loads and P index values calculated using the Pennsylvania P index for 21 field locations in Wisconsin.

Watershed-Scale Tools

Pete Nowak, Professor and Chair, Gaylord Nelson Institute for Environmental Studies, University of Wisconsin-Madison

John Norman, Professor, Department of Soil Science, University of Wisconsin-Madison

Dave Mulla, Professor and W.E. Larson Chair for Soil & Water Resources, Department of Soil, Water and Climate, University of Minnesota

Introduction

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 paper are the lessons learned through both 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 resource management objectives. A tool in the broadest sense is a means to accomplish a desired end, but in this context refers to the analytical, mechanical, structural and behavioral techniques used in pursuing conservation objectives. It is important to emphasize that tools are more than the practices installed or employed in the watershed. The changing nature of how we think about degradation in a watershed represented by the analytical processes and procedures used to characterize these situations are also tools.

Our thesis is that the effectiveness and efficiency of any watershed tool will be directly related to the spatial congruence between the objectives of a policy or program calling for the use of tools, the spatial dimensions of the watershed tools themselves, and analytical capacity to characterize and understand salient processes and situations across space within the watershed. Achieving the greatest effectiveness in the most efficient fashion in the use of watershed tools will occur when there is scalar congruence of these three components.

Discussion

The core principle behind the spatial congruency thesis just proposed is that degradation within a watershed is not random. It is spatially and temporally patterned. The spatial pattern that emerges is dependent on the interaction between the appropriateness of a behavior or activity and the relative vulnerability of the specific location where this interaction occurs. A critical, and as of now unmet, research need is an assessment of the optimal spatial scale to examine this interaction. Both the appropriateness of a behavior or activity and the vulnerability of a location will vary across time due to short and long-term climatic variation, changes in agricultural technologies, and our increasing abilities to monitor and measure forms of degradation. Another complication are the trade-offs that occur between forms of degradation. Minimizing one form of degradation in a particular time or location may exacerbate other forms of degradation.

Addressing this dynamic setting is only part of the challenge for an effective and efficient watershed tool. An overwhelming proportion of watershed tools are not implemented independently. These tools are typically employed as part of policies or programs¹ that are designed to address a specific form of degradation. These tools are promoted and implemented as part of a policy and program. Consequently, another challenge emerges associated with the spatial specificity of the policy and program. Attempts to focus conservation efforts on specific geographical areas to enhance resource management objectives are referred to as targeting. Spatial specificity of a conservation program is often linked to a measure of economic efficiency resulting in a program goal of targeting benefits in an attempt to achieve the greatest levels of conservation for the least amount of expenditures allocated. Major land resource areas, hydrologic units such as watersheds, farms, fields, and portions of fields such as those in a riparian area have all be the focal point of targeted conservation programs.

There is an extensive scientific literature that describes, analyzes and critiques the various tools that can be used in watersheds (see Mulla, Kitchen and David paper). There are three common dimensions across this analysis. First, regardless of the form of degradation, disproportionality is the norm. That is, a small proportion of any watershed contributes a disproportionate share of the degradation. Second, most watershed tools are incapable of optimizing across forms of degradation. While there are ample watershed tools available for addressing, for example, sedimentation, nitrogen leaching, or wildlife habitat, there are few practical tools available capable of addressing multi-media forms of degradation. Third, all available tools, implicitly or explicitly, are impacted by uncertainty. The stochastic variation found in climatic processes, behavioral patterns, or cross-scale nonlinearities has required the resulting uncertainty to be addressed through simplifying assumptions.

The Wisconsin Buffer Initiative

The Wisconsin Buffer Initiative (WBI) represents an attempt to address the congruency thesis proposed earlier. That is, there is an explicit attempt to match the spatial focus of the program with the hydrologic units targeted and with the analytic procedures and processes (i.e., tools) used within these targeted areas. Consequently, the WBI is an integrated, multi-scale initiative moving from the state to the sub-field and back up-scale again. Uncertainty is explicitly addressed through use of an adaptive management framework. Adaptive management is a process of learning and adapting based on the consequences of earlier actions. It is a process designed to address incomplete understanding of cause and effect relations, and “surprises” that may emerge due to changing circumstances.

Wisconsin’s nonpoint agricultural pollution abatement policy was the context for the WBI. Controversy emerged over the role of riparian buffers within the specific programs of this policy. Specifically, some argued for standard-width riparian buffers to be mandated for all the rivers and streams in Wisconsin, while others argued that existing federal and state programs that promote buffers was adequate. This standoff resulted in the state approaching the University of Wisconsin asking about the best available science regarding riparian buffers. For the last three years scientists with the university have been working with representatives from all the vested interests to determine the appropriate role for riparian buffers on the Wisconsin landscape.

¹ The term policy is used to represent a governmental process designed to achieve some desired outcome. A program can be viewed as the specific plans of action and responsibilities that emerge as a result of a policy.

Early meetings of the WBI were spent discussing program expectations, available tools, and data availability. A number of coordinated research efforts were initiated to identify gaps in the existing knowledge regarding the role of riparian buffer technology. Three decisions were made early in this process. First, it was agreed that vegetative strips adjacent to streams or rivers were not adequate to address the complex forms of degradation occurring across the Wisconsin landscape. The WBI therefore adopted the term buffer technology to represent the compliment of tools (i.e., a systems approach) that 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 areas where tools needed to be applied. The second decision was that fiscal constraints would prevent implementation of this approach on a wide scale basis. Consequently, the WBI explicitly addressed the scale question at the state, watershed, field, and sub-field levels in establishing priorities of where buffer technology needed to be implemented. Third, a decision was made that the current recommendations are based on the best available science at this time. Yet this was not deemed sufficient. An adaptive management framework is used to provide meaningful feedback so that both spatial recommendations and applications of buffer technology can be adjusted accordingly.

These early decisions resulted in three general questions that guided both research and the discussion of the recommendations. These questions were:

1. How to identify the hydrologic units most likely to show demonstrable improvements with investment in buffer technology?
2. What types of tools can be developed that can be employed at the local level to assist in identifying sub-watersheds and fields where buffer technology is an effective option?
3. How do we develop techniques for determining the optimal placement and configuration of buffer technology on designated landscapes?

The remainder of this paper will describe the responses developed to date regarding these three questions.

Identifying Watersheds that will be Responsive to Buffer Technology

Watersheds vary in size due to their nested nature. Selecting the appropriate spatial scale to be both a focal point of policy, and appropriate to the tools be employed is a critical decision. Successfully implementing a remedial program in a large watershed will produce a large number of environmental benefits, but will be very expensive for a number of reasons. Implementing a program in a small watershed will be very cost-effective, but environmental gains will be highly variable and tending to be minimal. At the coarse scale Wisconsin has 42 USGS 8-digit HUCs watersheds with an average size of 3400 km². The Wisconsin Priority Watershed Program was based on subdividing these 42 watersheds into 334 watersheds each of an average size of approximately 434 km². It was determined that the WBI would be based on all the third order and some fourth order streams in Wisconsin. This results in approximately 1600 watersheds with an average size of approximately 60 km². This decision was based on selecting a size where it would be feasible to determine if the watershed responded to the implementation of buffer technology, 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.

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. For the WBI three different criteria were selected to screen the approximate 1600 potential watersheds. These were sediment and nutrient loads, protecting and enhancing native biological communities, and third, the trophic status of lakes, reservoirs or impoundments down-gradient from the watershed. At present all three were given equal weight, but the WBI process may decide to apply differential values at a later date.

A spatially specific analysis was then conducted on each of these criteria for each of the 1600 watersheds. A wide range of data sources (federal, state, local, scientific literature, NGOs, etc) was sought out to assess the potential to reduce nutrient and sediment loading, enhance the biotic potential, and trophic status. Further decisions were made as part of the WBI process to focus on watersheds with the greatest potential for nutrient and sediment loading, where buffer technology could enhance marginal conditions relative to native biological communities, and where lakes, reservoirs or impoundments were close to a “tipping point” of moving to a hyper-eutrophic status. In summary, judgments were made that watershed tools should only be applied to situations where there was a good probability that the tool would generate a desired response. Waters that were significantly degraded were judged to be beyond the capabilities of the watershed tools that were being implemented as part of this program. They received lower weights in this process than waters that were marginal (slightly degraded or currently in good condition but with a trend toward degradation).

Weighting of each individual WBI watershed was based on the following calculations:

1. Predicting nutrient and sediment loads based on a regression model developed around land use and watershed loading data derived from USGS and other monitoring sources. In each of these watersheds there was an attempt to quantify sources where buffer technology would have little impact such as bank slumping and stream bed erosion.
2. The potential response of biological communities to buffer technology implementation. This was calculated 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 assessing biological community response potential to sediment reductions.
3. Calculating the potential for attenuation of lake eutrophication based on current lake conditions, monitoring data, and the likely response to reductions in phosphorus from contributing streams.

Each of these three ranking criterion are integrated in a GIS layer representing a composite ranking for each of the WBI watersheds. This allows for a rank order listing of all the watersheds in the state from those most likely to respond to buffer technology to those least likely to respond.

It can be argued that the analytical procedures used to produce this ranked list of watersheds increased the congruence of the policy and program to salient processes occurring in these watersheds. Political and administrative decisions will have a clear framework to follow in allocating scarce fiscal and personnel resources. The next step in enhancing the spatial congruence will be to specify implementation procedures within the watersheds that are selected for implementation as part of the political and administrative decision process.

What Planning and Implementation Tools can be used at the Local Level?

Because of the diversity of interests associated with the WBI process, it was determined that a series of conditions should be addressed in deciding what tools need to be deployed in the selected watersheds. It was decided that whatever tools are selected they 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 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.

These decisions led to seeking out data bases 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.

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 is also universally available. Digital land use data was 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 will be more variable as USGA data is only available in selected locations and the monitoring that accrued as a result of the previous Nonpoint Priority Watershed Program is also variable.

All this data is used to determine priority areas within the selected watersheds. That is, even though a watershed is selected as part of the implementation process, not all areas in the watershed would be targeted for implementation efforts. This second level of spatial targeting is an important dimension of the WBI process. Limiting the area within the watershed allows local staff to focus resources and efforts on those areas where there is the highest probability of degradation.

A map showing these high probability areas will be generated using the DEM, SSURGO, and land cover information. This initial map will be reviewed by local conservation staff that may be aware of local efforts or situations that are not represented in the initial representations of potential priority areas.

Following adjustments to this map, the priority areas will be assessed using a field-scale assessment tool (see Bundy and Mallarino paper). In Wisconsin this assessment will be built around the SNAP+

planning tool. This tool incorporates a phosphorus index with erosion calculations to provide a series of management options to 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 phosphorus index (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. Only on those fields with a PI greater than 6 will buffer technology 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 6 within buffer technology being brought into play.

Placement and Design of Buffer Technology

Only in those circumstances where the PI is greater than 6 and changing current farming practices will not reduce this value to an acceptable range will buffer technology come into play. As stated earlier, the buffer technology will be designed from riparian area to upland areas to fully incorporate the contributing area. The importance of this contributing area was developed in the WBI process through the application of the Precision Application Landscape Modeling System (PALMS). PALMS research demonstrated that standard width buffers are highly vulnerable to being breached by concentrated flow in select locations along the buffer. Consequently, buffer technology needs to be designed to prevent concentrated flow from developing in a contributing area. This will require more of a systems perspective moving back up across the landscape, even considering neighboring fields.

The design and placement of buffer technology is based on diffusing water and energy on the higher areas of the landscape rather than trying to control and mitigate this energy in the riparian zone. Buffer technology 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 tools is highly site-specific.

Conclusions

The field-specific design and placement of buffer technology is a direct result of policy process that asked where across the Wisconsin landscape riparian buffers were needed to achieve water quality objectives. It would be easy to view this buffer technology as the optimal watershed tool, the question addressed by this paper. Yet buffer technology, this systems approach to address the contributing area on a field(s) adjacent to a stream, is only a portion of the watershed tools that were needed in the WBI process. Important lessons were learned in this scientific and political process, i.e., civic science.

Perhaps the most important lesson is that one does not design watershed tools and then go looking for an application situation. Our thesis is that the process needs to be designed to optimize the spatial congruence between the objectives of a policy or program, dimensions of the tools, and the analytical capacity to characterize and understand salient local processes and situations within watersheds. Getting to the characterization of the appropriate watershed tool began with a political

process that asked where and how riparian buffers should be employed in Wisconsin to achieve water quality objectives. Determining what type of intervention was needed was also a political process where scientists played an important role in providing analyses to allow targeting down across several spatial scales. The next political decision, also led by scientific analysis, was that ribbons of riparian vegetative strips were not sufficient to achieve the water quality objectives. The final political decision will be acceptance of 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. It also needs to be considered in the selection or design and use of any watershed tool.

The second lesson learned in the WBI process is that we will probably never create any watershed tool that does not contain uncertainty or significant error. Hence the needs for an adaptive management approach to any application of these tools. Implementation, monitoring, and adjustment is a process determining our ability to answer questions such as where is intervention needed, to what extent did we achieve our goals, or how did that surprise or extreme event influence the performance of our tools? Adaptive management is based on our ability to acknowledge our ignorance, and design our tools and feedback mechanisms accordingly.

Members and Participants in the Wisconsin Buffer Initiative

1. Bill Pielsticker, Trout Unlimited
2. John Norman, UW-Madison, Soil Science
3. Mike Dahlby, Wisconsin Association of Land Conservation Employees
4. Jeff Maxted, UW-Madison, Center for Limnology
5. Denny Caneff, River Alliance
6. Steve Ventura, UW-Madison, Soil Science
7. Gene Hausner, USDA NRCS
8. Richard Gordor, Farm Bureau Federation
9. Dennis Frame, Discovery Farms
10. Susan Butler, USDA FSA
11. Keith Foye, WI DATCP
12. Bob Oleson, Wisconsin Corn Growers Association
13. Christine Molling, UW-Madison, Soil Science
14. Larry Cutforth, USDA-FSA
15. Todd Ambs, Wisconsin Department of Natural Resources
16. Bill Hafs, Wisconsin Association of Land Conservation Employees
17. Gordon Stevenson, Wisconsin Department of Natural Resources
18. Jake Vander Zenden, UW-Madison, Center for Limnology
19. Matt Diebel, UW-Madison, Center for Limnology
20. Paul Kaarakka, UW-Madison, Soil Science
21. Karen Talarczyk, Green County Land Conservation Department
22. Todd Jenson, Green County Land Conservation Department

23. Jim Jolly, Brown County, Land Conservation Department
24. Jason Thomas, Green County USDA-NRCS
25. Rick Klemme, UW-Extension
26. Timm Johnson, Wisconsin Agricultural Stewardship Initiative
27. Tom Cox, UW-Madison, Applied Agricultural economics
28. Dean Dornink, Professional Dairy Producers of Wisconsin
29. Fred Madison, UW Soils, Discovery Farms
30. Kevin McSweeney, UW-Madison, Soil Science
31. Paul West, The Nature Conservancy
32. Larry Bundy, UW-Madison, Soil Science
33. Bob Oleson, Wisconsin Corn Growers Association
34. Don Baloun, USDA-NRCS
35. K.G. Karthikeyian, UW Madison, Biological Systems Engineering
36. Paul Zimmerman, Wisconsin Farm Bureau Federation
37. Rebecca Baumann, Wisconsin Land and Water Conservation Association
38. Dave Hogg, UW-Madison, Dean, CALS
39. Paul Miller, UW Madison, Biological Systems Engineering
40. Frank Scarpace, UW-Madison, Civil and Environmental Engineering
41. Laura Ward Good, UW-Madison, Soil Science
42. Pete Nowak, UW-Madison, Gaylord Nelson Institute for Environmental Studies

DRAFT

Evaluating the effectiveness of agricultural management practices at reducing nutrient losses to surface waters

D. J. Mulla and A. S. Birr, Univ. Minnesota, St. Paul, Minnesota

N. Kitchen, USDA-ARS, Columbia, Missouri

M. David, University of Illinois, Urbana, Illinois

Executive Summary

Water quality impairments are widespread throughout the upper Mississippi River basin due in large part to agricultural production practices. 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:

- 1) Variability in weather, runoff and drainage lead to highly variable nutrient and sediment exports from one day or month to another and one year to another. Therefore, it is difficult to know how much of a change in nutrient and sediment export has resulted from a change in management, without long-term data. Long-term water quality data sets of sufficient monitoring intensity are generally not available, and short-term (1 to 5 year) 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.
- 2) 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.
- 3) Long lag times in response to changes in management. Because of large soil pools of N and P, response to implementation of management practices can take many years, perhaps as many as 5 to 10 years. 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.
- 4) Sparse non-targeted implementation of most management practices at watershed scales. 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.
- 5) Modeling limitations including uncertainty in many parameters (e.g., soil hydraulic properties, denitrification, mineralization rates, biological N₂ 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.

Introduction

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. A variety of ways to reduce the area of the hypoxic zone by 30% have been proposed. 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. 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. Small amounts of sediment, phosphorus or nitrogen lost from a large number of agricultural fields can collect in lakes or at the mouths of large watersheds 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 the most effective, technologically, and economically feasible means of preventing or reducing the pollutant load generated by nonpoint sources to a level that meets water quality goals (USEPA, 1980). Typically, a BMP both reduces pollutant load and maintains crop or animal productivity. Methods are needed to evaluate the impact of BMP implementation on water quality in the upper Midwestern region.

Tracking implementation of agricultural management practices

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. 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 ac of wetlands had been restored in the Upper Midwestern region, while 74,200 ac 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/yr, and reductions in phosphorus loss were estimated at 438,000 lb/yr. 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%, while BMPs have only reduced total phosphorus export by about 2%. 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 USDA National Agricultural Statistics Service (NASS) since 1840. 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 USEPA has tracked the status of water quality in lakes and rivers since 1992 at two year intervals through the National Water Quality Inventory (305 b) 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. By 2000, EPA 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.

EPA has historically used water quality monitoring data to identify impaired water bodies. A variety of 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 is of variable quality. Much of the sediment and phosphorus loads transported to rivers and lakes in the Upper Midwest are delivered during high intensity rainfall events that represent a small proportion of the time available for collection of water quality samples. If these events are not sampled, 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 (NANI) 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 impacts on water quality arising from implementation of BMPs. BMPs are typically implemented at very low density at this scale, making it difficult to identify impacts of BMPs on water quality. Also, 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 of 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.

Evaluation of management practices on small research plots

When new approaches are first developed to reduce non-point 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:

- Will the BMPs from this experiment be sufficient to reduce the area of the hypoxic zone 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?

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 combines equipped with yield monitoring systems and global positioning systems (GPS) are typically used to collect yield information in assessing the effect BMPs. Advanced statistical techniques are needed to evaluate the effectiveness of BMPs established 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. Also, it is often difficult to measure water quality impacts of BMPs at the farm scale due to difficulties in measuring runoff, erosion and drainage over large areas.

Regional Scale Evaluations of management practices

The USDA-ARS and USDA-CSREES provided funding for Management Systems Evaluation Areas (MSEA) and Agricultural Systems for Environmental Quality (ASEQ) during the 1990s. 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 main 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 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.

Effectiveness of management practices at watershed scale

The National Research Council reports that one of the primary needs of the TMDL program is information on the effectiveness of Best Management Practices (BMPs) and the related processes of system recovery (USEPA, 2002). A 1998 report by the United States Environmental Protection Agency (USEPA) stated that BMP effectiveness research ranks second among the EPA'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 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 is to evaluate the effectiveness of BMP implementation in select 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 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 for 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 the latter is a study conducted by Richards and Baker (2002) for four watersheds in Ohio. They studied log transformed water quality data from 1975-1995 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 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 river 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, 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.

Example 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 which occasionally caused the death of cattle and dogs who 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 each. Monitoring was conducted over a period of approximately 2.5 yr 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 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 ($\text{NO}_3\text{-N}$), ammonia nitrogen ($\text{NH}_3\text{-N}$), 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 (1998) 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 mouth of the watershed. Total phosphorus levels in the lake increased from 29 ppb before implementation of pollution control measures to 44 ppb fifteen 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 of 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 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 km² and 27.2 km²) in southern Wisconsin using monitoring data collected over a period extending from 1984 to 1998. The post-BMP monitoring data was collected eight years after BMP implementation began. BMPs included animal waste management, streambank protection, and upland erosion and nutrient management strategies. Significant reductions in NH₃-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 NH₃-N storm loads were observed based on regression residuals.

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. 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 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) 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. 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 conservation measures. During the

same period, buffer strip treatments only reduced water runoff by about 9%.

Dinnes et al. (2005) worked with eight producers in a 400 ha sub-watershed of the Walnut Creek watershed in Iowa to reduce nitrogen fertilizer applications through use of a late spring nitrogen test (LSNT). Water quality data collected 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 a 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 which it was tested (Walker, 1994; Sharpley et al., 2002).

Types of Questions Models Can Answer

Substantial advances have been made during the past decade 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 evaluate the impacts of BMPs on TMDL goals (Dalzell et al., 2004).

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 growing 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 in order to attain pre-determined 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), the hydrologic properties of soil horizons across the landscape (hydraulic conductivity, moisture characteristics), and the variability in agricultural management practices for different fields.

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 cannot answer every question. They are particularly limited when the accuracy or availability of input data are questionable. The accuracy of models that have been calibrated and validated can be quantified, but the accuracy of uncalibrated models cannot. Models should not be used if they do not accurately represent the processes and pathways for transport of pollutants within a field or watershed. 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 may be inappropriate.

There are a variety of approaches used to represent the effects of BMPs on water quality in models. Some of these approaches are deterministic, others are statistical, 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 detailed descriptions below). Each of these 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>.

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, which has the ability to simulate in-stream processes. The main weakness of HSPF in agricultural settings is that it does not explicitly account for rates of fertilizers, different types of mechanical tillage operations, or tile drainage management systems.

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). It has sophisticated 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 watershed boundaries or soil map unit boundaries. This is particularly problematic in small watersheds.

GLEAMS (Groundwater Loading Effects of Agricultural Management Systems) is a field scale model operating at a daily time step designed to estimate pollutant losses beyond 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 in depth.

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 (DeCoursey et al., 1992). 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 (<http://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, selecting an appropriate model, verifying the model code, conducting field experiments to calibrate the model, assessing the validity of the model, 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 quantitative models may require several hundred input parameters (although not all parameters are completely independent), whereas empirical qualitative models may only require several tens of 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. 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 river travel times.

In addition to the processes and pathways, it is essential to have 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 for the sales of fertilizer, number and species of farm animals, average rates of manure production and manure 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 landuse, dates of crop planting and harvest, and residue cover. At the scale of watersheds there is often considerable uncertainty in input data about these farm management practices.

At watershed scales, information concerning spatial and temporal variations in precipitation become important. During a particular storm, one area of the watershed may experience much more intense precipitation than the rest of the watershed. For this reason, having an accurate 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 scale of watersheds that are not

important at the scale of plots or hillslopes, 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 data set different, 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 from 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, the results of model application are highly dependent on the model and assumptions made during the model application process. Modeling standards/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, do not occur uniformly spatially or temporally 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). 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 non-point source pollution. The optimization of BMP type and placement (location) using models has rarely been done due to the computational requirements and complexity. This is an area of significant opportunity.

Pionke et al. (2000) found that 98% of the algal available phosphorus measured in a 25.7 ha agricultural watershed came from 6% of the watershed area. Edwards and Owens (1991) determined that 66% of the sediment load observed over a 28-year period in small upland agricultural watersheds was attributed to five of the biggest erosion-producing storms. In a study of two agricultural watersheds in Pennsylvania, Gburek and Sharpley (1998) determined that 70% of the flow-weighted dissolved phosphorus load was exported in the stormflow despite the fact that stormflow only accounted for 10% of the long-term flow distribution.

The most cost effective reduction of nonpoint source pollution loads at the watershed outlet in agricultural settings is dependent on the implementation of effective best management practices (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; Florinsky, 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.

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. 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, it has been proposed that 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 effectiveness of management practices

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. These types of interactions involve complexity.

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 left behind 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, 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.

Other Issues

Environmental management involves reducing the impact of multiple pollutants on the soil, water, air in both terrestrial or 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 university 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 also declining. EPA doesn't generally fund research, though they fund projects that implement BMPs to improve water quality. In the interest of improving the efficiency with which BMPs are implemented to improve water quality, perhaps EPA 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 Best Management Practices (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 wetter climate, increases in the proportion of land that is tile drained, or reductions in the amount of pasture. 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.

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 scale of small watersheds. 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, or failure to implement BMPs that correct the most important sources of pollution, or 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 (one watershed improved with BMPs, the other watershed left as a control). Also, more focus is needed to evaluate the effectiveness of BMPs targeted to portions of the landscape that contribute most to water quality degradation. The phrase “precision conservation” has recently been coined to refer to this targeted approach.

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 using a model that simulates all major transport pathways that occur at that scale, and having high quality input data to parameterize the model, and on using long-term 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 estimate the portions of the field or watershed that are most critical for control of non-point 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 soil or landscape characteristics. Caution must be used to avoid applying models under conditions that may be inappropriate, including catastrophic storm events, spatial scales for which the model was not intended, or watersheds in which inadequate or inaccurate input data are available to calibrate and validate the model.

References

- Addiscott T. M., J. Smith, and N. Bradbury. 1995. Critical evaluation of models and their parameters. *J. Environ. Qual.* 24:803-807.
- Anderson, M.P. and W.W. Woessner. 1992. *Applied groundwater modeling; simulation of flow and advective transport*: Academic Press, New York.
- 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. Minnesota Pollution Control Agency. St. Paul, MN. <http://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*. U.S. Environmental Protection Agency, National Exposure Research Laboratory, Athens, Ga. EPA/600/R-97/080. 755 p.
- 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. <http://www.usawaterquality.org/integrated/MNRRiver.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.
- Board on Water and Soil Resources. 2005. eLINK. <http://www.bwsr.state.mn.us/outreach/eLINK/index.html>
- K.S. Bracmort, B.A. Engel, and J.R. Frankenberger. 2004. Evaluation of structural best management practices 20 years after installation: Black Creek Watershed, Indiana. *Journal of Soil and Water Conservation* 59:191-196.
- 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., 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.
- Clausen, J. C. and J. Spooner. 1993. Paired watershed study design. U.S. Environ. Protection Agency

- 841-F-93-009. Office of Water, Washington, D.C.
- Dalzell, B. J., P. H. Gowda, and D. J. Mulla. 2004. Modeling sediment and phosphorus losses in an agricultural watershed to meet TMDLs. *J. Am. 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.
- DeCoursey, D., L. Ahuja, J. Hanson, M. Shaffer, R. Nash, C. Hebson and K. Rojas. 1992. Root Zone Water Quality Model: version 1.0: technical documentation. USDA-ARS, GPSR Technical Report No. 2., Great Plains Systems Research Unit, Ft. Collins, Colorado.
- Dinnes, D., D. Jaynes, D. Karlen, D. Meek, C. Cambardella, T. Colvin and J. Hatfield. 2005. The Walnut Creek nitrogen initiative. <http://www.nstl.gov/research/onepage/wcn220.html>.
- 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.
- Edwards, D. R. and L. B. Owens. 1991. Large storm effects on total erosion. *J. Soil Water Conserv.* 56:75-78.
- Florinsky, I. V. 1998. Accuracy of local topographic variables derived from digital elevation models. *Int. J. Geog. Info. Sci.* 12:47-61.
- Gallant, J.C. and J.P. Wilson. 2000. Primary topographic attributes. p. 51 - 85. In J.P. Wilson and J.C. Gallant (eds.) *Terrain analysis: Principles and applications*. John Wiley and Sons, Inc., New York.
- Gallichand, J., E. Aubin, P. Baril and G. Debailleul. 1998. Water quality improvement at the watershed scale in an animal production area. *Can. 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. Technol.* 28:441-449.
- Gburek, W. J. and A. N. Sharpley. 1998. Hydrologic controls on P loss from upland agricultural watersheds. *J. Environ. Qual.* 27:267-277.
- 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.
- 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.
- Graczyk, D. J., J. F. Walter, J. A. Horwath and R. T. Bannerman. 2003. Effects of best management practices in the Black Earth Creek priority watershed, Wisconsin, 1984-98. *Water Res. Investigations Rept.* 03-4163. U.S. Geological Survey, Washington, D.C.
- 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.
- 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. *Texas Inst. Appl. Environ. Res.*, Stephenville, TX.
- 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. *Transactions of 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.
- 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 Conservation* 59:96-103. <ftp://ftp-fc.sc.egov.usda.gov/NHQ/nri/web/swcs-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. Env. Qual.* 31:1610-1622.
- Moore, I. D. and J. L. Nieber. 1989. Landscape assessment of soil erosion and nonpoint source pollution. *J. Minn. 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, G.A. Peterson. 1993. Soil attribute prediction using terrain analysis. *Soil Sci. Soc. Am. 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.
- Mulvaney, R.L., S.A. Khan, R.G. Hoefl, and H.M. Brown. 2001. A soil organic nitrogen fraction that reduces the need for nitrogen fertilization. *Soil Sci. Soc. Am. 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.
- 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.
- 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.
- 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.
- Shuler, D. J. 1996. Lake Shaokatan restoration project: Final report. Yellow Medicine River Watershed District. Minnesota Pollution Control Agency, St. Paul, MN.
- 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. *Transactions of ASAE* 25(3):666-674.
- Spooner, J., D. E. Line, S. W. Coffey, D. L. Osmond, and J. A. Gale. 1995. Linking water quality trends with land treatment trends. <http://www.bae.ncsu.edu/programs/extension/wqg/brochures/ten.html>
- Smith, R.A., G.E. Schwarz, and R.B. Alexander. 1997. Regional interpretation of water-quality monitoring data. *Water Resources Research* 33:2781-2798.
- Theobald, D. M. 1989. Accuracy and bias issues in surface representation. p. 99-106. In: M. Goodchild and S. Gopal (eds.), *Accuracy of Spatial Databases*. Taylor and Francis, Bristol, PA.
- Tomer, M.D., D. E. James and T. M. Isenhardt. 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. 2005. Conservation Effects Assessment Project. <http://www.nrcs.usda.gov/technical/nri/ceap/>
- USEPA. 2002. The twenty needs report: How research can improve the TMDL program. EPA 841-B-02-002. Office of Water, Washington, D.C.
- USEPA. 1980. An approach to water resources evaluation on non-point silvicultural sources. EPA – 600/8-80-012. USEPA Environmental Res. Lab., Athens, GA.
- 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.
- 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.

DRAFT

DRAFT

Where do we go from here?

Dean W. Lemke, Iowa Department of Agriculture and Land Stewardship

Dennis P. McKenna, Illinois Department of Agriculture

The Upper Mississippi River Sub-Basin Hypoxia Nutrient Committee 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: Iowa State University, U.S. EPA Office of Wetlands, Oceans and Watersheds, EPA Regions 5 and 7 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. In the Action Plan the Task Force adopted three Long-Term Goals:

- Coastal Goal: By the year 2015, subject to the availability of additional resources, reduce the 5-year running average areal extent of the Gulf of Mexico hypoxic zone to less than 5,000 square kilometers through implementation of specific, practical, and cost-effective voluntary actions by all States, Tribes, and all categories of sources and removals within the Mississippi/Atchafalaya River Basin to reduce the annual discharge of nitrogen into the Gulf.
- Within Basin Goal: To restore and protect the waters of the 31 States and Tribal lands within the Mississippi/Atchafalaya River Basin through implementation of nutrient and sediment reduction actions to protect public health and aquatic life as well as reduce negative impacts of water pollution on the Gulf of Mexico.
- Quality of Life Goal: To improve the communities and economic conditions across the Mississippi/Atchafalaya River Basin, in particular the agriculture, fisheries, and recreation sectors, through improved public and private land management and a cooperative, incentive based approach.

The Task Force also agreed on five principles:

1. Encourage actions that are voluntary, practical, and cost-effective;
2. Utilize existing programs, including existing State and Federal regulatory mechanisms;

3. Follow adaptive management;
4. Identify additional funding needs and sources during the annual Agency budget process; and,
5. Provide measurable outcomes as outlined below in the three goals and strategies.

Implementation of the Action Plan for Reducing, Controlling, and Mitigating Hypoxia in the Northern Gulf of Mexico will require a significant level of commitment from the federal agencies and state governments and increased awareness and action by hundreds of thousands of stakeholders. A key to achieving these commitments and actions is coordination and outreach by the sub-basin committees established by the states in response to Action Item #2 of the Action Plan.

2 States and Tribes in the Basin, in consultation with the Task Force, will establish sub-basin committees to coordinate implementation of the Action Plan by major sub-basins, including coordination among smaller watersheds, Tribes, and States in each of those sub-basins;

Clearly, the Task Force recognized that no single approach to nutrient reduction would be effective throughout the portions of 31 states that lie within the Mississippi River Basin. Rather, because the soils, hydrology, land use and cropping practices vary considerably across the Mississippi River Basin, it was left to the Sub-Basin Committees to develop the appropriate strategies for their portion of the Basin.

The Action Plan proposed ten short-term actions to achieve the long-term coastal, basin and quality of life goals. Six of those short-term actions items were assigned to the states through the sub-basin committees:

#5 States, Tribes, and Federal agencies within the Mississippi and Atchafalaya River Basin will expand the existing monitoring efforts within the Basin to provide both a coarse resolution assessment of the nutrient contribution of various sub-basins and a high resolution modeling technique in these smaller watersheds to identify additional management actions to help mitigate nitrogen losses to the Gulf, and nutrient loadings to local waters, based on the interim guidance established by the National Water Quality Monitoring Council;

#6 States, Tribes, and Federal agencies within the Mississippi and Atchafalaya River Basin, using available data and tools, local partnerships, and coordination through sub-basin committees, described in action #2, will develop strategies for nutrient reduction. These strategies will include setting reduction targets for nitrogen losses to surface waters, establishing a baseline of existing efforts for nutrient management, identifying opportunities to restore floodplain wetlands (including restoration of river inflows) along and adjacent to the Mississippi River, detailing needs for additional assistance to meet their goals, and promoting additional funding;

#8 Clean Water Act permitting authorities within the Mississippi and Atchafalaya River Basin will identify point source dischargers with significant discharges of nutrients and undertake steps to reduce those loadings, consistent with action #6;

#9 States and Tribes within the Mississippi and Atchafalaya River Basin, with support from Federal agencies, will increase assistance to landowners for voluntary actions to restore, enhance, or create wetlands and vegetative or forested buffers along rivers and streams within priority watersheds consistent with action #6;

#10 States and Tribes within the Mississippi and Atchafalaya River Basin, with support from Federal agencies, will increase assistance to agricultural producers, other landowners, and businesses for the voluntary implementation of best management practices (BMPs), which are effective in addressing loss of nitrogen to waterbodies, consistent with action #6;

The states, with information-sharing and coordination through the sub-basin committees, are the only entities capable of fulfilling the responsibilities outlined in the Action Plan. Several of the states have already initiated programs to reduce nutrient losses from both point and nonpoint sources. The states also have a great potential to leverage existing state and federal funds to provide assistance to agricultural producers and others for implementation of management practices to reduce nutrient losses. Many states in the basin have cost-share and incentive programs through their Departments of Agriculture and Natural Resources. State water quality agencies also have discretion in targeting Section 319 funds to better address nutrients.

The sub-basin committees through their state members are uniquely qualified to identify the key stakeholders who can influence opinion and support needed changes in practices and programs. Each of the state agencies and their counterpart water quality or agricultural department has established relationships with their constituents, whether agricultural producers or regulated entities such as wastewater facilities. These relationships are based on understanding and an often hard-earned trust. Because the legal, legislative and administrative framework varies among the states, it is critical that the existing state relationships with key stakeholders be maintained and supported in developing and implementing strategies to reduce nutrient loads to the Gulf of Mexico and to water bodies within the Basin.

Contingent upon adequate funding, the Upper Mississippi River Sub-Basin Hypoxia Nutrient Committee will continue to conduct and expand its activities in support of the Action Plan for Reducing, Controlling, and Mitigating Hypoxia in the Northern Gulf of Mexico. Specific tasks include:

- publication and distribution of the proceedings of the Gulf Hypoxia and Local Water Quality Concerns workshop;
- continued technical networking with research and Extension faculty at the five land grant universities;
- meetings of the Tier 2 stakeholders group which includes representatives of the land-grant universities, agricultural and environmental organizations, public water supplies and publicly owned treatment works, state water quality and agricultural agencies and federal agencies;
- compilation of spatial information on nutrient sources and sinks within the Upper Mississippi River sub-basin for use in developing a nutrient reduction strategy for the sub-basin;
- preparation and distribution of materials on nutrient issues and potential solutions identified in the September 2005 workshop for distribution to existing watershed groups, agencies and environmental and agricultural organizations;
- planning of a second conference to address technical, social and policy issues related to implementation of a nutrient-reduction strategy for the Upper Mississippi River Basin. Additional funding will be needed to support this conference.

The activities of the Upper Mississippi River Sub-Basin Hypoxia Nutrient Committee (UMRSHNC) 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.

The formation of the UMRSHNC by the five states has addressed the Action Item #2 of the Action Plan. During the past year, the UMRSHNC has established a strong organizational framework and mission/vision statement, and has established the foundation for future accomplishments through forming a stakeholders group that includes representatives of key agricultural and environmental organizations, research universities, and municipal, state and federal agencies. An important component of the Committee's approach to reducing nutrient losses is outreach. We have identified three target audiences: the stakeholder group; local watershed planning groups; and local technical service providers including Extension, NRCS and soil and water conservation district staff and certified crop advisers. By working through the existing network, we can most effectively and efficiently influence the message that is delivered to producers about nutrient-caused impairments of local streams and lakes and of the Gulf of Mexico. The results of the September 2005 conference on management practice effectiveness will be re-packaged into short fact sheets and brochures to serve as a basis for educating these key leaders and technical advisers on nutrient issues and solutions.

Collaboration/Partnerships: To carry out the responsibilities of the Action Plan for the Upper Mississippi subbasin, a two-part strategy has been developed by the UMRSHNC. One is to facilitate and encourage technical networking between research/extension personnel of the five land grant universities, other research institutions, and technical agencies. The goal of the technical networking is to strengthen the knowledge base and technical exchange between the five states on practices and programs necessary to feasibly reduce nutrient transport to streams and rivers in a manner which is economically viable.

The second part of the strategy is to establish a broad-based input forum which can facilitate two-way communications between and within the primary stakeholder interests of the subbasin. This goal of this forum, the Stakeholder Advisory Group, is to obtain input, network between, and formulate recommendations from the varied interests of the stakeholders of the five states. The primary role of the group is to focus on issues relating to Gulf hypoxia, but secondarily the group will consider local water quality concerns as they relate to Gulf hypoxia. Specific details of the roles and function of the Stakeholder Advisory Group are:

- o Serve as the primary stakeholder input group for the Upper Mississippi River states concerning issues relating to the Gulf of Mexico hypoxic zone
- o Obtain inputs from broad stakeholder interests on issues relating to hypoxia and the actions of the Upper Mississippi River states to reduce nutrient contributions to the Gulf
- o Communicate issues of concern and recommendations back to the various stakeholder interests and organizations within the Upper Mississippi River states
- o Develop recommended positions on issues of concern for consideration by the UMRSHNC governing body and the Task Force
- o Represent stakeholder/agency's constituent area and interest in discussions and

development of recommended positions

- o Provide leadership to communicate issues of concern and recommended positions back to constituent area or agency

Stakeholder Advisory Group Members

Five State Agencies – Agriculture, Conservation or Environmental Protection

- o Illinois Environmental Protection Agency
- o Iowa Department of Natural Resources
- o Minnesota Department of Natural Resources
- o Missouri Department of Agriculture
- o Wisconsin Department of Agriculture, Trade and Consumer Protection

Five Land Grant Universities – Research/Extension

- o University of Illinois
- o Iowa State University
- o University of Minnesota
- o University of Missouri
- o University of Wisconsin

Five Agricultural Stakeholder Organizations

- o Illinois Fertilizer and Chemical Association
- o Iowa Farm Bureau Federation
- o Minnesota Soybean Association
- o Missouri Corn Growers Association
- o Professional Dairy Producers of Wisconsin

Five Environmental, Consumer and City Utility Organizations

- o Prairie Rivers Network
- o Metropolitan Water Reclamation District of Greater Chicago (Illinois)
- o Cedar Rapids (Iowa) Water Department
- o The Nature Conservancy

- o Audubon Upper Mississippi River Campaign

Five Federal Agencies

- o USDA Natural Resources Conservation Service
- o USDA Agricultural Research Service
- o U.S. Geological Survey
- o U.S. EPA Region V (ad hoc)
- o U.S. EPA Region VII (ad hoc)

Reassessment of the Hypoxia Action Plan

The following discussion is based upon materials provided by the U.S. Environmental Protection Agency to members of the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force.

The Action Plan for Reducing, Mitigating, and Controlling Hypoxia in the Northern Gulf of Mexico included an action item to assess the nutrient reductions achieved and the response of the hypoxic zone, water quality throughout the Basin and economic and social effects. The reassessment includes a series of actions that will develop the information necessary for the Task Force to review the 2001 Action Plan and make revisions as necessary. It will be completed over the next two years. These actions include reassessment of the primary causes of Gulf hypoxia and management approaches to address these causes. The reassessment will result in the following outcomes, in order to produce a revised Action Plan:

- A Peer Review of the August 2004, Region 4 White Paper on the role of nitrogen and phosphorus in causing Gulf Hypoxia
- A bibliography of all references pertinent to the science of Gulf hypoxia
- A summary of available information on implementation of federal and state management programs and activities
- An inventory of existing point sources throughout the Mississippi River Basin
- A public science symposium on the causes of Gulf hypoxia to feed into the Expert Science Panel report.
- An Expert Science Panel report that describes the best scientific understanding of the causes, sources, and controlling processes of the formation, extent and duration of hypoxia in the northern Gulf of Mexico
- At least two sub-basin workshops on management science
- A synthesis of management recommendations and options revisions to the Action Plan (which may include additional expert or peer review)

- Public participation and opportunities for comment and review

Peer Review of the August 2004, Region 4 White Paper on Causes of Hypoxia

EPA scientists in Region 4 (Atlanta) conducted a review of data and information regarding hypoxia in the northern Gulf of Mexico. This Region 4 staff assessment concluded that phosphorus, rather than nitrogen, may be the limiting nutrient controlling Gulf hypoxia. An unauthorized draft of the report was released in January of 2004. This report, because of its controversial conclusion regarding the role of phosphorus in Gulf hypoxia, caused a significant amount of concern among stakeholders. After considerable internal review by Region 4 EPA scientists, the Region released a significantly revised version of the draft report for a broader internal EPA review in April 2004. The report was revised based on the broader EPA review and released to the Hypoxia Task Force in August 2004, as an informational document with the specific purpose of encouraging discussion and posing questions for the reassessment of the Action Plan. The draft report, "Evaluation of the Role of Nitrogen and Phosphorus in Causing or Contributing to Hypoxia in the Northern Gulf, August, 2004," had not at that time been subjected to external peer review.

The August report, and the earlier drafts that were not prepared for release but were widely circulated outside EPA, raised concerns among Task Force members and stakeholders throughout the Mississippi Basin. In response, the Task Force requested an expedited scientific peer review of the August 2004 draft report to be coordinated by the Monitoring, Modeling and Research Workgroup (MMR) of the Task Force. Concerns were raised at the September 1, 2004 public meeting of the Task Force that earlier versions of the report (January and April drafts) should also be included in the review. Thus, the Task Force decided to include the two draft versions of the report, along with the August 2004 Report, in the peer review request.

The External Peer Review of the Draft Region 4 report, "Evaluation of the Role of Nitrogen and Phosphorus in Causing or Contributing to Hypoxia in the Northern Gulf, August, 2004," provided considerable insight regarding the various processes involved in hypoxia formation in the northern Gulf of Mexico. The peer review raised substantial questions and issues regarding the role and significance of these processes in producing hypoxia. EPA believes the discussion of these processes has produced a valuable dialogue regarding the issues needing consideration in future reassessment.

EPA concluded that further refinement, completion, and formal publication of the Draft Region 4 report would not contribute significantly to the resolution of identified issues or to the reassessment of the Action Plan and that the reassessment process provides a greater opportunity for continuing dialogue on the issues identified through the paper and the peer review. The results of the peer review are available at http://www.epa.gov/msbasin/taskforce/peer_review.htm.

Updates of Federal Data, Bibliography, Federal and State Programs, and Source Inventory

Two sub-committees of the Coordinating Committee will provide information for the reassessment as follows:

The Science Reassessment Team (SRT) -- will develop a bibliography of all references pertinent to the science of Gulf hypoxia. The bibliography will include references cited in the initial CENR science assessment conducted from 1998-2001. It will be supplemented

by a request to all Task Force participating agencies for references that may have been omitted from the initial science assessment and any new research published since 1998. The Monitoring, Modeling, and Research Workgroup (led by USGS and NOAA), which developed the MMR Strategy for the Task Force, will coordinate this team, with appropriate leadership and participation from states and other federal agencies.

The Management Action Reassessment Team (MART) – will develop two reports, collating information on point sources and available programs. The first report will present an inventory of point source location and loadings throughout the Mississippi River Basin. EPA is compiling this information from available federal data. The second report will update information on implementation of federal and state management activities to address the cause of hypoxia. The MART will compile information on management activities from all Task Force members, particularly USDA, ACOE and states. EPA and USDA Coordinating Committee members will coordinate this team with appropriate leadership and participation from states and other federal agencies.

Symposium on the Causes of Gulf Hypoxia

To support the work of the Expert Science Panel, the Task Force will convene a symposium on the causes of Gulf hypoxia, including examination of the factors that contribute to the timing, duration and extent of the hypoxic zone. This public symposium will serve as a kick-off of the reassessment process for all the interested stakeholders and provide an opportunity to present the most recent research on causes of Gulf hypoxia. This workshop, which is tentatively scheduled for April 2006, will be coordinated with the Expert Science Panel and serve to provide current information for their analysis.

Expert Science Panel Report on the Cause of Gulf Hypoxia

The purpose of this action is to develop an updated, independent assessment of the causes of Gulf hypoxia and recommend whether the most recent body of scientific evidence supports or suggests revisions to the assessment that formed the basis of the 2001 Action Plan. After review of several options, the Task Force agreed that an expert panel be chartered through EPA's Science Advisory Board (SAB) to review available scientific information and provide a report that synthesizes the current state of knowledge of the causes of Gulf hypoxia. The expert panel will develop a report that answers the following general question:

Using the Integrated Science Assessment (CENR, 2000), the six supporting reports, and other scientific contributions, what is the current scientific understanding of the causes, controlling processes and sources of formation, extent and duration of hypoxia in the northern Gulf of Mexico?

Description of SAB Process The Task Force selected the SAB because of the opportunities for public access, timeliness and cost. All SAB panel meetings and deliberations are open to the public in accordance with Federal Advisory Committee Act and EPA administrative policies for advisory committee management. The SAB will hold at least two meetings, one for initial discussion of their charge and the second for deliberations on their findings, both of which will be open to the public. As described in the SAB charter, all advice and recommendations developed by the SAB are provided directly to the EPA Administrator.

As part of the process to develop the Gulf hypoxia report, the SAB will select a chair and seek public input on panel members. The Task Force, as well as other members of the public, will be able to make recommendations for panel members. The SAB will develop a short list of potential names that will be announced for additional public review and from that list, a panel will be selected. How the panel would proceed is ultimately dependent on the chair and final make up of the Committee.

Under the SAB's charter, the Assistant Administrator of the official sponsoring Program Office (Ben Grumbles for the Office of Water), will act as the Task Force representative to the SAB. Charge negotiations and any communication, including the submission of questions pertaining to draft reports, will have to be communicated by Ben Grumbles to the DFO and the Chair of the panel. However, the Task Force, as well as any other interested parties at any time may submit additional information for the panel to consider. Upon completion, the draft report will be released for public comment.

Sub-Basin Workshops on Management Science and Policy

The Task Force will support and participate in a minimum of two workshops hosted by the major sub-basin committees in the Upper Mississippi and the Lower Mississippi. These workshops will provide opportunities for researchers and program managers to discuss management science and activities relevant to each of the sub-basins and recommend actions to reduce the causes of Gulf hypoxia within the context of the broader goals of the 2001 Action Plan. Each workshop will produce a report on the major discussions and recommendations from the workshop. While additional workshops may be needed as the reassessment proceeds, at a minimum, two workshops are being held:

Upper Mississippi Sub-Basin Workshop This workshop is gathering together available knowledge to assess tools and possible solutions to agricultural nutrient losses to water resources. The workshop will inform implementation agencies and policymakers on the science of nutrient loss reduction, and thereby lay the groundwork for future planning, policy and programs. The Upper Mississippi Sub-Basin committee is considering holding a second conference after completion of the Expert Science Panel Report on the Cause of Gulf Hypoxia, to more specifically address upper basin implementation programs needed to address the goals of the Action Plan.

Lower Mississippi Sub-Basin Workshop This workshop will be hosted by the Lower Mississippi Sub-basin Committee to address management issues in the lower Mississippi and Atchafalaya Basins from the Mississippi and Ohio Rivers confluence to the Gulf of Mexico. The focus will be on agricultural and point sources as well as hydrologic modifications and freshwater wetlands.

Management Recommendations Synthesis and Revisions to the Action Plan

The Coordinating Committee will evaluate and synthesize the recommendations for management from the SAB review, the workshops and the public comments into a set of recommendations and options for the entire basin. This synthesis will form the basis of the recommendation to the Task Force.

Public Review and Comment

The public will provide comment and input to the formation of the Expert Science Panel as well as to their report development. The public will participate in the sub-basin workshops. The draft Action Plan revision will be made available for public review and comment at the conclusion of the activities described above.

Implementation

We hope that the results of this workshop have made a significant contribution in describing the effectiveness 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-crop 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. Off-site practices, and possibly some cropping system changes, will likely also be needed to reach water quality goals. The potential and limitations of improving both in-field and off-site management practices/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 and an agricultural economy buffeted by high input costs and low commodity prices, 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 may 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 runoff or erosion rates. In addition, different management practices are often necessary to reduce nitrate and phosphorus movement to surface water: nitrate BMPs modify infiltration, leaching and soil water content; phosphorus

BMPs modify surface runoff and erosion. 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. 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.

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.

We must get the right practice in the right area at the right rate to make a difference. Government programs based primarily, or sometimes solely, on a first-come first-served approach or a dominant goal of spending the allocated funds are relatively easy to implement, but will not get the job done.

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.

References

Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. 2001. Action Plan for Reducing, Mitigating, and Controlling Hypoxia in the Northern Gulf of Mexico. Washington, DC.