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 N_2 fixation), incomplete representations of field and watershed processes and limited data for validation can make projections uncertain. Because of limited long-term data sets, modeling is often used to project responses to management, but there are many difficulties in this approach.

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 (NO_3-N), ammonia nitrogen (NH_3-N), total Kjeldahl nitrogen (TKN), and chemical oxygen demand (COD) based on trend analysis. Major BMPs implemented included nutrient management, pasture and hayland management, waste utilization, dead poultry composting, and waste storage structure construction (Edwards et al., 1997).

Garrison and Asplund (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 km^2 and 27.2 km^2) 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 NH_3-N load during storm flows were observed in the larger watershed based on regression residuals. For the smaller watershed, significant decreases in both total phosphorus and NH_3-N storm loads were observed based on 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 collect from this and an adjacent watershed since 1997 showed a 41% reduction in nitrate-N losses from the watershed where the LSNT approach was used relative to losses in the control watershed. Corn yields in the two watersheds were similar in three out of four years.

Birr and Mulla (2005) implemented conservation tillage on 70% of the moldboard plowed acreage for three years in 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; Florinksy, 1998). The interpretation of terrain indices must account for each of these factors such that the application of the data is appropriate given the limitations each of these factors presents for the data.

Some of the most useful terrain indices for targeting BMPs to critical areas include slope steepness, compound terrain index (Moore et al., 1991; Gallant and Wilson, 2000), and stream power index (Moore et al., 1993). To be effective, each of these must be considered in relation to the potential for transport to a nearby stream or lake. This potential is largely based on the direction of flow paths across the landscape and on the proximity of a given area to a stream or lake. To date there have been few attempts to use quantitative techniques in terrain analysis in conjunction with simulation models in order to estimate the impacts on water quality of BMPs that are targeted to critical areas on the landscape.

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 could fund more research on non-point source pollution than in the past, most of their non-point source funding is for projects to assess watershed pollution sources. In the interest of improving the efficiency with which BMPs are implemented to improve water quality, 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/MNRiver.html.

Bishop P. L., W. D. Hively, J. R. Stedinger, M. R. Rafferty, J. L. Lojpersberger and J. A. Bloomfield. 2005. Multivariate analysis of paired watershed data to evaluate agricultural best management practice effects on stream water phosphorus. J. Environ. Qual. 34:1087-1101.

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 ResearchUnit, 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.

Florinksy, 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. Horwatich 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. Hoeft, and H.M. Brown. 2001. A soil orgnaic 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. Gopol (eds.), Accuracy of Spatial Databases. Taylor and Francis, Bristol, PA.

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

Udawatta, R.P., J.J. Krstansky, G.S. Henderson, and H.E. Garrett. 2002. Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. J. Environ. Qual. 31:1214–1225.

USDA-NRCS. 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.