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Theme Overview: Agriculture and Water Quality in the Cornbelt: Overview of Issues and Approaches

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JEL Classification Code: Q25

More than three decades have elapsed since the passage of the Federal Water Pollution Control Act with its stated goal of zero discharge of pollutants into the nation's waterways. Yet, water quality remains poor in many locations and considerable loading of pollutants continues. This is particularly true for agricultural sources of water pollution and is typified by the Upper Mississippi River Basin, where more than 1,200 water bodies appear on the current U.S. Environmental Protection Agency (EPA) listing of impaired waterways. Additionally, nitrate export from this region has been implicated as a significant cause of the hypoxic zone in the Gulf of Mexico, which covered nearly 20,000 km^2 in 1999 and more than 17,000 km^2 in 2006 (http://www.epa.gov/gmpo/nutrient/hypoxia_pressrelease.html). Although a substantial body of evidence on the effectiveness of agricultural conservation practices on water quality continues to be developed, the net effect of these programs and practices at the watershed scale is unclear. Increasingly, studies are being focused on the watershed (or landscape) scale and complex interactions between agricultural practices and inputs, the types and configuration of conservation practices on the landscape, and the resulting downstream water quality. While low cost methods to reduce agricultural non-point source pollution exist, large changes in water quality in agricultural regions are likely to be costly and met with resistance. This is because to achieve large changes in water quality, major alterations to land use or installation of expensive structural practices may be required, and the costs are borne directly by producers and landowners, or by the taxpayer.

Given the potentially large cost for significant improvements in water quality, it is critical to develop tools that can support cost-effective design of conservation policy and/or voluntary implementation of watershed plans focused on water quality. The following set of themed papers related to water quality and agriculture discuss these issues, with a specific focus on using integrated water quality and economic models to support better public policy and watershed-based solutions to these problems. The article following this one describes detailed field-scale data collected as part of a Conservation Effects Assessment Project supported by CSREES and ARS. In addition to assessing the effects of current conservation activities on water quality in these watersheds, data are used to calibrate a water quality model and are being integrated with economic cost information to study the optimal placement of additional conservation activities in the watershed. That article discusses the historical evolution of conservation activities in the three watersheds, the current water quality challenges in the watersheds, and the role that the integrated models can play in solving the problems.

In the third paper of the series, Secchi et al. employ a more aggregate unit of analysis (scale) for calibrating a watershed model and a biophysical carbon sequestration model and integrating them with economic data covering the entire state of Iowa. The focus of their analysis is on the potential unanticipated environmental effects of developing markets in ecosystem services that focus on a single service, such as carbon sequestration.

The final paper in the set addresses a different water quality issue: drinking water and nitrate levels. Specifically, the paper by Burkart and Jha considers whether it would be cost-effective for farmers to reduce nitrogen applications at the farm level, thereby reducing nitrate concentrations in the water supplies for residential consumers, rather than continue to treat the water in a denitrification plant prior to use.

In the remainder of this theme overview, we attempt to provide the casual reader with adequate background information on agricultural water quality problems, as well as the institutional framework within which these water quality problems in agriculture are currently managed. This includes a brief primer on the key pollutants, their sources, and the range of conservation methods that can attenuate their effects. It is also necessary to understand the fundamentals of the policy environment, which differs markedly from approaches taken in other industries. Specifically, voluntary actions are the focus of state and federal agency efforts under the requirements that they have to develop and implement Total Maximum Daily Loads (TMDLs). We briefly describe the TMDL process and note the range of federal and state conservation programs that provide funding for voluntary conservation efforts.

Agriculture and Water Quality Primer

Production of food and fiber have inevitable impacts on land and water resources. Conservation practices are intended to reduce those impacts, ideally with as little effect on the productive and ecosystem service capacities of the land. The critical questions for planning and implementation of effective conservation systems are then: What water quality pollutants are of primary concern and what types of conservation practices will provide benefits for various environmental impacts? Here, briefly, we provide generic answers to these questions that are most pertinent to agricultural watersheds in the Corn Belt generally, and Iowa and the Upper Mississippi Basin, specifically. Through this discussion, we emphasize key differences among specific pollutants, in terms of the hydrologic pathways from field to stream, and the types of conservation practices that can minimize their transport to receiving waters. The primary pollutants of concern in the Corn Belt include nitrate-nitrogen, phosphorous and sediment, and pathogens.

Nitrates-Nitrogen

Nitrate-nitrogen (NO_3-N) is a key pollutant of concern for its potential widespread impact on both public health and ecosystem function. Nitrate-nitrogen is readily leached through soils to groundwater and enters surface water systems directly by groundwater flow and through the subsurface drainage systems (tile drains), which were installed across large areas of poorly drained Midwestern soils beginning about 100 years ago. These drainage systems have allowed the Midwest to become the highly productive agricultural area that it is today, while short-circuiting the much slower, natural groundwater pathway to the stream. Concentrations of nitrate-nitrogen in drainage and stream water often exceed 10 mg NO_3-N /L, resulting in losses exceeding 20 kg N/ha in some years (Tomer et al., 2003). Regional nitrogen budgets for the Mississippi River Basin have implicated tile-drained regions of the Midwest as disproportionately contributing to N loads to the Gulf (Burkart and James, 1999). Nitrogen fertilizer is commonly applied to corn, at rates varying from 100 to 200 kg/ha. The efficiency of N uptake by the crop varies because of environmental conditions. Nitrogen losses are most prevalent in early Spring when crops are not present or are too small to effectively immobilize the available nitrate.

The problem of nitrate-nitrogen export is not solely caused by N fertilizer management or any other single factor, but rather it is a combination of soil management practices and physical, chemical, and biological characteristics of the soil, along with temperature and precipitation patterns (Dinnes et al., 2002). As a result, reducing nitrate loss is more than a matter of reducing N-fertilizer rates and improving timing of applications (Jaynes et al., 2004). Effective practices to control N losses include diversified crop rotations that increase use of forages and improved nitrogen management (including improved timing and rates of application, and use of nitrification inhibitors). Improved engineering of aging drainage infrastructure, and use of wetlands, cover crops, and denitrification walls or subsurface drainage bioreactors are other alternatives that have been shown effective. Because nitrate in extensively tiled areas is transported to streams primarily in subsurface drainage water, any filtering ability of riparian buffers and edge of field filter strips will be bypassed.

Phosphorous and Sediment

Surface runoff is the dominant mechanism that transports phosphorus, sediment, and pesticides and bacteria from agricultural fields, as opposed to the subsurface pathways of nitrate. Ecological impacts of P and sediment include eutrophication and sedimentation of receiving waters. Phosphorus losses from agricultural fields may be only a fraction of those observed for N (< 1 kg/ha.yr is commonly reported), but such losses can have major implications for the ecological integrity of lakes and streams. Phosphorus runoff from agricultural fields is largely controlled by soil P concentrations and crop residue cover (Sharpley et al., 2002). Residue cover encourages infiltration and discourages erosion. To improve phosphorus management at watershed scales, the use of "P indices" are being implemented that identify soil erodibility, soil P concentrations, residue management practices, and proximity to streams, to rank fields for runoff P losses. These indices can be used to target conservation practices to control P losses (Birr and Mulla, 2001) via reduced tillage, limited manure or fertilizer applications, terraces, vegetated filter strips, and/or riparian buffers. These practices are known to reduce erosion and phosphorus. Watershed responses to these conservation practices may be less than initially expected because streambank erosion, rather than agricultural fields, can contribute significant amounts of sediment and phosphorus to streams and rivers. These sources may result from past management activities.

Sediment and nutrient losses from agriculture, therefore, can result in a legacy of impacts within watersheds, necessitating a long-term commitment to their amelioration. For example, elevated nitrate concentrations in groundwater have been shown to remain for decades (Rodvang and Simpkins, 2001). Also, phosphorus accumulations in sediment may have a legacy, providing a long-term, internal loading source of mineral P to the water column (Christophoridis and Fytianos, 2006) and may ultimately affect groundwater P concentrations (Burkart et al., 2004).

Bacterial Pathogens and Livestock Concerns

Livestock is an important economic component of U.S. agriculture, accounting for over 60% of agricultural sales. Production estimates for 2005 include 72.6 million hogs, 10.9 million beef cows, 3.1 million milk cows, 150 million egg layers, and 131 million broilers for the 12-state North Central Region. In the Midwest, swine are increasingly produced in concentrated animal feeding operations (CAFOs) making manure management increasingly important, both as a source of nutrients for subsequent crops and as a potential environmental problem. CAFOs are also important in poultry and beef production. Potential water quality issues arising from manure application are nitrate leaching and loss in tile drainage networks, and loss of phosphorus and pathogens in overland runoff. Conservation practices seek to prevent accumulation of excess nutrients (nutrient management plans), reduce and/or treat runoff from feed lots, and mitigate runoff from manured fields (buffers, filter strips). Several studies suggest that increasing CAFO size offers certain economic advantages in production, but increases the amounts of manure applied to land near the CAFO, which increases the risk of loss of excess nutrients (Kellogg et al., 2000).

Bacterial pathogens that threaten water quality include Escherichia coli O157:H7, Salmonella, Enterococcus, Listeria, and Campylobacter. Pathogenic protozoa include Cryptosporidium and Giardia. Although these microorganisms cause disease in humans, they are commonly carried in livestock without visible symptoms. Because of the difficulty and cost involved in screening water samples for these pathogens, public health and water supply authorities have long relied upon indicator bacteria. In the past, fecal coliforms tests filled this function, but two indicators are now being promoted by U.S. EPA, Escherichia coli and Enterococcus. Quick and reliable tests for both of these microorganisms are now available and the presence of these bacteria has been correlated with the presence of disease-causing microorganisms. Measured E. coli densities in stream water can be evaluated against EPA's current standards, but the identification of the E. coli sources is more complex and important to developing effective watershed management strategies. Microbial source tracking is an emerging technology that allows the source animal to be determined. Potential sources in most watersheds include wildlife, farm animals, and humans.

Heterogeneity of Conservation Practices

There is a wide range of conservation practices used on agricultural land intended to provide water quality benefits, including engineered structures, edge-of-field practices, in-field nutrient and crop residue management practices, and land retirement. Government programs since the

1930s have promoted installation of conservation practices on agricultural lands. Much of the early focus of conservation practices was specifically on soil conservation, where the goal was to preserve the soil and to maintain its productivity.

Structural practices that have been used for controlling soil loss and the formation of gullies include terraces, grassed waterways, sediment basins, and grade stabilization structures. Terraces are used to decrease the length of the hill-slope to reduce rill erosion and the formation of gullies. Many early conservation practices were intended, in part, for water conveyance to improve trafficability, and thereby maximize agricultural production. In addition to structural practices, there are a variety of in-field management practices such as contour farming and strip cropping and tillage management, such as conservation tillage and no-till. Also, in some areas marginal lands that are highly susceptible to soil loss have been taken out of agricultural production and converted back to perennial vegetation.

Over the past thirty years, there has been an increased concern related to the overall water quality impacts of agriculture, including nutrient, pesticide, and pathogen loss from agricultural lands. Some conservation practices have been installed with an intended purpose of reducing the export of these contaminants. Two of these are buffer systems (riparian or grassed) and the reintroduction of wetlands back into the landscape. In addition, relative to nutrient losses, there has been an emphasis on appropriate nutrient management practices within agricultural fields to reduce the application of excess nutrients.

We have also learned that some agricultural practices have effects that were not intended. Subsurface drainage was used historically to enhance productivity of poorly drained lands, but these production benefits are offset by the environmental impacts of increased export of nitrate-nitrogen from these drainage networks. Surface inlets to subsurface drain systems also create a direct conduit for surface water to enter streams effectively bypassing riparian buffers or wetlands. Much of the agricultural landscape has been altered through stream straightening channelization. Stream straightening and subsurface drainage have significantly altered the hydrology of the landscape, which has led to significant streambank stability problems in many areas. So, while many of the conservation practices mentioned above may reduce soil loss from agricultural fields, if they do not significantly reduce water flow in the streams, the stream power is not reduced. As a result, rather than carrying sediment from fields, the streams may erode sediment from the streambed and streambanks.

While there is a wide range of practices that can be used on agricultural lands for providing water quality benefits, many times the locations within the watershed where practices are implemented have not been specifically targeted to achieve the greatest reduction of contaminants in downstream water bodies. This is likely the result of the voluntary enrollment in federal conservation programs combined with ineffective targeting technology. Recent advances in remote sensing and geographic information systems offer an opportunity for dramatic improvements in our ability to target conservation practice installation in large watersheds. With the limited amount of resources available for conservation practices, there will likely be increased importance on targeting implementation to those areas where there may be the greatest benefit from a water quality perspective. One program that has used targeting with some effectiveness is the Conservation Reserve Program (CRP), which targets land choices based on an environmental benefits index. While the effects of CRP on soil quality, carbon storage, and wildlife have been assessed, the aggregate effects at the watershed scale are less understood.

Finally, it is important to understand that water quality monitoring in the United States is done by a variety of state and federal agencies, including USGS and USEPA, and many municipal and commercial water supply entities, but the great majority of streams and rivers are not routinely monitored. Thus, in many cases, the actual level of pollutants is simply unknown.

The Policy Environment: TMDLs and Voluntary Implementation

Voluntary cost-share and incentive programs sponsored by USDA and States are large in geographic scale and fiscal commitment (over $4.5 billion was spent in 2005 by USDA-funded programs alone). These programs generally provide varying incentives to farmers for the installation of structural or management practices described above. The criteria for participant eligibility vary from program to program, and conservation compliance provisions require that landowners who farm on highly erodible land undertake some conservation activities in order to be eligible for other government incentives or subsidies. In addition to the largest program, the CRP, there is a cost-share program entitled the Environmental Quality Incentive Program, which provides cost share to producers willing to install various conservation structures or practices on their farms. Notably, the 2002 Farm Bill contained a new program the Conservation Security Program a watershed-based initiative intended to compensate farmers for adopting conservation practices. Like the CRP, which covers the full cost of retiring land from production, the program was intended to cover the full cost of adopting conservation practices (rather than less than 100% of the cost as traditional cost share programs do), but the focus of the Conservation Security Program is on land that stays in production. However, funding constraints have prevented the program as it was initially envisaged from being fully implemented.

Ironically, while there are large conservation programs funded and administered through USDA, the primary law that addresses nonpoint source agricultural pollution loadings is under the auspices of the U.S. Environmental Protection Agency (EPA) via the Clean Water Act. Rather than assign standards and require that sources implement changes in production or invest in abatement technology to meet those standards, as has been the norm for air and water quality problems stemming from point sources, the Total Maximum Daily Load (TMDL) approach was adopted. Under the TMDL framework, states are responsible for compiling lists of water bodies not meeting their designated uses, which are then reported as "impaired waters." The sources of impairment vary across locations. For example, Iowa has 213 water bodies on the list and pathogens (bacteria) are the leading cause of listing, accounting for about 20% of the impaired water bodies, with sediment/turbidity accounting for about 10%. Nationally, it has been estimated that 40% of rivers and estuaries fail to meet recreational water quality standards because of microbial pollution (Smith & Perdek, 2004).

Note that water bodies are viewed as impaired only if they do not meet their "designated use." Thus, two water bodies can be equally contaminated with only one being listed as impaired if their designated uses are different (e.g., boatable vs. swimmable). This is part of what makes the TMDL rules so difficult to interpret and why a simple indication of whether a water body is listed or not is not necessarily a good indication of its level of water quality.

Once a water body has been identified as not meeting its designated use, the state is required to identify the sources of the impairment and the "maximum allowable daily load" of pollutants that would eliminate the impairment. Finally, states are to suggest reductions for the various pollutant sources that would allow the watershed to reach the TMDL. Importantly, there is no regulatory authority by the states or EPA to require that these reductions occur. Thus, the institutional environment in which nonpoint source water quality reductions may occur is fundamentally voluntary.

In the TMDL process, modeling and monitoring can play important roles in allocating pollutant loads to various sources, such as helping to determine the relative contributions of row crops, CAFOs, and urban sources to loads of nutrients and bacteria observed in large watersheds. Two models, the Soil and Water Assessment Tool and the Hydrological Simulation Program-FORTRAN models are most often used to support TMDL assessments (Benham et al., 2006). These models combine GIS-based spatial data of watershed physical features with information on cropping systems, animal densities, fertilizer and pesticide use, and point sources. For non-point source pollutants, conservation practices are a key to developing mitigation strategies that allow watersheds to meet TMDL goals. Since TMDLs may be designed to mitigate multiple pollutants (e.g., nitrate and bacteria), combinations of conservation practices may be necessary to achieve the necessary improvements in water quality.

Final Remarks

The purpose of this overview is to introduce readers to the set of water quality problems associated with row-crop agriculture and livestock operations in the Corn Belt and Upper Mississippi River Basin. The problems are complex, with a great many individual decentralized decision makers contributing, both positively and negatively, to their solutions. Adding to these complex problems are the ever-changing demands on agriculture to supply food, feed, fiber, and fuel. These demands are leading to new questions and concerns related to agriculture and may allow for some solutions that are economically viable and environmentally beneficial. Some concerns are related to potential use of marginal lands for row crop agricultural production and increasing continuous corn acreage to supply the bioeconomy. At the same time, the bioeconomy, particularly if cellulose biofuels become feasible, may provide opportunities for more diversified cropping systems that have environmental benefits. Associated with some of these issues is the increasing importance of agribusiness through decisions such as siting of CAFOs and ethanol plants. Siting decisions should consider the potential environmental impacts of these facilities both from a water quality and water quantity perspective.

In the remaining three papers of this water quality theme, the authors describe how data and models can be used to characterize the problems, model the underlying biophysical and economic processes, and ultimately (hopefully) contribute to solutions. Given the policy environment described above one of voluntary-based action and a myriad of conservation programs with diverse goals and ever-present funding constraints we believe that models of water quality processes carefully integrated with economic models are essential, both to assess existing programs, and more importantly, to design and implement cost-effective approaches to meeting society's water quality goals. These modeling efforts will be difficult and will appropriately come under a great deal of scrutiny.

The complexity of the ecology and the social issues (including a host of topics not addressed here such as international trade agreements, rural community viability, rural-urban conflicts, etc.) indicate a need for additional research that considers the breadth of the systems involved at scales that are appropriate. For example, much of our current knowledge of the efficacy of conservation practices is based on field scale research which cannot be simply "scaled-up" to understand the workings at watershed levels. While current research efforts are beginning at this more challenging scale, definitive results will be, in many cases, many years off.

Before we leave the reader to dive into the three following papers, we note a final thorny point concerning the potential for significant "legacy" problems possibly hiding in groundwater supplies. Over many decades of agricultural activity, we have added nutrients and other effluents to groundwater systems that have undoubtedly not yet emerged at the surface. When and where such pollutants will appear is not clear, but if conservation programs are designed only with current pollutant contributions in mind, our efforts may well fall short due to these legacy sources.

For More Information

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A Tale of Three Watersheds: Nonpoint Source Pollution and Conservation Practices across Iowa

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JEL Classification Code: Q25

Many conservation practices and implementation programs exist to address nonpoint source (NPS) pollution losses from agricultural landscapes (Helmers et al., this issue). In order to select the most appropriate practices and programs for reducing NPS pollution in a specific region while maintaining economic return for the landowner, the interacting processes of agricultural management and watershed hydrology need to be understood across broad spatial scales. On a nationwide basis, it is easy to see how NPS pollution in one part of the country might be different than those in another region of the country. For example, cotton growers in the South, dairy farmers in the Northeast, cattle ranchers in the West, and grain farmers in the Midwest all face unique challenges based on differences in climate, soil types, and cropping patterns. Each region relies on a different set of conservation practices and programs to address NPS pollution. To be effective, conservation systems must be based on an understanding of specific management impacts on water quality problems, and therefore be targeted to reduce, intercept, and/or treat contaminants moving via surface or sub-surface pathways from working agricultural lands.

Within agricultural regions, one might expect greater homogeneity in biophysical features and cropping practices and be tempted to think that one size fits all; i.e., that one set of conservation prescriptions can be used to address the negative impacts of agriculture on aquatic and terrestrial integrity. If this generalization could be made anywhere, certainly a state such as Iowa, dominated by its vast extent of corn and soybean fields, would be suited for a limited set of conservation prescriptions. However, as described in this tale of three watersheds, conservation practices must instead be tailored to individual landowner objectives and local landscape conditions in order to optimize their effectiveness.

The research described in this paper was conducted as part of USDA's Conservation Effects Assessment Project (CEAP) and its Watershed Assessment Studies (Mausbach et al., 2004). The objectives of the project are to evaluate the effects of agricultural conservation practices on water quality, with a focus on understanding how the suite of conservation practices, the timing of these activities, and the spatial distribution of these practices throughout a watershed influence their effectiveness. An additional component of the project is to evaluate social and economic factors influencing implementation and maintenance of practices.

Watershed Descriptions

To evaluate the effects of watershed conservation practices on water quality, and to assess the spatial distribution of these practices, we are focusing on three watersheds in distinct landscape regions of Iowa (Figure 1). By studying three watersheds with differing physical characteristics and possessing a unique set of pollutants, practices and programs, we can better assess the effectiveness of conservation activities and land management decisions.

Landforms

The Sny Magill Creek, Squaw Creek, and the South Fork of the Iowa River (South Fork) watersheds are representative of three distinct landform regions of Iowa (Prior,

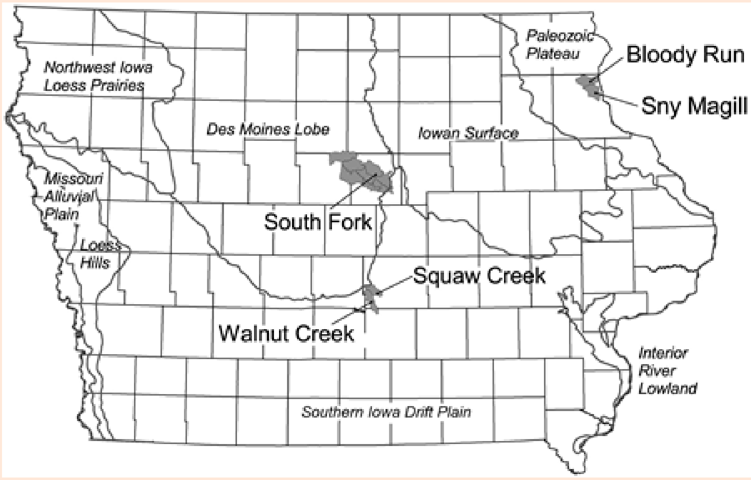


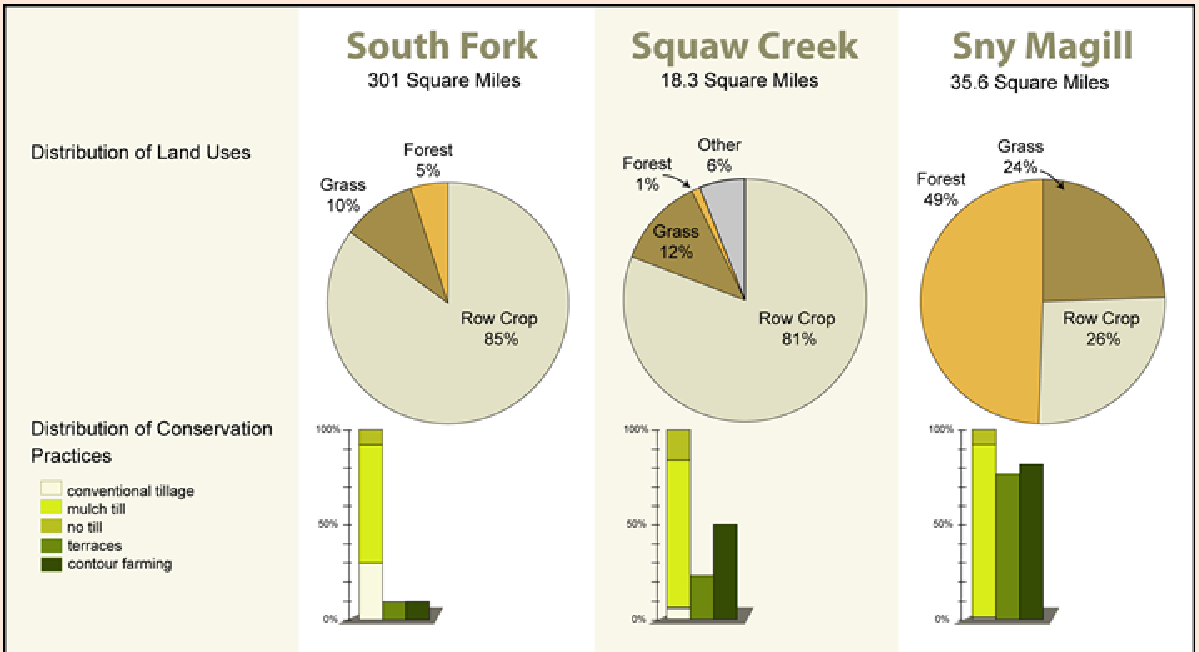
Figure 1. Location of the three watersheds (and controls) and EPA Ecoregions in Iowa.

Figure 2. Subsurface hydrologic features in Sny Magill and South Fork watersheds. In the Sny Magill watershed (left photo), springs and caves discharge natural drainage from unknown areas of karst terrain. In the South Fork watershed, once-prevalent wetlands were converted to cropland with tile drainage. This 36-inch clay pipe (right photo) has discharged drainage collected from about 4,500 acres of cropland for nearly 100 years (note monitoring lines).

1991). In Northeast Iowa, Sny Magill Creek is a third-order stream in Clayton County that drains 35.6 mi^2 of the Paleozoic Plateau landform region before discharging directly into the Mississippi River. The landscape of this region is characterized by narrow, gently sloping uplands that break into steep slopes with abundant outcrops of sandstone and limestone. The characteristic limestone bedrock of the area gives rise to karst features (sinkholes, caves, and springs) that are found throughout the Sny Magill Creek watershed (Figure 2). Nearly 80% of annual streamflow is 'baseflow' attributable to ground water discharge from these subsurface sources. This results in "cold water" conditions suitable for highly popular trout fisheries.

The 18.3 mi^2 Squaw Creek watershed is located in South-Central Iowa in Jasper County in the Southern Iowa Drift Plain. The landscape of this region is characterized by steeply rolling hills and a well-developed stream network that developed on a landscape composed of geologically old (>500,000 years) glacial till (poorly sorted mixture of gravel, sand, silt, and clay) overlain by geologically recent (17,000 to 31,000 year old) windblown silt (loess). Because of the sloping hillsides and poor infiltration capacity of the soils, rainfall is primarily directed to streams via overland runoff, and only 55% of the stream discharge is attributable to baseflow originating as ground water.

The largest of the study watersheds is the South Fork of the Iowa River, which covers 301 mi^2 within Hardin and Hamilton counties in Central Iowa. The landscape is representative of the Des Moines Lobe, the dominant landform region of North-Central Iowa. The terrain is young (about 12,000 years since glacial retreat), and thus much of the landscape is dominated by low relief and poor surface drainage. Prior to settlement by Europeans, the landscape was a complex of wetlands, and the stream network was poorly developed due to the relatively young landscape. The geology of the Des Moines Lobe region consists largely of glacial till deposits in moraines and flat to rolling uplands, clay and peat in depressional "prairie pothole" areas, and sand and gravel deposits in floodplains of rivers and streams. Soil wetness is a major concern for land management and agricultural production. Hydric soils (indicative of soil saturation on at least a seasonal basis) occupy about 54% of the watershed, and artificial tile drainage (Figure 2) was installed in these highly productive and nutrient rich soils to lower the water table and allow crops to be grown. Thus, about 70% of the stream flow in the South Fork watershed originates from subsurface drainage (Green et al., 2006), with most tile discharge occurring during spring and early summer.



	South Fork	Squaw Creek	Sny Magill
Physical Features			
Slope	1.88%	4.79%	10.63%
Drainage Density*	0.27	0.78	0.68
Water Quality			
Mean NO_3-N (mg/L)	14.2	9.5	2.8
Median Fecal Bacteria (cells/100 mL)	488	600	130
Median P (mg/L)	0.07	0.14	0.05
Sediment (tons/acre)	0.27	0.69	0.26
Livestock			
Hogs/square mile	2,034	0	158
Cows/square mile	10	53	104
Chickens/square mile	2,880**	0	0

*Drainage density is defined as miles of stream length per square mile of watershed area.

**All from one 900,000 layer facility in the South Fork.

Figure 3. Land use, conservation practices, and other characteristics of the three watersheds.

Physical features of the three watersheds have a great influence on the timing and magnitude of the routing of water to streams. Watersheds draining older landscapes have greater slope and greater stream density (number of streams per square mile) than younger landscapes (Figure 3). For example, the Sny Magill watershed has twice the average slope as Squaw Creek, which has more than twice the slope of the South Fork watershed. Slopes in Sny Magill are further accentuated because of the bedrock terrain and its proximity to the Mississippi River. The Squaw Creek and Sny Magill watersheds also have nearly three times more streams per square mile than the South Fork. The well-dissected landscape of the Sny Magill watershed shows a greater stream density; thus, rainfall can be quickly routed as overland runoff to sinkholes or streams. In the South Fork watershed, where natural drainage is poor, excess rainfall would collect in potholes or other surface depressions if not for prevalence of subsurface tile drainage, which has accelerated the routing of rainfall water off the land. Watersheds draining the Des Moines Lobe may yield as much water as those draining fractured carbonate bedrock (Schilling and Wolter, 2005).

Relation of Land Use to the Landform Region

Differences in land cover among the three watersheds can be traced largely to their watershed morphologies and the suitability of land for intensive row crop agriculture. Row crops in the Sny Magill watershed, primarily found on narrow upland divides and bottomlands, only comprise 26% of the land area (Figure 3). Grasses and forest are widespread in the Sny Magill watershed, located on steep terrain that is difficult to cultivate. In the Squaw Creek watershed, slopes are not as severe as in the Sny Magill watershed, and row crops are found on 81% of the land area. Grasses are distributed around the watershed on highly erodible land, a practice encouraged by conservation programs. The till plain of the Des Moines Lobe, represented by the South Fork watershed, is also heavily utilized for row crop production, which occupies 85% of the watershed area.

Animal Agriculture

In the early 1900s, most small farms in Iowa had livestock, often including cattle, swine, and chickens. As a result of changes in farm policy and economies of scale, all three watersheds have experienced shifts in animal agriculture that are representative of changes across the larger landform regions (Figure 1). Historically, the Sny Magill watershed had significant numbers of dairy cattle utilizing available grasslands for forage. While still a significant industry, dairy cattle within Sny Magill have decreased greatly, with a resulting shift in some grassland acreage to row crop agriculture (soybean acreage especially increased in Iowa as pasture and hayland decreased). Livestock is comparatively absent in the Squaw Creek watershed except for several cow-calf operations. Nowhere is the concentration of livestock more apparent than in the South Fork watershed, where most swine and chickens are raised in confined animal feeding operations (CAFOs). There are nearly 100 CAFOs (mostly swine) in the watershed. Based on inventories reported for permitted facilities, hogs and chickens in the South Fork watershed number 1,654 and 2,880 per square mile, respectively, which are densities considerably greater than the other two watersheds combined (Figure 3). All the reported chickens are housed in one large egg-producing facility, while swine facilities are abundant across the central part of the watershed. We estimate that about a quarter of the watershed receives manure applications annually, assuming this is applied prior to corn at a rate equivalent to that crop's uptake of nitrogen (about 190 lb N/ac). Usually these applications are done by injection, and carried out in the fall when soils tend to be dry and most easily trafficked by manure tankers and applicators.

NPS Pollutants and the Landscape

Because of their different watershed characteristics, land use, and livestock histories, non-point pollutant sources and transport vary greatly among the three watersheds (Figure 3). Pollutants of particular concern in Iowa are sediment, nutrients (nitrate and phosphorus), and fecal bacteria (*E. coli*). In Iowa, nitrate concentrations in streams relate to the amount of row crops in a watershed (Schilling & Libra, 2000), and nitrate-N concentrations are highest in the South Fork and Squaw Creek watersheds, with median concentrations of 14.2 and 9.5 mg/L, respectively. Tile drains contribute greatly to nitrate losses in the South Fork watershed. In the mid-1990s, the USGS found stream nitrate concentrations in the South Fork watershed to be among the highest observed in the United States (Becher et al., 2001).

In contrast, nitrate concentrations are considerably lower in the Sny Magill watershed, averaging 3.3 mg/L over 10 years, a value that would be the envy of most other regions of Iowa. The smaller concentration results from the differences in land use (Figure 3). Fecal bacteria counts are also highest in the South Fork watershed; however, multiple sources of bacteria are suspected because patterns do not always follow the distribution of livestock. Yet, research suggests that these bacterial losses in runoff are greatest when that runoff occurs within several weeks of manure application. Fecal bacteria concentrations in Squaw Creek are also elevated, which may be surprising, given the lower livestock densities. However, cattle with direct access to the streams, wildlife, and inadequate septic systems may all contribute to fecal contamination of Iowa streams.

Sediment loss is also a major concern in these watersheds (Figures 3 and 4). The greatest annual sediment loss per unit area was associated with the Squaw Creek watershed (0.69 tons/ac per year), whereas mean annual sediment loss from Sny Magill and South Fork watersheds averaged 0.26 and 0.28 tons/ac, respectively. Considering that row crops cover only 26%of the land area in the Sny Magill watershed, actual soil loss per acre of cropland may be substantially greater. Long-term sediment monitoring data in the Sny Magill and Squaw Creek watersheds indicates that sediment transport is very flashy in both watersheds, with much of the annual sediment loss transported by runoff from a few intense rainfall events. In Squaw Creek, on average, about 40% of the water-

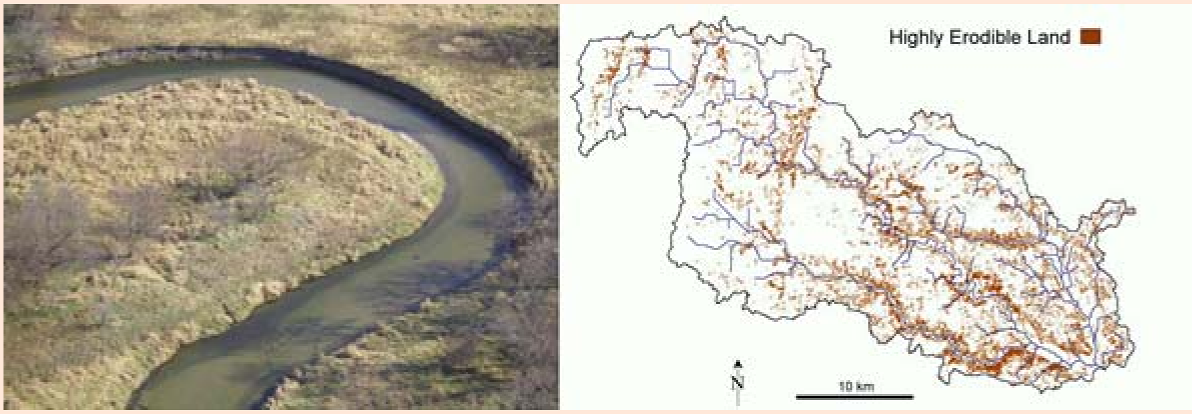


Figure 4. Eroding streambanks and erodible soils are possible contributors to sediment loads. In the South Fork watershed, both cut-bank meanders and erodible soils are mostly found in the lower (eastern) part of the watershed.

shed's annual sediment loss occurs on the single day of greatest runoff.

Sediment losses from watersheds result from overland flow across the landscape, causing sheet, rill, and gully erosion, as well as substantial contributions from streambanks. In the South Fork watershed, sediment losses are actually about three times higher than typically measured in the Des Moines Lobe region. In the lower third of the watershed, lands become more highly erodible in an area of hilly moraines near the Des Moines Lobe's edge, and the river erodes its banks as it meanders across an alluvial valley (Figure 4). In some Iowa watersheds, streambank erosion can contribute more than half of the annual sediment load exported from a watershed.

Phosphorus is strongly adsorbed to sediment, which was reflected in Squaw Creek having both the greatest average sediment yields and greatest median concentrations of total P (0.14 mg/L) among the three watersheds. Squaw Creek's median P concentration is more than twice what EPA has proposed as the standard for Midwest streams. Concentrations of P in the South Fork, by comparison, had a median of 0.07 mg/L during three years of weekly-biweekly sampling. Recent groundwater sampling from 24 wells located throughout the South Fork watershed has shown median and maximum total P concentrations of 0.030 and 0.340 mg/L, respectively. These groundwater P concentrations are found in similar materials and landscapes in Iowa (Burkart et al., 2004), and suggest that groundwater can also be a P contributor to streams.

Tailoring Conservation Practices to Watersheds and NPS Pollutants

Conservation practices used on row crop fields in the three watersheds reflect respective watershed characteristics and land use histories. A field-by-field assessment of conservation practices was conducted in each watershed to assess the variety and distribution of practices. An analysis of tillage practices, terraces, and contour farming shows the degree to which land managers have used these conservation practices to reduce nutrient and sediment losses from the three watersheds. Of the three tillage practices assessed (conventional tillage, mulch till, and no till), mulch till was most widely utilized in all watersheds. Mulch tillage (>30% residue cover) was used on 62 to 91% of all row crop fields, whereas no till was used on 8 to 16% of the fields. Conventional tillage (<30% residue cover) was rarely used in Sny Magill and Squaw Creek, but was used on 30% of cropland in South Fork. Erosion losses from crop fields in South Fork are not a major concern in the flat, till plain portion of the watershed. In areas of the Northern United States, with relatively flat terrain and poorly drained soils, many producers still view conventional tillage as the most viable practice because the exposed soil is warmed faster in spring, often allowing earlier seeding and emergence of the crop.

In the Sny Magill watershed, contour farming, terraces, and other engineered structures are prevalent practices for reducing sediment losses from the steeper slopes in that watershed. Although row crop fields occupied only 26% of the land area in Sny Magill, most are terraced (77%) and/or farmed using contour planting (92%). Other engineered conservation practices are also used extensively throughout the Sny Magill watershed, including a total of over 150 sediment basins and grade stabilization structures. Terraces are not as common in Squaw Creek (23%), but half the farm fields are planted on contours. Terraces and contour farming are not common in the South Fork watershed; fields with terraces occupy less than 10% of the watershed's cropland.

A Tale of Three Watersheds - revisited

In this tale of three Iowa watersheds, significant differences in NPS pollutants and practices emerged in a state considered by many to be uniformly agricultural. Much of the differences can be attributable to their unique landform history that has been exploited uniquely for intensive row crop and livestock development. In the South Fork watershed, extensive wetlands on recently glaciated till plains were drained by settlers, and agricultural development then intensified during the past century. The land is well suited for crop and livestock production, but subsurface tile drainage increases losses of nitrate, and the rapid routing of tile discharge, combined with surface runoff, may enhance movement of bacteria, P, and sediment. In Squaw Creek, with steeper slopes in row crops, conservation practices such as reduced tillage and contour farming methods are more prevalent. However, losses of nutrients, sediment, and fecal bacteria remain as major concerns in the watershed, possibly because hydrologically sensitive areas are used for row crops or grazing. Row crop acreage in the Sny Magill watershed constitute only about 25% of the land area and most row crop fields have conservation tillage or structural practices such as terraces. However, the steep slopes and karst drainage in the watershed make Sny Magill watershed perhaps the most vulnerable among the three streams evaluated.

Apparent in this tale of three Iowa watersheds is that, in order to provide the greatest return on the public's investment in conservation, it is imperative that practices be tailored to the most local of landscape conditions and landowner objectives. Targeting is needed to place specific conservation practices on the land to either reduce pollutant concentrations or attenuate their transport. No single practice can be viewed as the answer in all cases, and a one-size-fits-all approach is likely doomed to failure, or at least doomed to provide little return on the public investment. Recent advances in assessment technologies and record keeping are only now beginning to allow us to understand the distribution of practices on the land and their impacts on water quality. Significant challenges remain to develop better assessment, monitoring, and modeling techniques to capture the inherent differences among our watersheds in order to design conservation practices and programs providing greater water quality benefits for lower cost. The challenges are not only in assessing resource needs against the mosaic of land use and terrain that occur within watersheds, but also to then develop better policy and planning tools that can help achieve watershed-scale conservation goals through implementation at the individual farm scale.

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Watershed Highlight 1: Historical and Human Dimension: Squaw Creek and Walnut Creek Paired Watershed Study.

Because Squaw Creek represents typical agricultural land management in Southern Iowa, the watershed was selected to be the control basin for a large land use experiment occurring in the neighboring Walnut Creek watershed (Figure 5). In the Walnut Creek watershed, large tracts of row-cropped land are being reconstructed to native prairie at the Neal Smith National Wildlife Refuge by the U.S. Fish and Wildlife Service. Before restoration began, land cover in both watersheds was about 70% row crop. From 1992 to 2005, nearly 220 acres of prairie was planted each year, so that by 2005, native prairie occupied 23.5% of Walnut Creek watershed. Surface water samples collected in the treatment (Walnut Creek) and control (Squaw Creek) watersheds from 1995 to 2005 documented the effects of prairie restoration on water quality (Schilling et al., 2006). Stream nitrate concentrations were found to have decreased 1.2 mg/L over the 10-year project period at the Walnut Creek outlet, with nitrate concentrations decreasing up to 3.4 mg/L over the same time period in one monitored subbasin with substantial landuse conversion. Interestingly, land use in the control basin of Squaw Creek did not remain static during the same 10-year monitoring period. Row-crop land area increased 9.2% in Squaw Creek as lands previously enrolled in the Conservation Reserve Program as grassland were converted back to row crop production in the late 1990s. Stream nitrate concentrations increased 1.9 mg/L at the Squaw Creek outlet, with annual nitrate in one monitored subbasin increasing nearly 12 mg/L in 10 years where substantial acres were converted to row crops. These results attest to the sensitivity of water quality parameters to changes in watershed management that are, in aggregate, the result of individual landowner decisions.

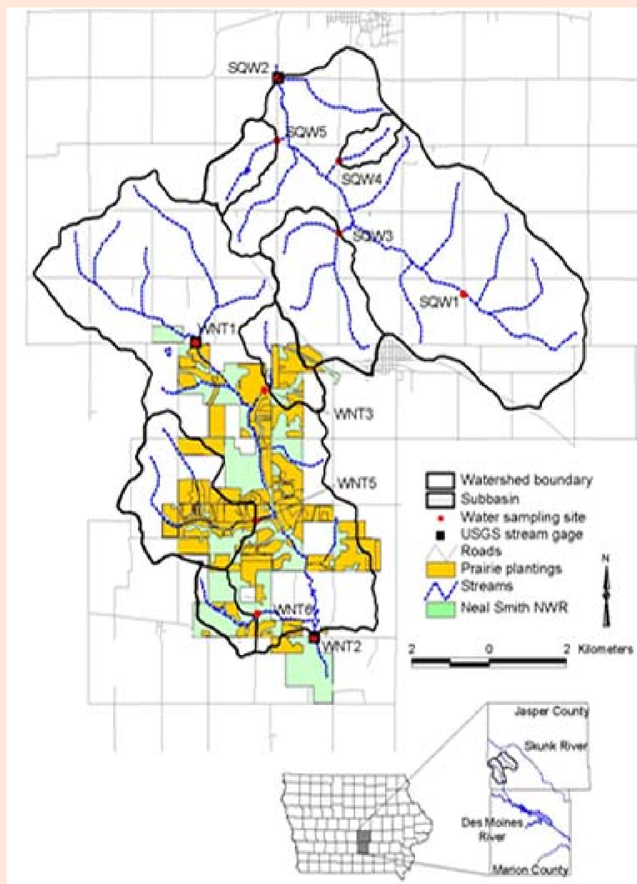

Figure 5. Extent of Prairie plantings in the Neal Smith National Wildlife Refuge within the Walnut Creek watershed.

Watershed Highlight 2: Long-term Implications of BMP Implementation in the Sny Magill Creek Watershed.

Three separate projects were carried out spanning the time period of 1988 to 1999 to improve water quality in the Sny Magill Creek watershed. The cumulative adoption percentages and total levels of key BMPs implemented during the 1990s through the Sny Magill Hydrologic Unit Area (HUA) and Sny Magill Creek Watershed projects are listed for selected years in Table 1. The cumulative adoption of terraces in the watershed is also shown in Figure 6 for 1991, 1995, and 2005. A paired watershed approach was used to assess Sny Magill Creek water quality improvements from 1992 to 2001 (Fields et al., 2005). Analysis of Sny Magill stream flow and water quality data collected during 1991-2001 was performed using a pre/post statistical model.

The statistical results indicated that discharge at the watershed outlet increased by 8% over the 10-year period; this could partly be due to routing of runoff water captured by terraces into surface inlet drains (that are often installed just upslope of a terrace) and to the stream. The statistical analysis also showed that the BMPs installed during the 1990s resulted in a 42% decrease in turbidity but only a 7% decrease in total suspended solids (TSS). The TSS results imply that stream bed and bank erosion continued to contribute significant sediment loads to Sny Magill Creek, even after BMP installation reduced sediment delivery from upland areas. The increase in discharge may have further magnified the in-channel sediment contributions. Overall, the TSS results suggest that a long lag time may occur before the full impacts of the installed BMPs are realized.

The statistical analysis also revealed that an increase in nitrate concentrations of 15% was found at the SMCW outlet. This indicates greater N leaching, which is consistent with increased infiltration of rainfall that naturally results when conservation practices successfully decrease surface runoff. However, the nitrate concentration level still only slightly exceeded 3 mg/L at the end of the 10-year time period, which is quite low compared with the concentrations measured in most other Iowa stream systems, including the South Fork and Squaw Creek watersheds.

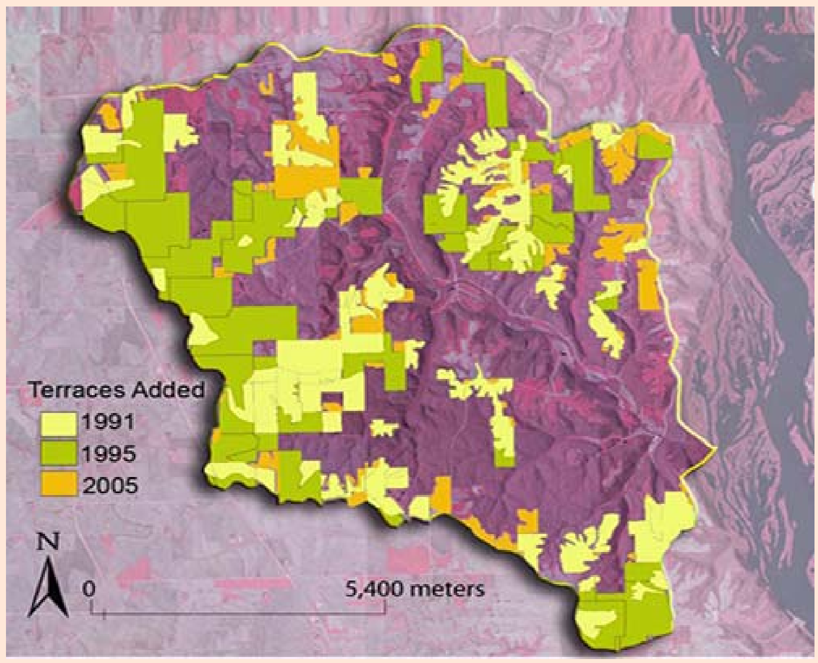

Figure 6. Cumulative additions of terraces to specific land tracts in the Sny Magill Creek Watershed.

Table 1. Cumulative percentages of total BMP adoption that was cost shared by year (expressed as a percentage of the total amount implemented as given in the bottom line).

Year	Terrace	Subsurface tile	Sediment basin	Grade stabilization	Field border	Contouring
1992	28	22	28	92	16	11
1995	65	65	79	93	99	53
1998	95	94	98	100	100	100
2001	100	100	100	100	100	100
Total Units	269,585 ft	160,345 ft	61 total	90 total	26,700 ft	1,907 ac

Watershed Highlight 3: Evaluating Targeting of Conservation Practices in the South Fork Watershed.

The Conservation Title of the 1985 Farm Bill (Food Security Act) included provisions to reduce soil erosion on highly erodible land (HEL) through conservation practices such as Conservation Reserve Program (CRP) plantings and reduced tillage. Land enrolled in CRP was planted to perennial, non-harvested vegetation for at least a ten-year period in exchange for annual rental payments. Soil survey data, including slope, soil texture and depth are used to identify HEL. Those producers farming on HEL-dominated fields were to employ reduced tillage practices to remain eligible for USDA commodity programs; this was known as the conservation compliance provision of the 1985 Farm Bill.

A one-time inventory of conservation practices in the South Fork watershed was conducted during 2005. We compared the distribution of no-tillage management and CRP plantings with the distribution of HEL, which occupies 12% of the watershed (Figure 7). Very little (2.4%) of the watershed's cropland had been enrolled into CRP by producers, partly because this is some of the most productive rain-fed agricultural land in the U.S. While CRP has also been used to install buffers along streams and around livestock facilities, there has been apparent success in targeting of CRP towards HEL. That is, the proportion of HEL in the watershed in CRP is 4.6%, as opposed to only 2.2% of non-HEL (Table 2). The same is true of no-tillage practices that are highly effective in controlling erosion: although relatively few producers in this watershed have implemented no-tillage, largely due to concerns about planting delays during wet, cool spring conditions, a greater proportion of HEL (11.3%) is under no-tillage than is non-HEL (6.7%). There is little comparative data to evaluate whether these practices are better targeted towards HEL in this watershed than in other areas. However, targeting success may also be indicated if conventional tillage practices that increase soil susceptibility to erosion have shifted away from HEL as these conservation practices were implemented. This does not appear to be the case, as conventional tillage occupies nearly the same proportion of HEL and non-HEL cropland, to within 2%.

It is important to note that in 1985, when current policies where initiated, most of this watershed was probably tilled conventionally. This inventory offers only a snapshot of conservation practices. Current conservation policies, which have had a goal of controlling soil erosion from the most sensitive soils for 20 years, have encouraged better management on the most vulnerable lands in the watershed. Yet, the least desirable tillage practices apparently have not preferentially shifted away from HEL. This raises questions about social-behavioral responses to conservation policies, which are made by individual producers, yet in sum determine the impact of those policies in each watershed.

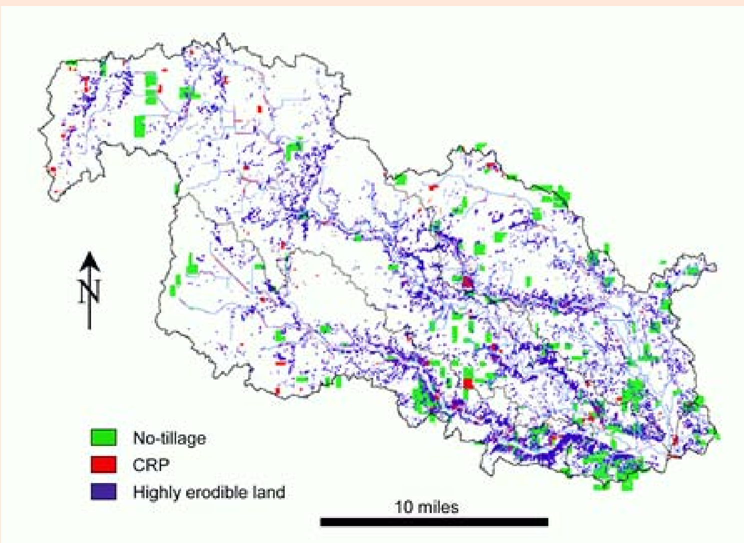

Figure 7. Distribution of key conservation practices for erosion control in the South Fork watershed, compared to the distribution of Highly Erodible Land.

Table 2. Comparative distributions of CRP and no-tillage conservation practices on highly erodible and non-highly erodible lands in the South Fork watershed, along with conventional tillage practices.

Management	HEL (12%)	Non- HEL (88%)	Total (100%)
Conservation Reserve Program	4.6%	2.2%	2.4
No-tillage	11.3%	6.7%	7.2
Conventional tillage	28.0%	30.0%	29.8

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Privatizing Ecosystem Services: Water Quality Effects from a Carbon Market

By Silvia Secchi, Manoj Jha, Lyubov Kurkalova, Hongli Feng, Philip Gassman, and Catherine Kling

JEL Classification Code: Q25

With the specter of a new farm bill on the horizon, considerable discussion is occurring concerning the possible redirection of conservation programming and financing. Notably, interest in the increased use of incentive systems and market-like instruments continues to expand. One source of this interest lies in the desire to shift some of the burden of providing ecosystem services, such as protecting stream and river channels from erosion, maintaining biodiversity, and providing clean water and air, to private sector pockets. For example, in the fall of 2006, USDA and EPA announced a joint partnership to support expanded water quality credit trading for nutrients in the Chesapeake Bay watershed, allowing farmers to receive compensation for water quality improvements. Carbon markets, such as the active program in the European Union, are also being discussed as a possible model for expanded market-like programs in agricultural conservation policy.

While the potential cost effectiveness of providing environmental goods from incentive-based methods appears to be broadly understood, there is an additional attribute that is less broadly acknowledged: due to numerous inter-linkages in natural ecosystems, the development of a market that provides one ecosystem service may significantly change the level of provision of other ecosystem services. Thus, by developing the institutional structure to support and encourage the provision of one ecosystem service, changes, either positive or negative, in other services may result.

The example we consider here is the case in which a carbon market that would allow U.S. farmers to receive payment for sequestering carbon when they retire land from production is implemented. This could occur if the United States were to unilaterally implement such a market, or if at some future time the United States chose to sign on to a Kyoto-like accord, where carbon sinks were allowed to generate credits that could be traded to meet mandatory carbon reduction requirements. Under such a scenario, land retirement decisions would be driven by the prices paid for carbon and the amount of carbon that a particular parcel could sequester (we abstract from the important question of measuring the carbon storage potential of each parcel see the excellent work of Mooney et al., 2004).

Removing a parcel of land from production will change the suite of environmental benefits associated with the parcel. In many cases, these effects are likely to be positive for example, taking land out of active agricultural production and placing it in perennial cover or forested lands will usually yield reduced erosion and nutrient runoff relative to row crop agriculture. Indeed, the findings of our study are consistent with this outcome. However, if conservation practices are already in place on a working land field, water quality improvements from retiring the land might be small and, in fact, could be negative. The latter could occur if land retirement results in the planting of land cover that, on the whole, is not as effective in capturing nutrients and sediments as a working land system that already has effective conservation practices and management in place.

In this paper, we consider the possible water quality consequences of a carbon trading policy that allows farmers to receive carbon credits from retiring their land from agricultural production. To do so, we make a number of simplifying assumptions about the structure of the carbon market and the choices farmers make in response to the

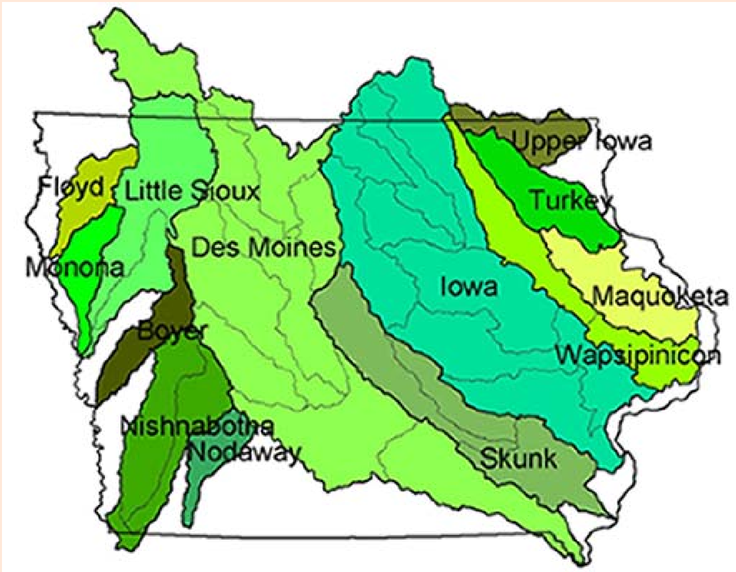

Figure 1. Study area and watershed delineations.

existence of that market; many of these assumptions may not, in fact, represent how an actual market might be implemented. Rather than view the results of this analysis as definitive, we present the findings in the spirit of raising awareness of the potential environmental consequences that can occur when a single environmental benefit or target (carbon sequestration) forms the basis of environmental policy, as would be the case if carbon trading markets that allowed land retirement to yield carbon credits were functioning with high carbon prices.

A Bit about Our Data and Models

To develop our models, we draw heavily from the National Resource Inventory to provide data on the land use, cropping history, and farming practices in the state of Iowa. This inventory is the most comprehensive data set on land use in the United States, and we use data on the 14,472 physical points in Iowa that represent cropland. Conceptually, our data and models are based on individual producer and farm-level behavior, and we treat an NRI point as a producer with a farm size equal to the number of acres represented by the point (the expansion factor provided by the survey). Figure 1 illustrates the 35 watersheds corresponding to the United States Geological Survey 8-digit Hydrologic Cataloging Units that are largely contained in the state and are modeled in this study. To compute the amount of carbon sequestered when a land unit is retired from cropland, we rely on estimates from the Environmental Policy Integrated Climate Model version 3060. When land is retired from crop production, we assume that annual grasses are planted and maintained on the land, and we run a 30-year simulation to predict the carbon sequestration level associated with this change.

In addition, we also rely on estimates from a watershed-based model to assess the conservation policies. Unlike carbon sequestration, the degree to which land retirement improves in-stream water quality depends on critical interactions between land uses in different locations within a watershed. Thus, for otherwise identical tracts of land, more water quality improvement may occur from retiring a piece of land from production that is located downstream from numerous other cropped points relative to one that is not. The potential filtering effect is just one example of the physical processes that need to be captured to assess the in-stream water quality effects of land retirement.

So that we can capture these land use interactions within a watershed setting, we employ the Soil and Water Assessment Tool, a biophysical water quality model, to estimate changes in nitrogen, phosphorous, and sediment loads from retiring a particular set of parcels from production within a watershed. To estimate the in-stream water quality consequences of the increase in land set aside, we have calibrated the water quality model for each of the watersheds identified in Figure 1 to baseline levels (Jha et al., 2005; Gassman et al., 2005). By running the model at the set-aside levels "after" the policy, we can compute the changes in water quality attributable to the increase in land set-aside. The watersheds studied correspond to 13 outlets, at which the in-stream water quality is measured. The water quality measures of interest are sediment, nitrogen, and phosphorus.

Water Quality Effects of a Carbon Market

To demonstrate the possible consequences of a carbon market that pays farmers for the sequestration of carbon in agricultural soils on water quality, we consider a simple scenario. Suppose that through an active carbon market, the price of carbon is such that about 10% of Iowa cropland is retired; suppose further that the cost of retiring all land within the state is about the same. While rental rates for farmland do vary across the state, they vary relatively little with respect to productivity (see Secchi & Babcock, forthcoming), and this simplifying assumption allows us to focus on environmental outcomes of the scenario without overly complicating the analysis. Under these assumptions, the cropland that will be removed from production will be the land that produces the highest carbon sequestration benefits per acre as this land will earn the highest return from carbon sequestration credits. In consequence, the land removed from production may or may not represent significant portions of the watersheds under consideration.

Based on this scenario, the land retired would be focused in the central part of Iowa, in the ecoregion known as the Des Moines Lobe, a flat area, with very productive agriculture and particularly suited for carbon sequestration. Figure 2 illustrates the quantity and location of the carbon that the carbon simulation model predicts would be sequestered across the state under this scenario. Approximately 2 million acres of land is removed from production under this scenario with about 2.7 million tons of carbon being sequestered annually.

Does this land retirement, induced by a private market that pays for ecosystem services, yield other environmental benefits to the region? To answer this question, we estimate the in-stream water quality effects of this land retirement using the water quality model and present the percentage reductions in three common indicators in Figures 3-5.

Figure 3 reports the estimated in-stream sediment reductions from the retirement of this set of land parcels.

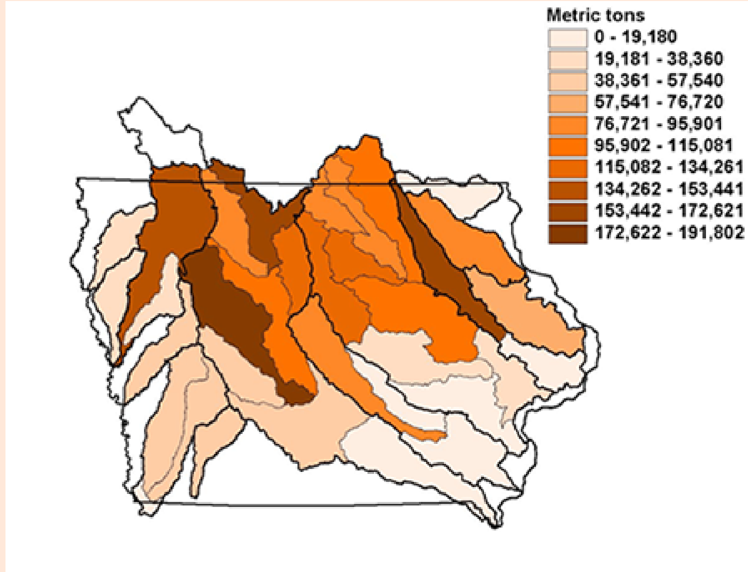

Figure 2. Carbon sequestration from carbon trading.

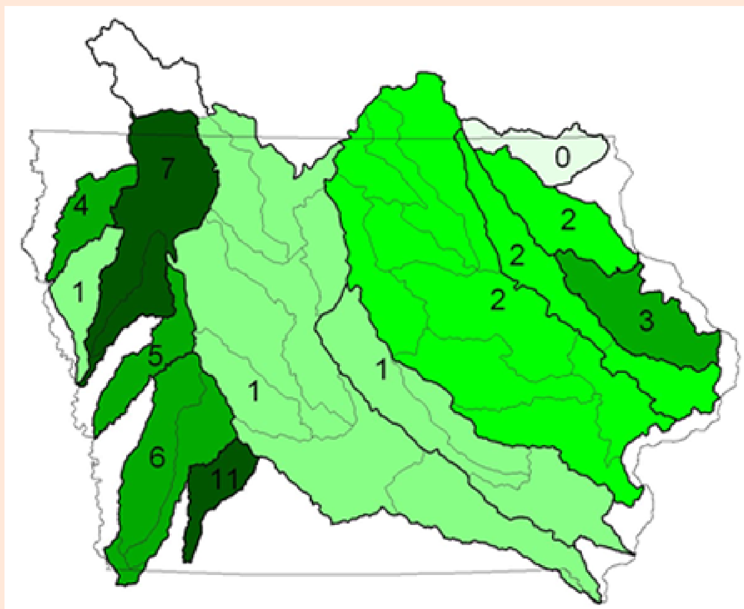

Figure 3. In-stream sediment changes from carbon trading (percentage reduction).

We find that for the two largest watersheds whose sediment dynamics is influenced by the presence of large reservoirs, the Des Moines and Iowa River watersheds, there is only a little improvement in sediment. In contrast, there are larger reductions in sediment in the South-Western part of the state, likely because the land that is retired from production there is more erodible. In general, however, sediment reductions in percentage terms are lower than the reductions in nutrients, because land that has high carbon sequestration potential also has good productivity levels and is, therefore, more heavily fertilized.

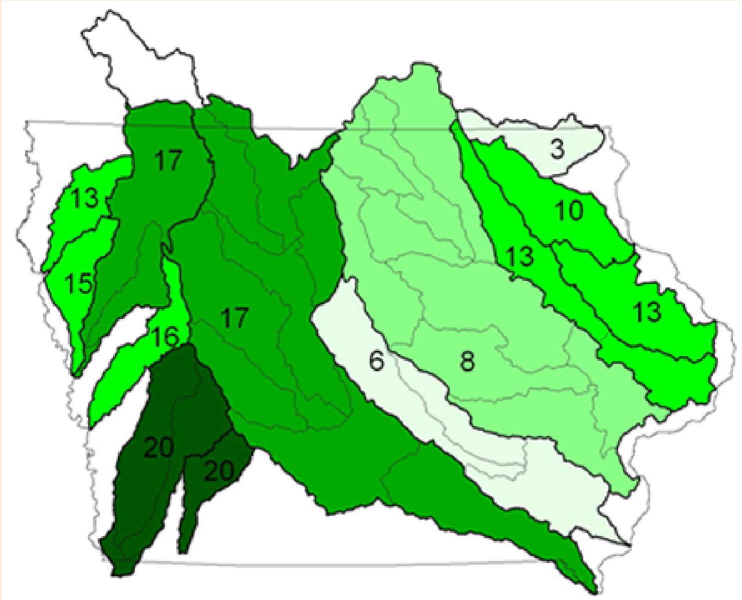

Figure 4. In-stream nitrogen changes from carbon trading (percentage reduction).

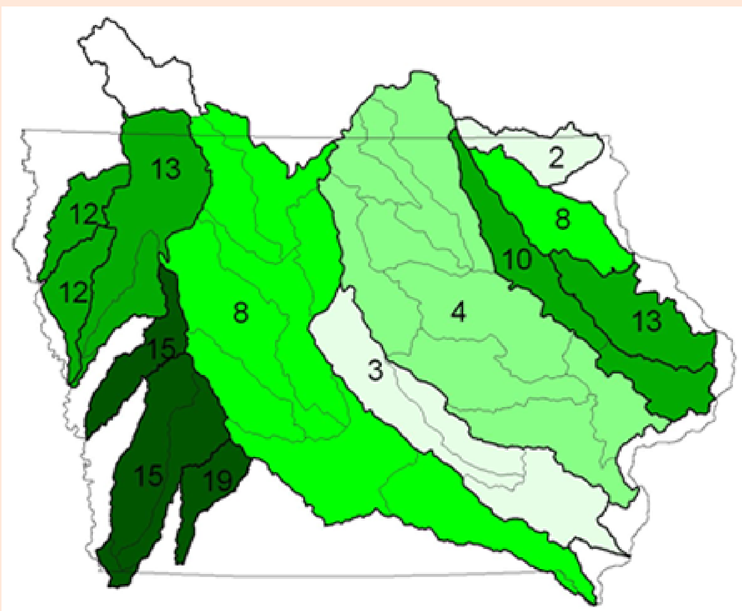

Figure 5. In-stream phosphorous changes from carbon trading (percentage reduction).

The retirement of land generally improves the total N level as seen in Figure 4. The reason for the improved N levels in many cases, as mentioned above, is that the land taken out of production is largely prime agricultural land: heavily fertilized and reliant on tiled drainage systems. Some of the highest reductions, a total of over 10,000 tons annual average, are in the Des Moines River watershed, which comprises large parts of the Des Moines Lobe and includes some of the most productive land in the state where most of the acres of land retirement are located.

Finally, Figure 5 reports the results for the in-stream phosphorous levels predicted to occur as a result of the carbon trading program. Like the sediment results, the Western watersheds show the highest improvements. This is not surprising given the sediment results, since phosphorous typically moves with sediment.

The development of more market-like programs to provide ecosystem services from agriculture is a concept with expanding interest. In this paper, we have identified an additional issue associated with this strategy, changes in other environmental goods of interest. In the case analyzed here, these changes were all positive; thus, the market-based system yields positive gains for other ecosystem services. By recognizing that a system that pays for carbon sequestration via land retirement potentially has effects on other environmental services, and that the spatial distribution of different environmental services is likely to differ, policy makers can incorporate these effects in planning and implementing markets for ecosystem services.

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Nitrate Reduction Approaches

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As noted in the overview to this set of papers, water quality continues to be a growing concern. Nutrients applied as commercial fertilizer and manure enter surface and ground water, leading to several forms of water quality impairment. These impairments manifest themselves in a number of ways. Excess phosphorus is responsible for algae blooms, losses in water clarity, and even the presence of toxic cyanobacteria in fresh water. Excess nitrogen is believed to be the limiting factor in low-oxygen dead zones in several dozen locations around the globe. In some locales, nitrate concentrations reach levels that are toxic to both humans and aquatic animals. In the United States, local nitrate concentrations are largely uncontrolled. The only widely applied standard affects water used for human consumption. This is regulated by the Environmental Protection Agency via National Primary Drinking Water Regulations (EPA NPDWR). Similar requirements and guidelines exist in Canada and Europe.

Several technologies can remove nitrates directly from water and are employed by municipal water works in order to comply with drinking water standards during periods of high nitrate concentrations in source water. These technologies are costly to operate, suggesting an opportunity for cost savings via upland reductions of fertilizer application. This article explores possible tradeoffs in the context of a nutrient-application-right trading scheme. Simulations of both water quality and economic effects in a test watershed suggest that simple upland fertilizer reductions are more costly than direct nitrate removal if the goal is compliance with drinking water standards. Other water quality goals merit consideration, but are difficult to model without objective standards and given the current nitrate removal technology.

Watershed Background

The area used for simulation is the Raccoon watershed, located in the state of Iowa in the United States. The Raccoon River is the main stream for the watershed and drains a large area containing an abundance of fertile soil. The total area of the watershed is approximately 2.3 million acres, 1.7 million of which are devoted to rotations of corn and soybean production. Nitrogen and phosphorus fertilizer are applied at high levels on the corn crop and constitute the primary nonpoint nutrient pollutant source in the watershed. Figure 1 shows a land-use map of the watershed. The outlet of the watershed is near the capital city of Des Moines, which along with other municipalities in the area, uses the Raccoon River as a source of drinking water. The Des Moines Water Works is the supplier of drinking water and currently operates the world's largest denitrification facility.

In-stream nitrate levels frequently exceed the maximum allowed concentration of 10 milligrams per liter. In these instances, source water is run through the denitrification facility before being treated for use as drinking water. The facility uses an ion exchange process which produces waste water with a high saline content in addition to the nitrate removed. This waste water is currently discharged downstream at no cost to the facility. Downstream municipal water supplies are not adversely impacted by this discharge, as they are able to meet their water needs from deeper ground water aquifers. For purposes of NPDWR compliance this is not an issue, and the discharge is permitted by the EPA under the National Pollutant Discharge Elimination System.

The nitrate removal facility was constructed in 1990 at a cost of approximately $3 million. The scrubbers and media were the primary components of this large sunk cost, and would also be the bulk of the cost associated with

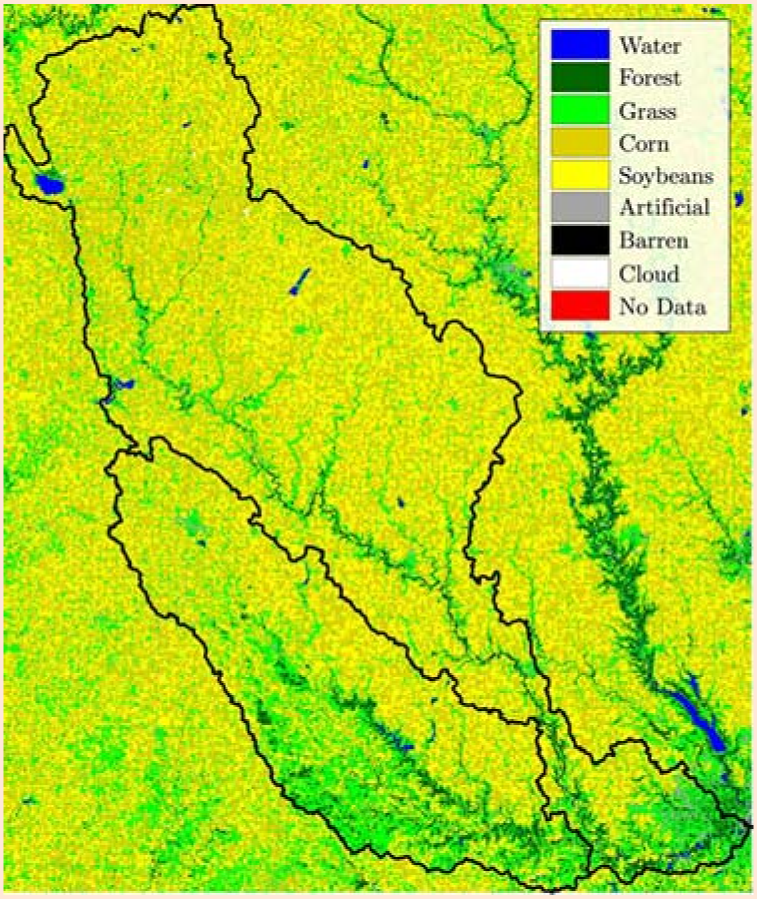

Figure 1. Land use in the Raccoon watershed.

an expansion of the facility unless another removal technology were employed. Current processing volume does not appear to require expansion in the near term, and there has been no observed deterioration of the scrubber components. Operating costs of the facility are approximately \$300 per million gallons of water, with a capacity of 10 million gallons of water per day. In an average year, the facility runs approximately 50 days.

Modeling Approach

While drinking water standards are given high importance due to their direct effects on human health, high nitrate levels cause other problems. However, control of ambient water pollution in this watershed is still being developed, and there are no existing regulations outside of drinking water standards. Ameliorating problems such as hypoxia and nitrate toxicity for aquatic animals would require both a lower threshold for nitrates and complete removal of the nitrate from the watershed. Meeting the latter requirement with the technology currently used for drinking water purposes is inappropriate as it reintroduces the nitrate to the environment. The analysis here proceeds in the framework of existing regulations and the technology currently in place, but it is important to note that there are other impacts that merit consideration: namely, the effects of nitrate levels outside of drinking water considerations. Upland fertilizer reductions prevent nitrates from entering waterways in the first place, and have positive effects beyond contributing to drinking water standard compliance.

The goal of the modeling framework is to capture changes in water quality generated by implementation of policy, as well as the associated economic effects. This requires the coupling of an economic model with a physical model. Nutrient application levels predicted by the economic model are used to supply land-use inputs to the physical model. The output from the physical model in turn provides the water quality measure of interest: nitrate concentration over time. A hydrologic model is used to link the effects of upland fertilizer reductions to direct nitrate removal at the outlet. The watershed-based Soil and Water Assessment Tool (SWAT) simulates the effects of watershed management on water quality and water flow on a daily time step. It is primarily used for modeling nonpoint source contributions to nutrient and sediment loads within a watershed. The SWAT implementation employed uses data from the National Resources Inventory (NRI) to populate the watershed with spatially detailed information. A point in the NRI effectively represents a farm. Site-specific nutrient application data are generated by the economic model. The economic model predicts nitrogen fertilizer application rates based on prices of corn and fertilizer and a site-specific soil characteristic. It also predicts yield, and thus returns to fertilizer application. Changes in nitrogen fertilizer prices, for example, via a tax on fertilizer or a cap on application, will cause a loss in returns for the farmer. This provides a measure of the cost imposed by the policy. Data used to construct the model comes from a

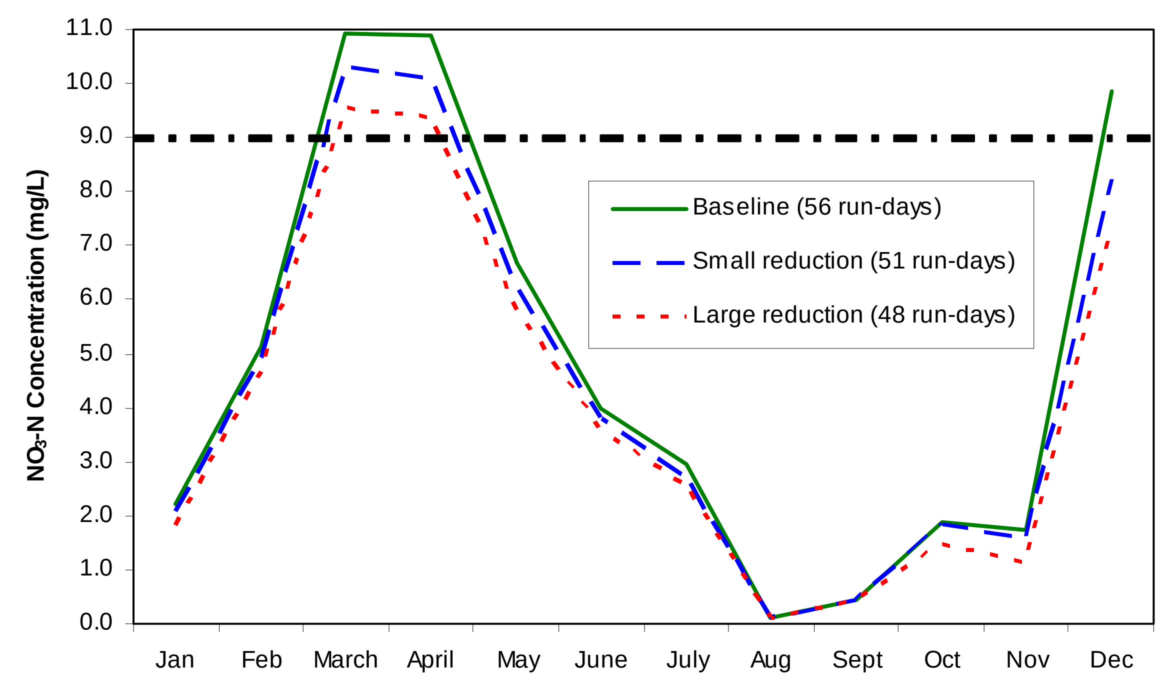

Figure 2. Nitrate loads by monthly average.

farm operator survey, the Agricultural Resource Management Survey, historical prices, and from a detailed soil grid.

Policy Simulations

Three scenarios are run through the modeling system described above. One is a baseline in which the economic model leaves prices and nitrogen fertilizer applications unchanged, and the water quality model predicts the associated nitrate concentrations at the watershed outlet. The other two scenarios represent reductions in fertilizer applications simulated by the imposition of a nonpoint source trading scheme. This scheme works as follows: each farm is allocated fertilizer application permits for the total acreage it farms; for example, a 100-acre farm might receive 12,000 pounds worth of permits if the permit level is 120 pounds per acre. A farm has three choices in using its permits. One is to apply exactly the permitted amount. Another is to apply less than permitted, and sell the surplus permits to the third group, those who purchase permits in order to apply at greater levels than initially permitted. Farmers make their choice of total application according to the model, taking into account the prices they face, their soil type, and the market price of a permit, which is determined by the distribution of farmer types. The total watershed application is reduced as long as the total permit allocation is smaller than the total amount originally applied. For purposes of simulations, this is done at two levels of permit allocations. From a baseline average application rate of 135 pounds per acre, one scenario restricts the per-acre permit allocation to approximately 120 pounds per acre and results in a simulated 6% reduction in annual load of nitrate at the watershed outlet. The other restricts the allocation to approximately 108 pounds per acre and results in an approximate 12% reduction in annual nitrate load. These reductions are the result of the total mass of nitrogen being applied in the watershed being reduced.

Imposing the permit restrictions benefits those farmers who can sell excess permits, but increases the costs of those who must purchase additional permits. Since the total amount of nitrogen application is being reduced, the net result is a loss for farmers in the watershed as a whole. Loss or gain from the policy scenarios can be measured for individual farms and then aggregated to the watershed level to gauge the cost of the policy. Under the small reductions, the total farm watershed loss is approximately \$161,000, and under the larger reductions, losses are approximately \$700,000.

To compare the water quality changes resulting from the implementation of these policies to operation of the nitrate removal facility, the water-quality model is run on a daily time step and the nitrate concentration for each day recorded. The trigger concentration for the nitrate removal facility to run is 9mg/L (the legal limit is 10mg/L). Under the baseline scenario, that level was exceeded 56 days of the year. The small and large reduction scenarios reduced the number of run-days to 51 and 48. Figure 2 shows a summary of nitrate loads by monthly average. Saving days of operation for the nitrate removal facility implies cost savings and illustrates the shortening in the number of run-days required to maintain a safe level of nitrate. The energy, labor, and raw material costs of one run-day are approximately $3,000. The lifetime of the media used in the removal process is currently uncertain, making it difficult to calculate the true cost of operation. The original media is still in use and shows no sign of deterioration after 14 years of use. As nitrate loads and water demand grow, there may be a need for expansion in the future, involving significant capital costs and raising the cost of a day of operation. Such expansion may also involve a change in nitrate removal technology.

Trading nitrogen permits between point and nonpoint sources can lower costs of reductions (Randall & Taylor, 2000). This is usually considered in the context of a nonpoint source generating excess permits by purchasing upland reductions. In that type of trading arrangement, a trading ratio is established to equilibrate a pound of upland reduction to a pound of point source discharge. Conceptually, this approach could work in reverse as well: nonpoint sources could generate permits for themselves by paying for the removal system. While these trading opportunities are attractive possibilities, a quick look at the difference in costs in this case suggests that it would be much more efficient to simply run the nitrate removal facility a few extra days rather than implement any restrictions on farmer application. Five run-days at $3,000 per day is $15,000, far less than the $160,000 in losses that would be incurred by farmers. Eight run-days of the nitrate removal facility are likewise much less expensive than the $700,000 in losses associated with the stricter cap-and-trade policy.

While the upland fertilizer reductions examined here are more costly than direct nitrate removal, this analysis does not take into account other possibilities. There are also concerns beyond drinking water standards, such as hypoxia and low-level nitrate toxicity (Camargo et al., 2005), that have important impacts on ambient water quality. Perhaps because drinking water issues pose the most immediate threat to human health, it is the only form of existing pollution regulation that impacts this watershed. As new standards with broader impacts in mind are developed, such as Total Maximum Daily Loads, this analysis can be revisited, possibly with different conclusions. The upland reductions have an effect on the ambient and downstream nitrate loads that the removal process does not and would be more effective at meeting expanded standards. Even if under more comprehensive standards upland reductions become more cost effective, there would be transaction costs involved in any trading scheme that would also need to be considered.

There are also combinations of reduction strategies that could result in superior reductions with similar costs, even in the existing framework. Coupling reductions with buffer strips, grassed waterways, changes in tillage, and application timing all can contribute to reductions in nutrient loads to a watershed. In addition, a more complete comparison would require information on possible deterioration of the nitrate removal media and the associated replacement costs, though these are at present uncertain.

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